RESOURCE SELECTION, MOVEMENT PATTERNS, AND SURVIVAL OF POST-FLEDGING GRASSLAND BIRDS IN MISSOURI

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by

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The undersigned, appointed by the Dean of the Graduate School,
have examined the dissertation entitled

RESOURCE SELECTION, MOVEMENT PATTERNS, AND SURVIVAL OF

POST-FLEDGING GRASSLAND BIRDS IN MISSOURI

Presented by Kimberly M. Suedkamp Wells

A candidate for the degree of DOCTOR OF PHILOSOPHY

And hereby certify that in their opinion it is worthy of acceptance.

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RESOURCE SELECTION, MOVEMENT PATTERNS, AND SURVIVAL OF POST-FLEDGING GRASSLAND BIRDS IN MISSOURI

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ABSTRACT

Current grassland management paradigms focus on addressing nesting requirements for grassland birds, but ignore post-fledging requirements. I described resource selection, movement patterns, and survival of dickcissels (*Spiza americana*) and eastern meadowlarks (*Sturnella magna*) in southwestern Missouri, from 2002 to 2004 using radio telemetry.

Across species at the micro-scale, there was support for the predation hypothesis (both species) and the thermal refuge hypothesis (dickcissels only). Woody cover (both species) and vegetation height (meadowlarks) had the highest relative importance across years. At the landscape scale, uniformly shrubby prairies, longer distances to forests, roads, and grazing were negatively associated with juvenile dickcissels. Crops, pastures, increasing distances to ponds and streams were positively associated with juvenile meadowlarks. Core home range sizes (50%) were similar across species, but 95% home ranges were 25% larger for meadowlarks (80.9 ± 13.9 ha) than dickcissels (51.2 ± 8.8 ha). Home range patterns were mostly non-linear and categorized as central or exploratory. Across years, biological factors (number of siblings, order of fledging) were the best predictors of home range size. Survival was higher and the instantaneous probability of death declined faster for meadowlarks compared to dickcissels. My results indicate that our working concepts of suitable breeding habitat need to be modified to accommodate post-fledging requirements and maximize the effectiveness of conservation strategies.

CHAPTER 1

MICRO-SCALE RESOURCE SELECTION OF POST-FLEDGING GRASSLAND BIRDS IN MISSOURI

ABSTRACT

Current grassland management paradigms focus on addressing nesting requirements for grassland birds, but ignore post-fledging requirements. The resulting effect is that management strategies for grassland birds may not be effective because they assume post-fledging requirements will be satisfied while managing for nesting requirements. We evaluated micro-scale resource selection patterns in two species of grassland birds during the post-fledging period in southwestern Missouri from 2002 to 2004. We used an Information Theoretic approach and constructed *a priori* models associated with three hypotheses related to predation, starvation, and thermal refuge to explain resource selection patterns in juvenile birds. Each year, we iteratively revised our candidate models by incorporating our observations and new information in the literature. Candidate models associated with the Predation Hypothesis included variables related to predator hiding places or travel corridors such as woody patches and roads. Candidate models associated with the Starvation Hypothesis included variables related to food substrates such as forb and grass cover. The Thermal Refuge Hypothesis contained candidate models relating to temperature and interactions with habitat features that might moderate high temperatures. We located nests of both species and attached 0.7-gram radiotransmitters to nestlings one to three days prior to fledging. From May to August in 2002 to 2004, we obtained a minimum of 30 detections on 74 individual dickcissels

(*Spiza americana*) and 64 eastern meadowlarks (*Sturnella magna*) using radio telemetry. We modeled resource selection for each species using logistic regression. There was support for the Predation (two years) and Thermal Refuge Hypothesis (one year) for juvenile dickcissels and inconsistent support for the Predation Hypothesis (one year) for eastern meadowlarks. Model fit, as indicated by concordance valued was adequate across two years for dickcissels indicating micro scale variables were relevant for this species. However, model fit was poor for two of the three years included for meadowlarks, which indicates micro scale variables were not as relevant. Across year, percent woody cover had the highest relative importance for dickcissels. Woody cover and vegetation height had high relative importance values across years for eastern meadowlarks. Woody shrubs and draws were important hiding places for juvenile dickcissels and perches for eastern meadowlarks. Our results demonstrate that the nesting and post-fledging requirements are slightly different and that woody features may play an important role during the post-fledging phase. Future conservation efforts need a more complete understanding of the entire breeding cycle to effectively increase grassland bird populations.

INTRODUCTION

Declining trends among grassland birds are well-documented in the literature (Knopf 1994, Sauer et al. 2004, Rich et al. 2004). Despite these declines, the majority of ornithological research has historically focused on forest birds (Askins 1993). Although research interest in grassland birds has increased during the last two decades (Vickery et al. 1999), the majority of efforts are focused on breeding season requirements (Vickery et al. 1999, Vickery and Herkert 2001). Given that 40% of the species on the Partners in

Flight Continental Watchlist breed in the prairie biome (Rich et al. 2004), future conservation efforts need to focus more attention on this group of threatened species. Within the breeding season, researchers need additional information on the post-fledging period because it comprises half of the entire breeding season (Faaborg et al. 1995). Although several authors have described post-fledging requirements for wood thrush (*Hylocichla mustelina*) (Anders et al. 1997, Vega Rivera et al. 1998, Powell et al. 2000, Lang et al. 2002, Fink 2003) and Swainson's thrush (*Catharus ustulatus*) (Gardali et al. 2003), information on the post-fledging needs of grassland birds is sparse (Kershner 2001, Yackel Adams 2001).

Understanding post-breeding habitat requirements of migratory birds is a critical conservation and management need that has been largely ignored (Vega Rivera et al. 1999). Information on the post-fledging period, defined as the time period between obtaining independence from parents (i.e. leaving the nest) and migration (Vega Rivera et al. 1999), is largely unknown (King and Belthoff 2001). The post-fledging period is a crucial life history stage for juvenile birds as they complete the prebasic molt and begin to build fat reserves for migration while trying to avoid predators (Moore 1993, Vega Rivera 1998). Despite the importance of the post-fledging period, it is widely regarded as the least understood part of the avian life cycle (Part 1990, Morton 1991, Baker 1993, King and Belthoff 2001). Knowledge about habitat use, movement patterns, and survival during the post-fledging period would provide critical information about productivity levels needed to sustain local populations (Anders et al. 1997).

Knowledge of post-fledging requirements for grassland birds is particularly important because it coincides with periods of intensive habitat management. The current paradigm in grassland management focuses on delaying management practices until mid-summer, which avoids peak nesting periods for the dominant species on our sites (Lanyon 1995, Vickery 1996, Ehrlich et al. 1998, Winter 1999). However, this paradigm is problematic because it focuses only on the first half of the nesting cycle and ignores the post-fledging period. If nesting and post-fledging requirements are different (Anders et al. 1997), then a more inclusive paradigm may be appropriate. For example, the current paradigm embodied by the Bird Conservation Area (Winter et al. 2000a) concept proposed by the Midwest Working Group of Partners in Flight (Pashley and Fitzgerald 1996) recommends eliminating woody cover in hostile landscapes to increase reproductive rates. However, preliminary tests of the BCA concept (Winter et al. 2000a) have not provided consistent evidence to support the eradication of woody species. Although woody species are only one example of a habitat feature that may play different roles during the nesting and post-fledging phases, the lack of information suggests rigorous evaluations of potential differences are necessary and timely.

Our goal was to characterize resource selection during the post-fledging period for juvenile dickcissels (*Spiza americana*) and eastern meadowlarks (*Sturnella magna*) in southwestern Missouri. We selected both species because their populations have been declining nationwide (Sauer et al. 2003), they are abundant on our study sites, and their body size (> 40 g) facilitates the use of transmitters for time periods long enough to address resource selection during natal dispersal. Both species are omnivorous during the breeding season and nest in grasslands (Harmeson 1974, Lanyon 1995, Ehrlich et al. 1998). However, dickcissels typically nest in low shrubs on our sites including persimmon (*Diospyros virginiana*) and smooth sumac (*Rhus glabra*); whereas, eastern meadowlarks favor grass and litter clumps. Both species also utilize elevated perches such as fence rows, trees, shrubs, and power lines.

METHODS

Study Sites

We conducted this study at Taberville Conservation Area (38° N, 93° W) and Wah'Kon-Tah Prairie (37°N, 94° W) in Cedar and St. Clair Counties in southwestern Missouri. Taberville Conservation area is a 680-ha prairie owned and managed by the Missouri Department of Conservation (MDC) and embedded in an agricultural matrix of crops (wheat, soybeans, and corn) and private land. Wah'Kon-Tah Prairie is a 1930-ha prairie owned by the Missouri Chapter of The Nature Conservancy (TNC) and jointly managed by MDC and TNC located at the northern periphery of El Dorado Springs, Missouri (population ~ 4,000 people). Both sites are part of a network of focal areas targeting grassland bird conservation in Missouri. Dominant land management practices included livestock grazing, prescribed burning, seed harvesting, and haying. Woody removal is a minor management practice that occurs along woody draws, fence lines, and pasture borders. The study sites are divided into management units that receive some sort of management practice (primarily prescribed burning or haying) at least once every three years. Dominant vegetation was composed of bluestem grasses and included big bluestem (*Andropogon gerardi*), little bluestem (*Schizachyrium scoparium*), and indian grass (*Sorghastrum nutans*). Forb species included coneflowers (*Echinacea* spp.), white wild indigo (*Baptisia alba*), blazing star (*Liatrus* spp.), compass plant (*Silphium laciniatum*), milkweeds (*Aesclepias* spp), and sunflowers (*Helianthus* spp.). Dominant, native woody species included smooth sumac (*Rhus glabra*), persimmon (*Diospyros virginiana*), blackberry (*Rubus* spp.), and roses (*Rosa* spp.).

Bird Capture and Handling

We located nests of both species using systematic searches and haphazard walks from 0600 to 1400 hours each day between the third week of April and the second week of August from 2002 to 2004. When possible, we used behavioral cues of the parents to indicate the presence of a nest nearby. After locating each nest, we recorded the GPS coordinates and marked the location by placing colored flagging tape at least 5 m away. At each nest we recorded the species, content, parental activity, and presence of any non-host eggs. If the nest contained nestlings, we attempted to age the nestlings using the presence of down, whether the eyes were open or not, the extent of pin feather development, or the presence of a full complement of feathers. Based on our observations, we were usually successful at aging nestlings within two days of their true age depending on growth rates and weather conditions. We monitored each nest every three to four days until just prior to fledging and then switched to daily nest checks. Two to three days prior to fledging, we attached a metal USFWS band to the left leg and a unique combination of plastic, UV-resistant Darvic bands (Avinet, Dryden, New York) to the right leg and weighed each individual.

Following a modification of the Rappole and Tipton method (1991) previously evaluated (Suedkamp Wells et al. 2003), we attached 0.7-gram transmitters with a 10-cm whip antennae (Biotrack, Dorset, United Kingdom) to the back of each bird using a leg harness. Battery life for each transmitter was expected to range between 55 and 60 days. We constructed the leg harness from cotton, elastic beading cord to allow room for growth. Using super glue (Duro, Avon, Ohio), we secured the bottom of the transmitter to the back of the bird. After attaching transmitters to each bird, we placed the brood back in the nest. Handling and processing time usually was between 2 and 5 minutes per bird.

Nest Success Calculations

We calculated daily nest survival estimates following Mayfield (1975) and used the modification for standard errors from Johnson (1979). We classified the fate of each nest as successful or unsuccessful based on our observations of parental behavior, the presence of fledglings, and the condition of the nest. The majority of nests in our sample had at least one chick with a transmitter, so we were comfortable classifying nest fates for this portion with little uncertainty. For the small proportion of nests in our sample without transmitters on any chick, we confirmed the fate of each nest using parental feeding behavior, begging vocalizations of fledglings, and the presence of fecal sacs in the nest. We classified nests as successful if parents were observed bring food to the nest area or we could hear fledglings begging. If we were uncertain about the date of an event (depredation for example), we used the last known observation with confirmation of eggs or nestlings to calculate exposure days. Nests that fledged at least one host chick were classified as successful. We used nesting periods of 19 days and 26 days for dickcissels and meadowlarks, respectively. Nesting intervals were based on data provided in Lanyon (1995), Ehrlich et al. (1998), and on our observations of the earliest day a nestling was observed fledging (day seven for dickcissels, day 11 for meadowlarks).

Radiotracking

Using telemetry, we began tracking birds using homing for visual confirmation (Mech 1983) the day after attaching transmitters. If the brood remained in the nest the day after attachment, we returned each subsequent morning and began tracking when at least one brood member fledged. We tracked each bird twice daily in non-consecutive time blocks for a minimum of 50 detections per individual for a total of 25 individuals of both species (Garton et al. 2001). The four tracking blocks were early morning (0600 to 0930 hours), mid-morning (0930 to 1230 hours), afternoon (1230 to 1700 hours), and evening (1700 to 2130 hours). We grouped the time blocks to reflect biological activity and environmental constraints such as hot temperatures when activity is reduced. The first day of tracking for an individual occurred in the early morning and afternoon periods. On the second day, tracking occurred in the mid-morning and evening hours. We continued to alternate days on this schedule to capture locations representative of all diurnal activities (Garton et al. 2001). We avoided tracking before 0600 hours and after 2130 hours to reduce the risk of mortality when juvenile birds could not be visually located. After reaching 50 detections, each individual was tracked once daily alternating between the first two and last two periods of the day until the bird died, the transmitter was recovered, or the study period ended.

We tracked each individual until we recovered the transmitter or dead bird through the end of August in each field season. Immediately after being unable to locate an individual, we performed extensive searches of the immediate area on foot with a team of assistants. If we were unable to locate the individual, we broadened the search to include all roads within 3.2 km of the last known location using an omni antennae mounted on the roof of a truck. We continued to search for missing individuals twice daily for one week after their disappearance. At the start of the second week, we reduced search time to one attempt per day. We also attempted to locate missing birds by flying at least 5 km strips over the study area in a helicopter twice monthly between 1 June and 30 August each year. If we recovered a transmitter, we recorded a description of the recovery site (e.g. burrow or pond), condition of the transmitter (presence of teeth marks or snake feces), and any other information that could be used to identify the potential predator or cause of death.

Resource selection measurements

At each bird location and a paired, random point, we recorded the GPS coordinates, presence of parents and siblings, and a suite of micro-scale habitat variables. We selected the direction of a paired, random point using the quadrant of the minute hand on a watch. Likewise, we selected the distance (in paces) to the paired, random point using the second hand of a watch. Paired random points were within 60 paces of the used point where the bird was located. Habitat variables included ambient temperature (°C, except 2002), primary substrate, functional group composition (% grass, litter, forb, and woody species), plant height (cm), concealment and shading indices (except 2002; 1 = < 25%, 2 = 26 – 50%, 3 = 51 – 75%, and 4 = > 75%), the distance to woody plants within 1 and 5 m and the species of plant, distance to the nearest raptor perch (m) that was at least 2 m in height, distance to the nearest woody patch that was at least 3m in diameter, and patch type (fence row, draw, bush clump, tree clump, pond border, or forest). With the exception of distance variables, we measured all habitat features within a 2 x 5 dm Daubenmire frame (Daubenmire 1959) centered on the bird location. We measured distances to the nearest woody plant with a tape measure and distances to raptor perches and woody patches with a laser rangefinder (Bushnell Yardage Pro Scout, Lenexa, Kansas). During the first two weeks out of the nest, we avoided taking habitat measurements (with the exception of ambient temperature) until the juvenile bird could fly and was not at risk of being stepped on. We returned immediately after the bird could fly to measure habitat features at used and random points. Typically, the time lapse did not exceed two weeks which was sufficient to prevent substantial changes in vegetation structure or functional group composition.

A Priori Hypotheses and Model Building

Prior to conducting field work, we identified two hypotheses (Predation and Starvation) to explain resource selection patterns at the micro scale. After one year of data collection, we added a third hypothesis (Thermal Refuge) based on our observations. The Predation Hypothesis states that juvenile birds select resources at the micro scale to avoid risk of predation. The Starvation Hypothesis states that juvenile birds are selecting resources at the micro scale to maximize foraging opportunities. The Thermal Refuge Hypothesis is an extension of previous work (Suedkamp 2000, Lusk et al. 2003) that stated that juvenile birds select resources that minimize the intensity and duration of their exposure to temperatures that induce heat stress. To construct *a priori* models under each hypothesis, we used single variables (Appendix 1) and two-variable interactions that were biologically important and interpretable. Based on our observations and preliminary data analysis each year, we revised candidate models iteratively to incorporate improved knowledge about each species. For example, after our first field season in 2002, we analyzed our data using the *a priori* models provided in Appendices 2 – 4. Based on those results and our observations, we often revised candidate models for the next year so that they included combinations most likely to explain resource selection. We list model descriptions, structures, and expected results for each species and year in Appendices 2 – 7. For example, candidate models related to the Predation Hypothesis included variables related to potential predator perches (for raptors), hiding places (e.g. woody patches), or travel corridors (roads). Candidate models related to the Starvation Hypothesis included habitat features that are associated with substrates used by arthropod food sources such as litter, grass, or forb cover and the number of siblings competing for food sources. Finally, candidate models under the Thermal Refuge Hypothesis are based on a combination of variables relating to ambient temperature, variation of temperatures within a day or season, and interactions with habitat features that may moderate heat stress or provide shade such as woody cover.

Resource Selection Analysis

To model resource selection at the micro scale, we used individuals of both species with ≥ 30 detections and pooled within species and year. We analyzed each year separately for each species to determine if patterns of resource selection were consistent across years. We screened each variable using a paired *t*-test to reduce the number of variables in the next modeling stage. Any variable that was significant ($P < 0.25$) was retained for further analyses (Hosmer and Lemeshow 2000). We also tested for multicollinearity using PROC REG in SAS (SAS Institute 2001) and removed any variables with tolerance values < 0.40 (Allison 1999). Finally, we checked for normality using probability plots in SYSTAT (SPSS 1999) and applied an appropriate transformation if necessary (Steel et al. 1997).

We used an Information Theoretic approach (Burnham and Anderson 1998) to fit resource selection models for each year using generalized linear mixed models in SAS (SAS Institute 2001). As part of the Information Theoretic Approach, we evaluated support for each of the three hypotheses and associated candidate models that we have proposed (Predation, Starvation, and Thermal Refuge). Generalized linear mixed models are a class of mixed models that allow for random and fixed effects (Littell et al. 1996). We suspected that random effects would be more appropriate than fixed effects for site and brood to enable inference to other sites and populations and to correctly calculate the coefficient standard errors with correlated data. To evaluate whether random effects were appropriate, we fit resource selection functions with random effects of site and brood for each year using the GLIMMIX macro in SAS. We used the lowest AIC score to infer whether random or fixed effects were appropriate for both effects. If random effects were not appropriate (AIC value higher than fixed effects), we fit resource selection models in PROC GENMOD in SAS. Next, we determined the appropriate covariance structure by using AIC to rank several structures including compound symmetry, auto regressive, and unstructured, as recommended by Littell et al. (1998).

Using a two-stage approach modified from several authors (Franklin et al. 2000, Washburn et al. 2004), we evaluated support for each hypothesis using Likelihood Ratios tests and model selection. In the first stage, we assessed model fit by comparing the sub global model associated with each of the hypotheses (predation, starvation, thermal refuge) to a null model using likelihood ratio tests. Sub global models and their associated candidate models were retained if the sub global model fit the data ($P < 0.05$). During the second stage, we used model selection to evaluate support for the remaining sub global models associated with each hypothesis and their candidate models. We ranked candidate models using the AIC value and calculated the weight of evidence (w_i) across all models in the data set. If there was evidence of model uncertainty ($w_i < 0.90$), we used model averaging to calculate parameter estimates across candidate models with support (within 7 AIC units of the best model) (Burnham and Anderson 2002). In addition to assessing model fit by comparing sub global models to the null model using likelihood ratio tests, we assessed predictive power using the percent correct classification to generate concordance values using the LACKFIT option with PROC LOGISTIC in SAS (Allison 1999).

For years with good model fit (LRT $P < 0.05$) and adequate concordance values (> 60%), we present the odds ratios and 95% confidence intervals along with parameter estimates and their standard errors. To assess the influence of each variable across all three years in the data set, we calculated relative importance values for each year and species combination for all models regardless of fit and predictive ability. However, we stopped at this step and did not continue assessing effect size (using odds ratios) or the direction of the relationship (using parameter coefficients) for models with poor fit and predictive ability. Finally, we illustrate the role of variables with the highest relative importance across years and those variables that are most biologically interpretable (grass cover, litter cover, forb cover, woody cover, and vegetation height) using summary graphs that show mean values at used compared to available points within each year. To illustrate the entire model development and evaluation process, we have included a flow chart (Fig. 1).

RESULTS

We located 258 dickcissel nests and 113 eastern meadowlark nests between 2002 and 2004 (Table 1). Eastern meadowlark nests were located mostly during the incubation (45%) and brooding stages (36%); whereas, dickcissel nests were mostly located during the laying stage (51%). Estimates of daily nest survival were identical across species although survival during the entire nesting interval differed (Table 1). We used 74 juvenile dickcissels from 47 broods and 64 juvenile eastern meadowlarks from 32 broods with > 30 detections in the resource selection analysis (Table 2).

Due to problems with missing data and multicollinearity (tolerance < 0.10), we dropped the distance to woody plant variables (within 1 m and between 1 and 5 m) from the data set. The amount of woody cover was non-normally distributed so we transformed the data using a log function (Steel et al. 1997) for percent data. The cover and shade indices showed evidence of multicollinearity (tolerance <0.20) so we added them for a combined index which alleviated the problem (tolerance > 0.60).

Dickcissels

In 2002, the Predation Hypothesis sub global model (see Appendices 2 – 4 for models fit each year) fit the dickcissel data adequately (LRT $P < 0.01$) and showed acceptable predictability (59% concordance) but the Starvation Hypothesis sub global model fit poorly (LRT $P = 0.21$) and had low predictability (concordance 47%) (Table 3), so it was not included in the second stage of model selection. Models with random effects of site and brood were no better than models with fixed effects because the AIC values were identical (AIC = 2589). The model with an interaction between woody cover and vegetation height associated with the Predation Hypothesis was the best model (w_i =

0.94, Table 3). The resource selection function for dickcissels in 2002 was associated with the Predation Hypothesis and included an interaction between woody cover and vegetation height (Table 4). Woody cover was the best predictor in the data set (relative importance = 0.99, Table 5).

In 2003, the sub global models associated with all three hypotheses fit adequately (all LRT $P < 0.01$) and showed acceptable predictability (64 to 75% concordance, Table 6), so all sub global models and their candidate models were included in the second stage of model selection. Models including random effects of site and brood were better (AIC = 2176) than models with fixed effects (AIC = 2226). There was model uncertainty associated with the 2003 resource selection models for dickcissels (all candidate models $w_i < 0.90$, Table 6), so we used model averaging to calculate parameter coefficients and standard errors over models with some support (ΔAIC < 7). The resource selection function for dickcissels in 2003 included models associated with the Predation and Thermal Refuge Hypotheses (Table 7). Woody cover was the best predictor in the data set (relative importance = 1.00, Table 5).

In 2004, the sub global models associated with all three hypotheses fit the dickcissel data adequately (all LRT $P < 0.01$) and showed good predictability (71 – 77% concordance, Table 8). Models including random effects of site and brood were better (AIC = 3030) than models with fixed effects (AIC = 3040). The best model for dickcissel resource selection in 2004 was the Predation Hypothesis sub global ($w_i = 1.00$, Table 8). The resource selection function for dickcissels in 2004 is shown in Table 9. Woody cover was the best predictor across the data set (relative importance = 1.00, Table 4). The amount of woody cover at used points was consistently higher compared to paired, random points, especially five days post-fledge (Fig. 2). In Figs. 3a - 7a, we show mean values of grass cover, litter cover, forb cover, woody cover, and vegetation height for each year at used and available points.

Meadowlarks

In 2002, the sub global models associated with the Predation and Starvation Hypotheses (see Appendices 5 – 7 for models fit each year) fit the meadowlark data poorly (all LRT $P > 0.48$) and predictability was low (concordance = 50 – 52%, Table 10). Models with random effects of site and brood were better (AIC = 1000) than models with fixed effects (AIC = 1113). Vegetation height and grass cover were the best predictors across the data set (relative importance > 0.34 each, Table 5).

In 2003, sub global models associated with all three hypotheses fit the data poorly (all LRT $P > 0.20$) and predictability was low (51 – 52% concordance, Table 11). Models with random effects of site and brood were better (AIC = 4140) than models with fixed effects (AIC = 4141). Woody cover, the cover and shade index, and vegetation height were the best predictors across the data set (relative importance > 0.13, Table 5).

In 2004, sub global models associated with all three hypotheses fit the data adequately (all LRT $P < 0.01$) and showed good predictability (62 – 70% concordance, Table 12). Models with random effects of brood and site were better (AIC = 3128) than models with fixed effects (AIC = 3314). The sub global model associated with the Predation Hypothesis was the best model (w_i = 1.0; Table 12) and the resource selection function is shown in Table 13. Woody cover was the best predictor across the data set (relative importance = 1.0, Table 5). In Figs. 3b - 7b, we show mean values of grass cover, litter cover, forb cover, woody cover, and vegetation height at used and available locations each year.

DISCUSSION

Our goal was to evaluate support for three hypotheses to explain resource selection patterns and the consistency among those patterns for two species of grassland birds during the post-fledging period. The Predation Hypothesis had consistent support across all years to explain resource selection patterns in dickcissels. In addition, the predictive power of those models, as indicated by the concordance values, show the micro scale is useful for understanding habitat selection in dickcissels. Dickcissels have relatively small territories ranging from 0.15 to 0.45 ha (Harmeson 1974), so it is not surprising that the micro scale was relevant. However, the same scale may not be as relevant to eastern meadowlarks. Their territories range from 1.2 to 6.1 ha in size (Lanyon 1995) and we frequently observed males defending territorial boundaries much larger in size (K. Suedkamp Wells, unpubl. data). Using micro scale measurements in this study combined with smaller sample sizes (Table 2) during the first year may explain some of the inconsistency we documented in resource selection by eastern meadowlarks. For example, 2004 was the only year that we observed adequate fit of the sub global models associated with each hypothesis and good predictive ability as indicated by the concordance values. However, sample sizes for the number of individuals and the number of detections per individual meadowlark between 2003 and 2004 are comparable so we believe that larger scales are more useful for explaining resource selection patterns in this species. In Chapter two, we describe the utility of landscape-scale resource selection functions for both species.

Across years, there was consistent support for the Predation Hypothesis and partial support (2003 only) for the Thermal Refuge Hypothesis to explain resource selection in dickcissels. Our results highlight the importance of predation as a selective factor shaping juvenile resource selection and are consistent with other research on other post-fledging birds (Sullivan 1989, Zann and Runciman 1994, Anders et al. 1997, Fink 2003). For example, mortality attributed to predators during the first 15 days of the pre-independence period ranged from 60 – 70% across species in this study (K. M. Suedkamp Wells, unpubl. data), which is comparable to other studies. Anders et al. (1997) reported a survival probability of 0.716 in Wood thrush (*Hylocichla mustelina*) for the first three weeks post-fledge. Zann and Runciman (1989) reported that 67% of Zebra finches (*Taeniopygia guttata*) were lost during the first 35 days post-fledge, presumably due to predators. The association of juveniles with increased woody cover shown in our results (Table 4, Fig. 1) and those of others (Anders 1996, Fink 2003) is likely a response to avoid predation risk in more open habitats. In this study, juvenile dickcissels often moved to woody draws and corridors where detection by snakes, the dominant predator of both species on our sites (K. Suedkamp Wells, unpubl. data), may be reduced. Although we frequently observed snakes moving through grassland areas, we rarely observed snakes in or near woody corridors except the Black rat snake (*Elaphe obsoleta*). Anders (1996) reported a similar trend by juvenile Wood thrush in Missouri forests that showed movement into clearcuts, forest edges, and thick riparian corridors. Fink (2004) also reported that shrub density was one of the best predictors of the presence of Wood thrush juveniles in Missouri forests, which he attributed to predator avoidance behavior. Depredation patterns of other dominant snake species on our site, including the prairie king snake (*Lampropeltis calligaster*) (see Chapter 4), were more frequent in field habitats compared to forest habitats in Missouri which may explain movement patterns if they are to avoid predators (Thompson and Burhans 2003).

Although our results only show support for the Thermal Refuge Hypothesis in one year (2003) for dickcissels, we believe the role of microclimate on post-fledging resource selection needs additional attention. In open habitats, such as grasslands, juvenile birds may be more affected by microclimate conditions due to reduced shading or opportunities to seek shelter. During the pre-independence period, relatively immobile juveniles may not be able to escape temperatures over 39°C, which is the point at which most birds are unable to balance heat gain with cooling measures (Webb 1987). We observed one example where woody removal using a brush hog resulted in the juvenile succumbing to heat stroke within a few hours of afternoon sun in July due to exposure (K. Suedkamp Wells, unpub. data). Although the role of microclimate factors on site selection and nest success has been described for a few species in open habitats (With and Webb 1993, Nelson and Martin 1999, Suedkamp 2000, Lusk et al. 2003), potential effects on post-fledging juveniles have not been addressed. Increased use of woody cover by juvenile birds during the post-fledging period also may be related to seeking thermal refuge, but this possibility has not been addressed to our knowledge.

Although patterns of resource selection for eastern meadowlarks were inconsistent, two themes were evident. One is that woody cover had high relative importance values across years and was positively associated with the presence of meadowlarks in the only year where model fit was adequate (2004). The importance of woody cover for juvenile meadowlarks may seem surprising because this species is not typically associated with woody cover but rather grass cover (adults, Lanyon 1995; juveniles, Kershner 2001). Nest placement was rarely associated with woody cover, but males often showed a preference for shrubs at the perimeter of territories for defense purposes and females often utilized the same shrub perch prior to landing on the ground and walking towards the nest to deliver food or parental care (K. Suedkamp Wells unpub. data). However, our data are not consistent with the dominant paradigm in grassland bird management that relies on previous data showing nest predation and parasitism were higher near wooded edges (Johnson and Temple 1990, Winter et al. 2000b). Differences in predator communities and definitions of woody cover may partly explain the paradox between post-fledging versus nesting requirements. Our assertion that snakes are the dominant predator is consistent with video evidence from old fields in Missouri by Thompson et al. (1999) and Thompson and Burhans (2003) and hypotheses offered by Zimmerman (1984). However, the predator community shifts towards small mammals in northern prairies (Pietz and Granfors 2000), indicating a need for additional research. Mixed and unclear definitions of woody cover also may be contributing to the apparent paradox. For example, Johnson and Temple (1990) used a definition that included a mixture of sites with mature forest and agricultural woody features (e.g. fence rows) but Winter et al. (2000b) divided edges into four types including shrubby and forest edges. We believe that future work should focus on describing woody features more precisely in relation to natural occurrence (wooded riparian corridor versus woody fence row) and identifying which of those features might pose threats to grassland birds. We address this concern in Chapter 2.

The second consistent theme in resource selection patterns across two of the three years for meadowlarks is the height of emergent vegetation (tall) which is consistent with other research. Juvenile meadowlarks in Illinois also showed a preference for grassy habitats and crops that were associated with tall and dense vegetation (Kershner 2001). Herkert (1994) also showed that one of the best predictors of adult meadowlark presence was the height of live grass. Grass was usually the tallest plant type in quadrats on our site and the dominant nest substrate for meadowlarks there (K. Suedkamp Wells) and at other locations (Roseberry and Klimstra 1970, Lanyon 1995). Tall vegetation height is likely important for predator avoidance because juvenile meadowlarks are fairly immobile during the first week out of the nest and often bury themselves in thick vegetation for long periods of time and are difficult to locate. In contrast, juvenile dickcissels become mobile earlier and elect to perch in visible locations (K. Suedkamp Wells, unpub. data).

CONSERVATION IMPLICATIONS

Our results demonstrate that nesting and post-fledging requirements are slightly different. In contrast to recommendations issued for maximizing reproductive success (Johnson and Temple 1990, Winter et al. 2000a), woody species play an important role for both of these species during the post-fledging period in different respects. Woody cover likely provided shelter from predators and hot temperatures for juvenile dickcissels and adequate perch sites for territorial defense and nest attendance and feeding for meadowlarks. To satisfy requirements for both the nesting and post-fledging periods, we suggest that researchers and managers need to re-evaluate the role of woody species in grasslands and their subsequent approach to managing those woody species.

The main challenge with the current approach is that woody features, which are often either poorly defined or not defined at all, are generally labeled "hostile" by groups with substantial management influence such as the Midwest Working Group of Partners in Flight (Pashley and Fitzgerald 1996). Although this label may reflect reality under some scenarios, we believe this one size fits all approach is not likely to satisfy post-fledging requirements and is also not consistent with recommended strategies for restoring heterogeneity in grasslands. For example, patch burning and bison (*Bison bison*) grazing techniques at the Tallgrass Prairie Preserve in Oklahoma have been successfully used to increase biodiversity in grassland systems (Fuhlendorf and Engle 2001). As a result, heterogeneity of vegetation structure across multiple scales was maximized compared to other management practices. At Konza Prairie in Kansas, a similar experiment showed that fire frequency is also important for controlling woody species in the presence of bison grazing (Briggs et al. 2002). Regardless of the type of grazing, we suggest that reintroducing combinations of disturbance regimes (burning and grazing) are most likely to mimic natural conditions, thereby, satisfying nesting and post-fledging requirements in addition to controlling woody encroachment.

LITERATURE CITED

Allison, P. D. 1999. Logistic regression using the SAS system. SAS Institute, Cary, North Carolina.

Anders, A. D. 1996. Post-fledging survival, dispersal, and habitat selection of juvenile wood thrushes. M.A. thesis. University of Missouri, Columbia.

_____, D. C. Dearborn, J. Faaborg, and F. R. Thompson III. 1997. Juvenile survival in a population of Neotropical Migrant birds. Conservation Biology 11:698-707.

Askins, R. A. 1993. Population trends in grassland, shrubland, and forest birds in eastern North America. Current Ornithology 11:1-34.

Baker, R. R. 1993. The function of post-fledging exploration: a pilot study of three species of passerines ringed in Britain. Ornis Scandinavica 24:71-79.

Briggs, J. M., A. K. Knapp, and B. L. Brock. 2002. Expansion of woody plants in tallgrass prairie: a fifteen-year study of fire and fire-grazing interactions. American Midland Naturalist 147:287-294.

Burnham, K. P. and D. R. Anderson. 1998. Model selection and inference. A practical Information-Theoretic Approach. Springer, New York, New York, USA.

_____, and _____. 2002. Model selection and multi-model inference. A practical Information-Theoretic Approach. Second edition. Springer, Norwell, Massachusetts, USA.

Daubenmire, R. 1959. A canopy-coverage method of vegetation analysis. Northwest Science 33:43-63.

Ehrlich, P. R., D. S. Dobkin, and D. Wheye. 1988. The birder's handbook. A field guide to the natural history of North American birds. Simon & Schuster, New York, New York, USA.

Faaborg, J., A. D. Anders, M. Baltz, and W. K. Gram. 1995. Non-breeding season considerations for the conservation of migratory birds in the Midwest: post-breeding and wintering periods. *In* Management of Midwestern landscapes for the conservation of Neotropical migratory birds. F. R. Thompson, editor. U.S. Forest Service General Technical Report NC-187, Detroit, Michigan, USA.

Fink, M. L. 2003. Post-fledging ecology of juvenile Wood thrush in fragmented and contiguous landscapes. Ph.D. Dissertation, University of Missouri, Columbia, Missouri.

Franklin, A. B., D. A. Anderson, R. J. Gutierrez, and K. P. Burnham. 2000. Climate, habitat quality, and fitness in Northern spotted owl populations in northwestern California. Ecological Monographs 70:539-590.

Fuhlendorf, S. D., and D. M. Engle. 2001. Restoring heterogeneity on rangelands: ecosystem management based on evolutionary grazing patterns. BioScience 51:625-632.

Gardali, T., D. C. Barton, J. D. White, and G. R. Geupel. 2003. Juvenile and adult survival of Swainson's thrush (*Catharus ustulatus*) in coastal California: annual estimates using capture-recapture analysis. Auk 120:1188-1194.

Garton, E. O., M. J. Wisdom, F. A. Leban, and B. K. Johnson. 2001. Experimental design for radiotelemetry studies. Pages 16 – 44 in J. J. Millspaugh and J. M. Marzluff, editors. Radio tracking and animal populations. Academic Press, Inc. San Diego, California, USA.

Harmeson, J. P. 1974. Breeding ecology of the dickcissel. Auk 91:348-359.

Herkert, J. R. 1994. The effects of habitat fragmentation on midwestern grassland bird communities. Ecological Applications 4:461-471.

Hosmer, D. W. and S. Lemeshow. 2000. Applied logistic regression. Second edition. Wiley & Sons, New York, New York, USA.

Johnson, H. G., and S. A. Temple. 1990. Nest predation and brood parasitism of tallgrass prairie birds. Journal of Wildlife Management 54:106-111.

Kershner, E. L. 2001. Conservation of grassland birds in an agricultural landscape: the importance of habitat availability and demography. Ph.D. Dissertation, University of Illinois at Urbana-Champaign., Urbana, Illinois.

King, R. A. and J. R. Belthoff. 2001. Post-fledging dispersal of burrowing owls in southwestern Idaho: characterization of movements and use of satellite burrows. Condor 103:118-126.

Knopf, F. L. 1994. Avian assemblages on altered grasslands. Studies in Avian Biology 15:247-257.

Lang, J. D., L. A. Powell, D. G. Krementz, and M. J. Conroy. 2002. Wood thrush movements and habitat use: effects of forest management for red-cockaded woodpeckers. Auk 119:109-124.

Lanyon, W. E. 1995. Eastern meadowlark (*Sturnella magna*). *In* The birds of North America, No. 160. A. Poole, and F. Gill, editors. The Academy of Natural Sciences and the American Ornithologists' Union, Washington, D. C., USA.

Littell, R. C., G. A. Milliken, W. W. Stroup, and R. D. Wolfinger. 1996. SAS system for mixed models. SAS Institute, Cary, North Carolina, USA.

_____, P. R. Henry, and C. B. Ammerman. 1998. Statistical analysis of repeated measures data using SAS procedures. Journal of Animal Science 76:1216-1231.

Lusk, J. J., K. M. Suedkamp Wells, F. S. Guthery, and S. D. Fuhlendorf. 2003. Lark sparrows (*Chondestes grammacus*) nest-site selection and success in a mixed-grass prairie. Auk 120:120-129.

Mech, L. D. 1983. Handbook of animal radio-tracking. University of Minnesota Press, Minneapolis, Minnesota, USA.

Moore, F. R., S. A. Gauthreaux, Jr., P. Kerlinger, and T. R. Simmons. 1993. Stopover habitat: management implications and guidelines. Pages 58-69 *in* Status and management of Neotropical Migratory birds (D. M. Finch and P. W. Stangel, Eds.). US. Forest Service General Technical Report RM-229. Fort Collins, Colorado, USA.

Morton, M. L., M. W. Wakamatsu, M.E. Pereyra, and G. A Morton. 1991. Postfledging dispersal, habitat imprinting, and philopatry in a montane, migratory sparrow. Ornis Scandinavica 22:98-106.

Nelson, K. J., and K. Martin. 1999. Thermal aspects of nest-site location for Vesper Sparrows and Horned Larks in British Columbia. Studies in Avian Biology 19:137–143.

Part, T. 1990. Natal dispersal in the collared flycatcher: possible causes and reproductive consequences. Ornis Scandinavica 21:83-88.

Pashley, D. and J. Fitzgerald. 1996. Results of Prairie Pothole Joint Venture/Partners in Flight meeting, July 11 – 12. Unpublished memorandum.

Pietz, P. J., and D. A. Granfors. 2000. Identifying predators and fates of grassland Passerine nests using miniature video cameras. Journal of Wildlife Management 64:71-87.

Powell, L. A., J. D. Lang, M. J. Conroy, and D. G. Krementz. 2000. Effects of forest management on density, survival, and population growth of Wood thrushes. Journal of Wildlife Management 64:11-23.

Rappole, J. H., and A. R. Tipton. 1991. New harness design for attachment of radio transmitters to small passerines. Journal of Field Ornithology 62:335-337.

Rich, T. D., C. J. Beardmore, H. Berlanga, P. J. Blancher, M. S. W. Bradstreet, G. S. Butcher, D. W. Demarest, E. H. Dunn, W. C. Hunter, E. E. Inigo-Elias, J. A. Kennedy, A. M. Martell, A. O. Panjabi, D. N. Pashley, K. V. Rosenberg, C. W. Rustay, J. S. Wendt, and T. C. Will. 2004. Partners in Flight North American Landbird Conservation Plan. Cornell Lab of Ornithology, Ithaca, New York, USA.

Roseberry, J. L. and W. D. Klimstra. 1970. The nesting ecology and reproductive performance of the eastern meadowlark. Wilson Bulletin 82:243-267.

SAS. 2001. SAS, version 8.0. SAS Institute, Cary, North Carolina, USA.

Sauer, J. R., J. E. Hines, and J. Fallon. 2004. The North American Breeding Bird Survey, Results and Analysis 1966 - 2003. Version 2004.1. USGS Patuxent Wildlife Research Center, Laurel, Maryland, USA.

SPSS. 1999. SYSTAT, version 9.0. SPSS, Chicago, Illinois, USA.

Steel, R. G. D., J. H. Torrie, and D. A. Dickey. 1997. Principles and procedures of statistics. A biometrical approach. Third edition. McGraw Hill, Boston, Massachusetts, USA.

Suedkamp, K. M. 2000. Effects of temperature on nest-site selection of ground-nesting grassland birds. M.S. Thesis, Oklahoma State University, Stillwater, Oklahoma, USA.

Suedkamp Wells, K. M., B. E. Washburn, J. J. Millspaugh, M. R. Ryan, and M. W. Hubbard. 2003. Effects of radio-transmitters on fecal glucocorticoid levels in captive dickcissels. Condor 105:805-810.

Sullivan, K. A. 1989. Predation and starvation: age-specific mortality in juvenile juncos (*Juncophasenotus*). Journal of Animal Ecology 58:275-286.

Thompson, F. R. III, W. Dijak, and D. E. Burhans. 1999. Video identification of predators at songbird nests in old fields. Auk 116:259-264.

_____, and D. E. Burhans. 2003. Predation of songbird nests differs by predator and between field and forest habitats. Auk 67:408-416.

Vega Rivera, J. H., J.H. Rappole, W. J. McShea, and C. A. Haas. 1998. Wood thrush postfledging movements and habitat use in northern Virginia. Condor 100:69-78.

_____, W. J. McShea, J. H. Rappole, and _____. 1999. Postbreeding movements and habitat use of adult wood thrushes in northern Virginia. Auk 116:458-466.

Vickery, P. D. 1996. Grasshopper sparrow (*Ammodramus savannarum*). *In* The Birds of North America, No. 239. A. Poole and F. Gill, editors. The Academy of Natural Sciences and the American Ornithologists' Union, Washington, D. C., USA.

_____, P. D., P. L. Tubaro, J. Maria Cardosa da Silva, B. G. Peterjohn, J. Herkert, and R. B. Cavalcanti. 1999. Conservation of grassland birds in the Western Hemisphere. Studies In Avian Biology 19:2-26

_____, and J. R. Herkert. 2001. Recent advances in grassland bird research: where do we go from here? Auk 118:11-15.

Washburn, B. E., D. J. Tempel, J. J. Millspaugh, R. J. Gutierrez, and M. E. Seamans. 2004. Factors related to fecal estrogens and fecal testosterone in California Spotted owls. Condor 106:567-579.

Webb, D. R. 1987. Thermal tolerance of avian embryos: a review. Condor 89:874-898.

Winter, M. 1999. Nesting biology of dickcissels and Henslow's sparrows in southwestern Missouri prairie fragments. Wilson Bulletin 111:515-527.

_____, D. H. Johnson, T. M. Donovan, and W. D. Svedarsky. 2000a. Evaluation of the Bird Conservation Area Concept in the Northern Tallgrass Prairie. Annual Report: 2000. Northern Prairie Wildlife Research Center, U.S. Geological Survey, Jamestown, ND: Northern Prairie Wildlife Research Center Home Page. http://www.npwrc.usgs.gov/resource/2001/bca2000/bca2000.htm (Version 26 JAN 2001).

_____, M., D. H. Johnson, and J. Faaborg. 2000b. Evidence for edge effects on multiple levels in tallgrass prairie. Condor 102:256-266.

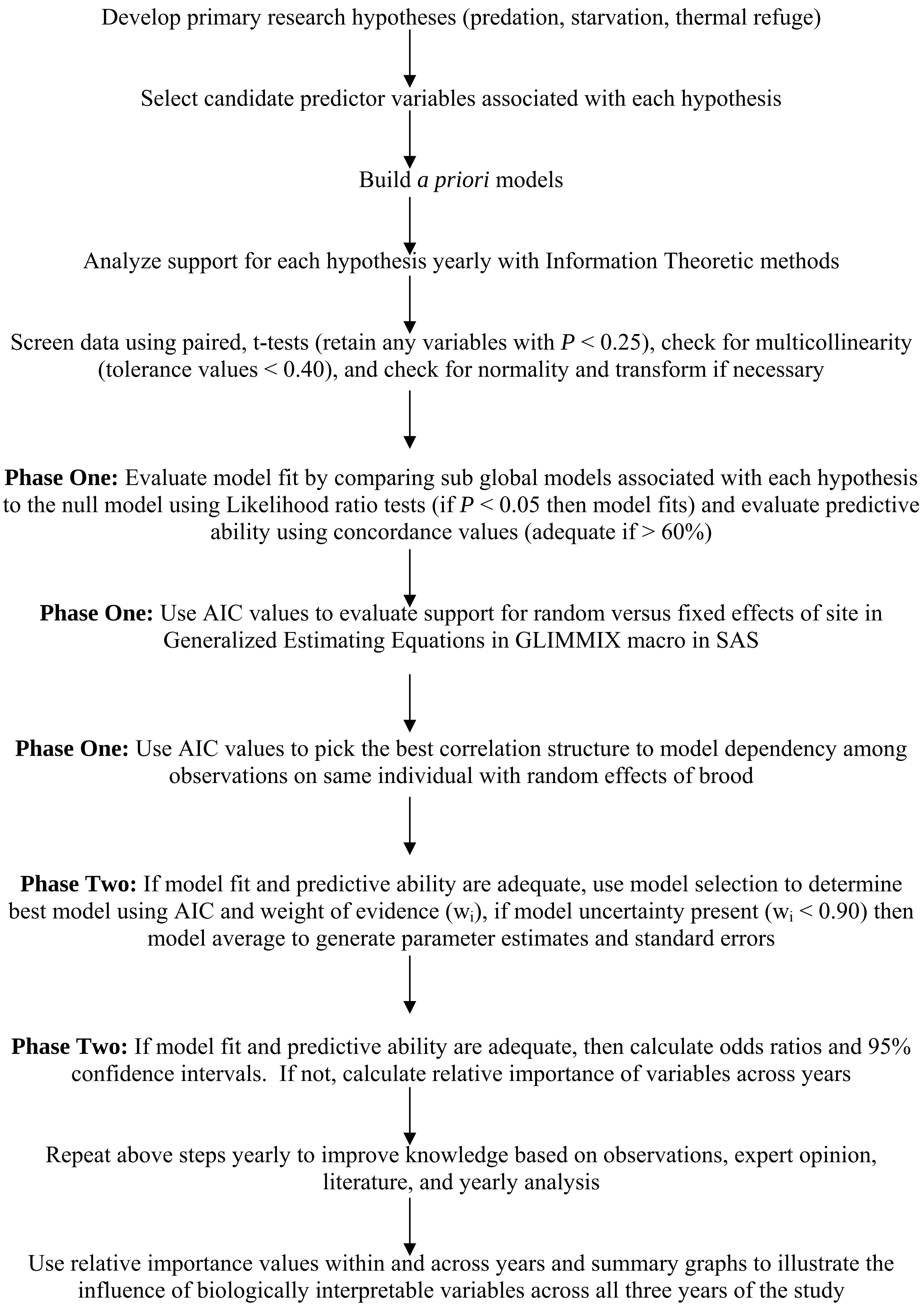
With, K. A., and D. R. Webb. 1993. Microclimate of ground nests: the relative importance of radiative cover and wind breaks for three grassland species. Condor 95:401- 413.

Yackel Adams, A. S. K. Skagen, and R. D. Adams. 2001. Movements and survival of lark bunting fledglings. Condor 103:643-647.

Zann, R. and D. Runciman. 1994. Survivorship, dispersal and sex ratios of zebra finches *Taeniopygia guttata* in southeast Australia. Ibis 136:136-146.

Zimmerman, J. L. 1984. Nest predation and its relationship to habitat and nest density in dickcissels. Condor 86:68-72.

Fig. 1. Flow chart depicting process of model development and application using Information Theoretic methods and yearly iterations from 2002 to 2004.



Develop primary research hypotheses (predation, starvation, thermal refuge)

Select candidate predictor variables associated with each hypothesis

Build *a priori* models

Analyze support for each hypothesis yearly with Information Theoretic methods

Screen data using paired, t-tests (retain any variables with $P < 0.25$), check for multicollinearity (tolerance values < 0.40), and check for normality and transform if necessary

Phase One: Evaluate model fit by comparing sub global models associated with each hypothesis to the null model using Likelihood ratio tests (if $P < 0.05$ then model fits) and evaluate predictive ability using concordance values (adequate if > 60%)

Phase One: Use AIC values to evaluate support for random versus fixed effects of site in Generalized Estimating Equations in GLIMMIX macro in SAS

Phase One: Use AIC values to pick the best correlation structure to model dependency among observations on same individual with random effects of brood

Phase Two: If model fit and predictive ability are adequate, use model selection to determine best model using AIC and weight of evidence (w_i), if model uncertainty present ($w_i < 0.90$) then model average to generate parameter estimates and standard errors

Phase Two: If model fit and predictive ability are adequate, then calculate odds ratios and 95% confidence intervals. If not, calculate relative importance of variables across years

Repeat above steps yearly to improve knowledge based on observations, expert opinion, literature, and yearly analysis

Use relative importance values within and across years and summary graphs to illustrate the influence of biologically interpretable variables across all three years of the study

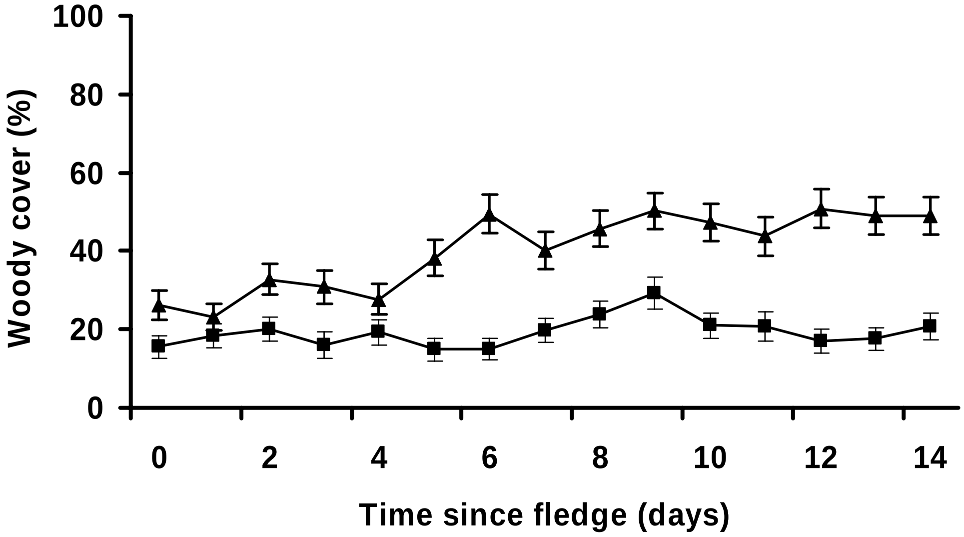


Figure 2. Mean woody cover (%) shown with one standard error at used (filled triangles) and paired random points (filled squares) for juvenile dickcissels during the first two weeks post-fledge from 2002 to 2004 in southwestern Missouri. Although only the first two-weeks post-fledging are shown, the pattern persisted throughout the rest of study period for this species (also see Fig. 5).

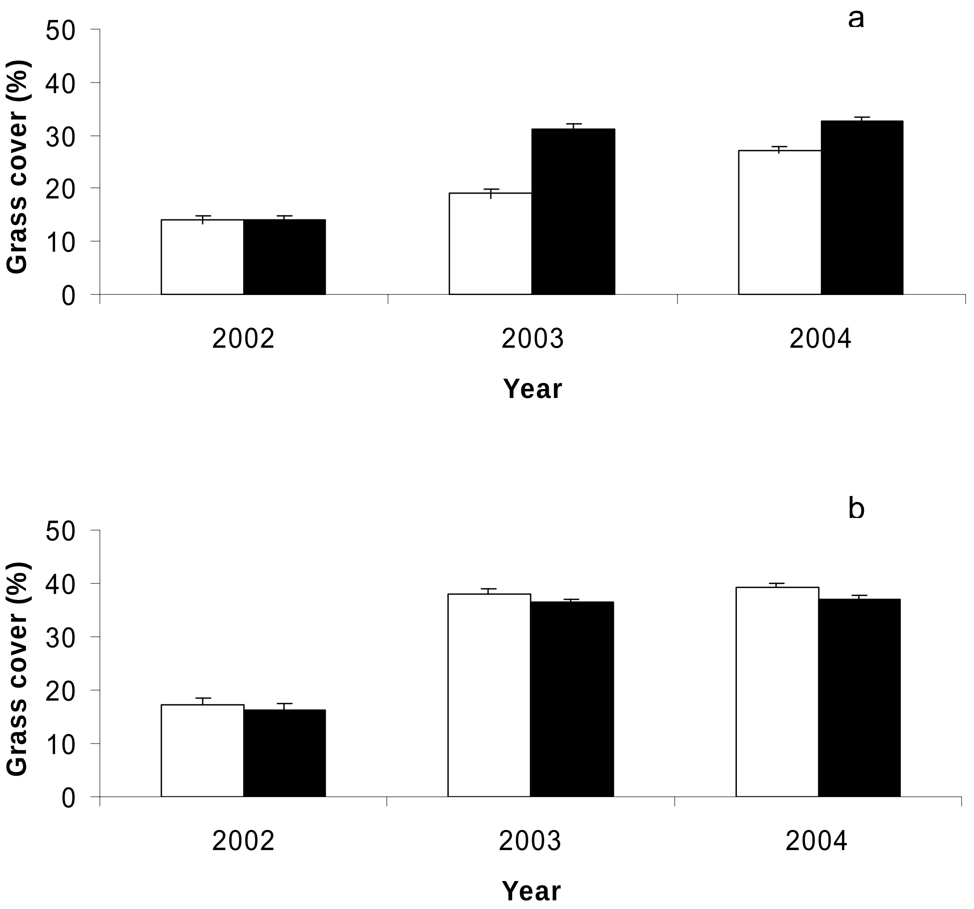


Fig. 3. Mean grass cover (%) at telemetry locations shown with one standard error at used (white bars) and available points (black bars) for a) juvenile dickcissels (*n* = 74) and b) juvenile meadowlarks (*n* = 64) in southwestern Missouri, 2002 to 2004.

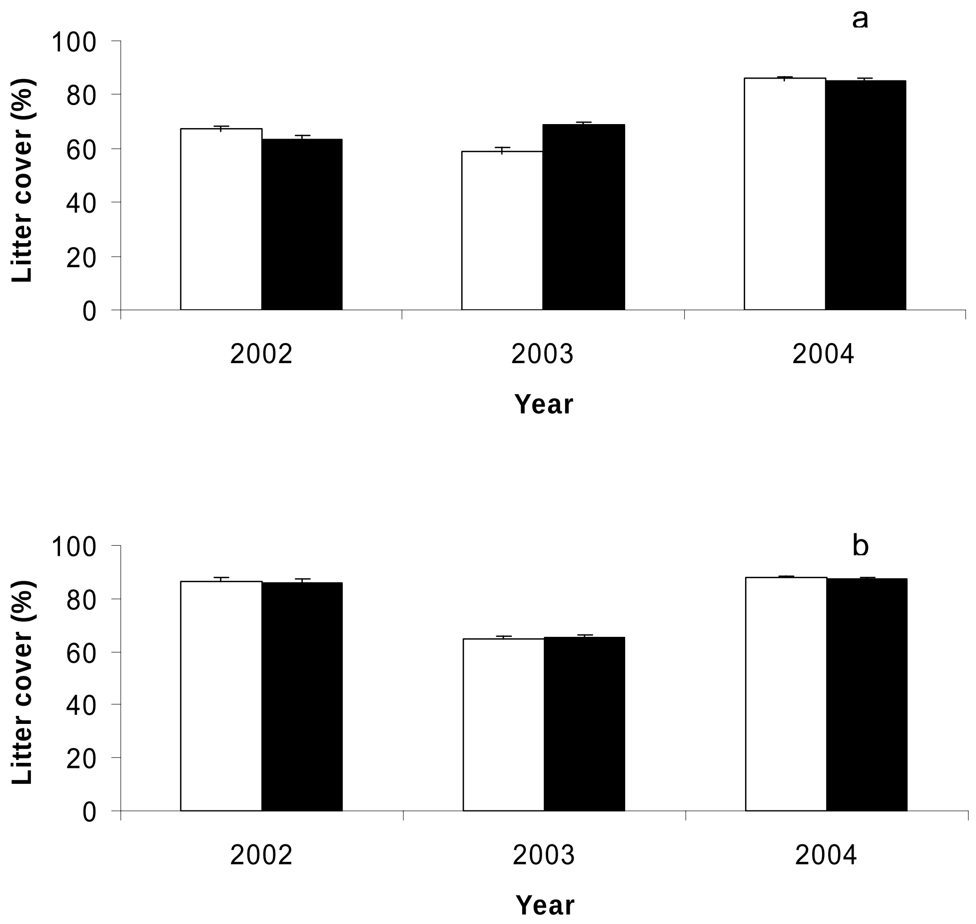


Fig. 4. Mean litter cover (%) at telemetry locations shown with one standard error at used (white bars) and available points (black bars) for a) juvenile dickcissels (*n* = 74) and b) juvenile meadowlarks (*n* = 64) in southwestern Missouri, 2002 to 2004.

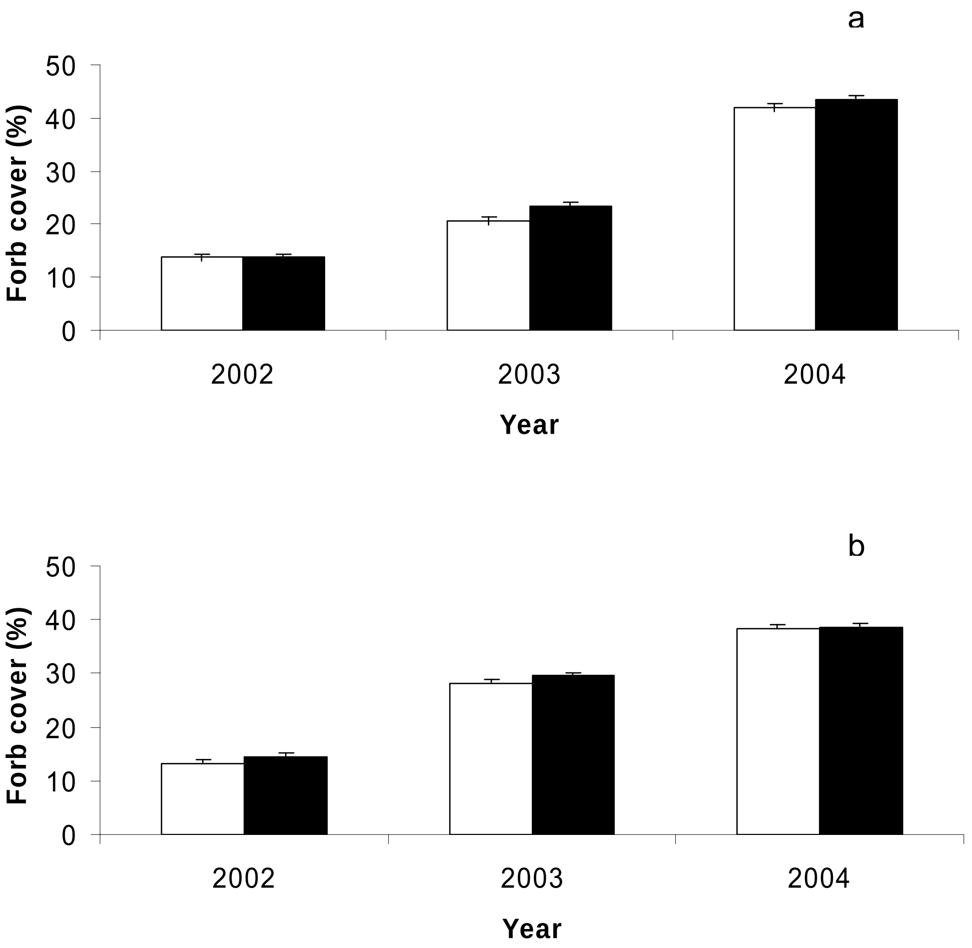


Fig. 5. Mean forb cover (%) at telemetry locations shown with one standard error at used (white bars) and available points (black bars) for a) juvenile dickcissels (*n* = 74) and b) juvenile meadowlarks (*n* = 64) in southwestern Missouri, 2002 to 2004.

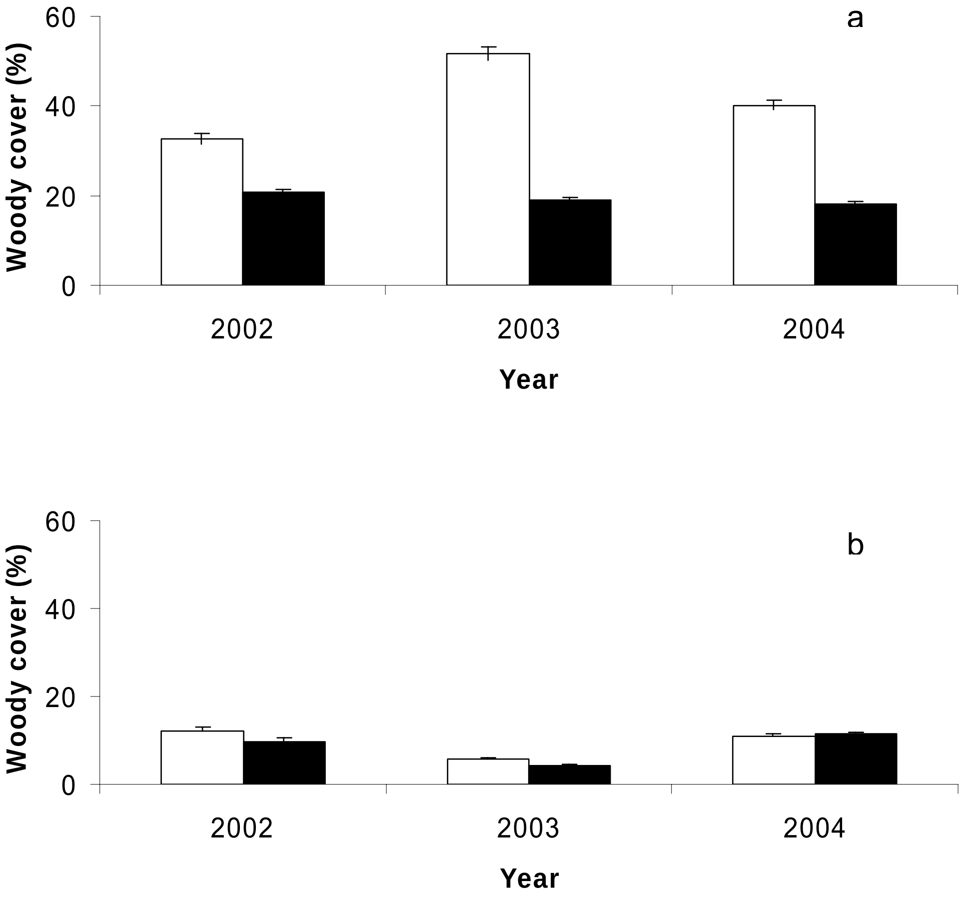
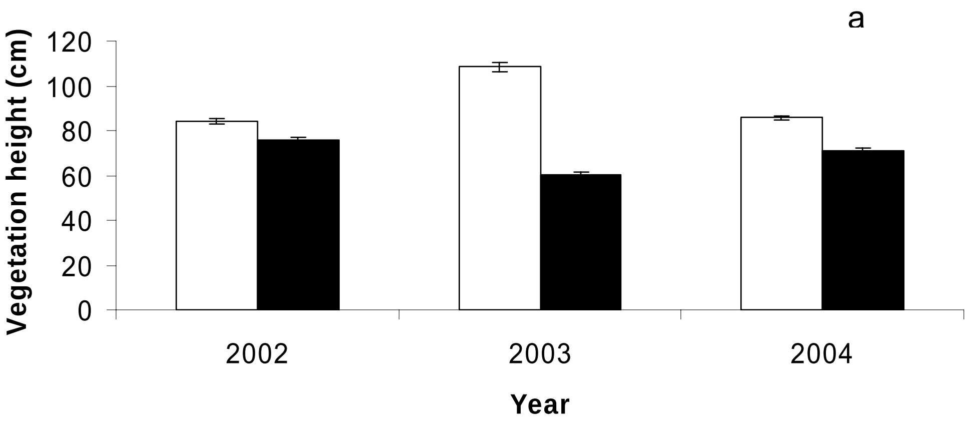


Fig. 6. Mean woody cover (%) at telemetry locations shown with one standard error at used (white bars) and available points (black bars) for a) juvenile dickcissels (*n* = 74) and b) juvenile meadowlarks (*n* = 64) in southwestern Missouri, 2002 to 2004. See Methods for a list of the small, shrub species included as woody plants.



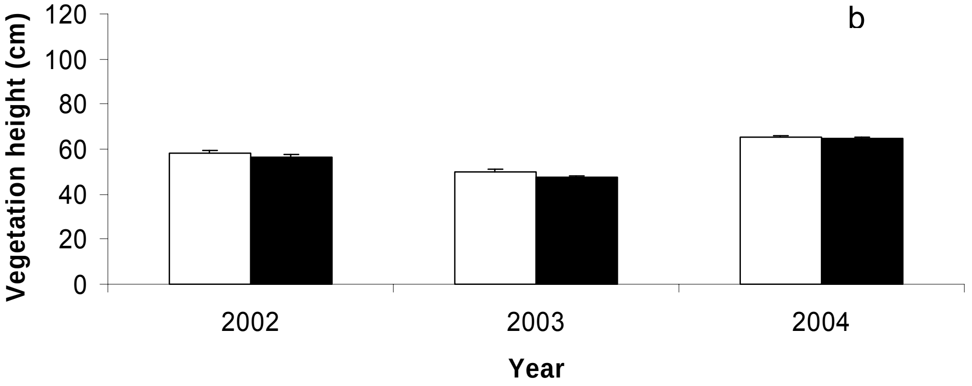


Fig. 7. Mean vegetation height (cm) at telemetry locations shown with one standard error at used (white bars) and available points (black bars) for a) juvenile dickcissels (*n* = 74) and b) juvenile meadowlarks (*n* = 64) in southwestern Missouri, 2002 to 2004.

Table 1. Nesting summary for dickcissels (DICK) and eastern meadowlarks (EAME) in Southwestern Missouri, 2002 to 2004. Variables are shown with one standard error in parentheses where appropriate.

	Species	
Variables	DICK	EAME
Total number of nests	258	113
Found building	47	9
Found laying	131	8
Found incubating	15	51
Found brooding	62	41
Successful nests	103	43
Depredated nests	121	52
Parasitized nests	23	2
Mean clutch size	4.1 (0.9)	4.4 (1.0)
Mean number of fledgings per nest	1.4 (0.1)	1.5 (2.0)
Mean number of fledglings per successful nest	3.2 (0.1)	3.7 (1.2)
Daily nest survival[a]	0.95 (0.001)	0.95 (0.001)
Mayfield nest survival (%)[b]	31	21

[a] Daily nest survival was calculated following Mayfield (1975) and standard errors following Johnson (1979).

[b] Mayfield nest survival calculated by raising the daily nest survival estimate to the power of the number of days in the nesting period (19 for dickcissels and 26 for meadowlarks).

Table 2. Summary of juvenile dickcissels (DICK) and eastern meadowlarks (EAME) used for resource selection analysis in southwestern Missouri, 2002 – 2004.

	Species	
Variable	DICK	EAME
Total number of individuals fitted with transmitters	248	164
Number of broods represented	94	46
Number of individuals with at least 30 detections		
2002	25	8
2003	19	26
2004	30	30
Total	74	64
Number of individuals with at least 50 detections		
2002	19	7
2003	13	21
2004	25	26
Total	57	54

Table 3. Final set of *a priori* candidate models to explain resource selection in dickcissels for 2002. Models are shown in decreasing rank according to their weight of evidence (w_i) with the best model shown in bold. Each candidate model is also shown with the letter code for the corresponding hypothesis for Predation (P) or Starvation (S). Concordance values are shown only for sub global models associated with each hypothesis.

Model	K^a	$Deviance^b$	AIC^c	ΔAIC^d	w_i^e	Hypothesis	Concordance
Woody*tall	**2**	**2589**	**2593**	**0**	**0.94**	**P**	
Predation sub global	5	2589	2599	6	0.05	P	0.59
Woody	2	2606	2610	17	0.00	P	
Tall	2	2616	2620	27	0.00	P	
Grass*tall	2	2645	2649	56	0.00	P	
Starvation sub global	4	2640	2645	52	0.00	S	0.47
Forb	2	2645	2649	56	0.00	S	
Litter	2	2640	2645	52	0.00	S	
Grass	2	2645	2649	56	0.00	S	

Table 3. continued.

Model	K[a]	Deviance[b]	AIC[c]	ΔAIC[d]	w_i[e]	Hypothesis	Concordance
Litter + grass	3	2640	2645	52	0.00	S	
Null[f]	1	2645	2647	54	0.00		

[a] Number of parameters in the model including one for the intercept.

[b] Deviance = - 2 (log likelihood)

[c] Akaike's Information Criteria (AIC) = Deviance + 2K (Allison 1999)

[d] Change in AIC is the difference between the model with the lowest AIC and the model of interest.

[e] Weight of evidence (w_i) is the likelihood estimate normalized to sum to 1 (Burnham and Anderson 2000).

[f] Comparing the null model to the predation sub global indicates model fit is adequate ($P < 0.01$) but comparing the null model to the starvation sub global indicates model fit is poor ($P = 0.21$).

Table 4. Parameter coefficients (β), standard errors (SE), odds ratios (OR), and 95% confidence intervals (LCL, UCL) from the best model ($w_i = 0.94$) describing the resource selection function for juvenile dickcissels (*n* = 25) in southwestern Missouri in 2002. Variable codes are from Appendix 1.

Variable	β	SE	OR	LCL, UCL
Intercept	-0.0184	0.0124	1.0002	-0.0428, 0.0058
Woody*tall	0.0002	0.0001	1.0002	0.9999, 1.0005

Table 5. Variables with relative importance > 0.10 across all candidate models in the data set for dickcissels (DICK) and eastern meadowlarks (EAME) in southwestern Missouri, 2002 – 2004.

Species	Year	Variable Code[a]	Relative Importance[b]
DICK	2002	Woody	0.99
	2003	Woody	1.00
	2004	Woody	1.00
EAME	2002	Tall	0.35
		Grass	0.34
		Woody	0.17
		Litter	0.13
	2003	Woody	0.42
		Cindex	0.17
		Tall	0.12
	2004	Woody	1.00

[a] See Appendix 1 for a description of each variable code.

[b] Relative importance was calculated by summing the weights of evidence (w_i) across all models in the data set for each variable following Burnham and Anderson (2002).

Table 6. Final set of *a priori* candidate models in relation to three hypotheses to explain resource selection in dickcissels for 2003. Models are shown in decreasing rank according to their weight of evidence (w_i) with the best models shown in bold. Each candidate model is also shown with the letter code for the corresponding hypothesis for Predation (P), Starvation (S), or Thermal Refuge (T). Concordance values are shown only for sub global models associated with each hypothesis.

Model	K^a	Deviance^b	AIC^c	ΔAIC^d	w_i^e	Hypothesis	Concordance
Predation sub global	**11**	**2176**	**2198**	**0**	**0.88**	**P**	**0.75**
Thermal Refuge sub global	**9**	**2184**	**2204**	**6**	**0.12**	**T**	**0.74**
Woody	2	2320	2324	126	0.00	P	
Tall	2	2329	2333	135	0.00	P	
Cindex	2	2524	2428	330	0.00	P	
Disper	2	2597	2601	403	0.00	P	
Diswdp	2	2596	2600	402	0.00	P	
Woody*cindex	2	2300	2304	106	0.00	P	
Woody*diswdp	2	2602	2606	408	0.00	P	
Woody + disper	3	2320	2326	128	0.00	P	

Table 6. continued

Model	K[a]	Deviance[b]	AIC[c]	ΔAIC[d]	w_i[e]	Hypothesis	Concordance
Cindex*tsf	2	2588	2592	394	0.00	P	
Starvation sub global	8	2482	2498	300	0.00	S	0.64
Forb	2	2598	2602	404	0.00	S	
Litter + grass	3	2502	2508	310	0.00	S	
Jdate*forb	2	2598	2602	404	0.00	S	
Jdate*grass	2	2513	2517	319	0.00	S	
Jdate*tsf	2	2603	2607	409	0.00	S	
Jdate*time	2	2603	2607	409	0.00	S	
Woody*temp	2	2333	2337	139	0.00	T	
Woody*jdate	2	2313	2317	119	0.00	T	
Jdate*temp	2	2603	2607	409	0.00	T	
Tall*temp	2	2477	2481	283	0.00	T	
Time*cindex	2	2560	2564	366	0.00	T	

Table 6. continued

Model	K[a]	Deviance[b]	AIC[c]	ΔAIC[d]	w_i[e]	Hypothesis	Concordance
Tsf*temp	2	2603	2607	409	0.00	T	
Null[f]	1	2603	2605	407	0.00	T	

[a] Number of parameters in the model including one for the intercept.

[b] Deviance = - 2 (log likelihood)

[c] Akaike's Information Criteria (AIC) = Deviance + 2K (Allison 1999)

[d] Change in AIC is the difference between the model with the lowest AIC and the model of interest.

[e] Weight of evidence (w_i) is the likelihood estimate normalized to sum to 1 (Burnham and Anderson 2000).

[f] Comparing the null model to the predation sub global ($P < 0.01$), starvation sub global ($P < 0.01$), and thermal refuge sub global ($P < 0.01$) indicates model fit is adequate.

Table 7. Parameter coefficients (β), standard errors (SE), odds ratios (OR), and 95% confidence intervals (LCL, UCL) from model averaging ($w_i < 0.90$) describing the resource selection function for juvenile dickcissels (*n* = 19) in southwestern Missouri in 2003. Variable codes are from Appendix 1.

Variable	β	SE	OR	LCL, UCL
Intercept	-4.5431	0.6449	0.0106	0.0030, 1.0030
Tall	1.9387	0.5179	6.9499	2.5183, 12.4084
Woody	0.2824	0.6788	1.3263	0.3505, 1.4198
Cindex	0.3280	0.5533	1.3881	0.4693, 1.5988
Disperch	-0.0023	0.1552	0.9976	0.7358, 2.0873
Diswdp	0.0095	0.1586	1.0096	0.7398, 2.0955
Woody*cindex	0.4069	0.4931	1.5022	0.5713, 1.7707
Woody*diswdp	-0.0059	0.1448	0.9940	0.7483, 2.1135
Woody*disperch	0.0006	0.1406	1.0006	0.7594, 2.1371
Cindex*tsf	-0.0100	0.1927	0.9900	0.6784, 1.9708
Woody*temp	0.0038	0.0316	1.0038	0.9434, 2.5688
Woody*jdate	0.0010	0.0255	1.0010	0.9577, 2.6057
Jdate*temp	-0.0000	0.0106	0.9999	0.9793, 2.6627
Tall*temp	-0.0001	0.0349	0.9999	0.9337, 2.5439
Time*cindex	-0.0000	0.0106	0.9999	0.9792, 2.6624
Tsf*temp	-0.0002	0.0113	0.9999	0.9780, 2.6591

Table 8. Final set of *a priori* candidate models in relation to three hypotheses to explain resource selection in dickcissels for 2003. Models are shown in decreasing rank according to their weight of evidence (w_i) with the best model shown in bold. Each candidate model is also shown with the letter code for the corresponding hypothesis for Predation (P), Starvation (S), or Thermal Refuge (T). Concordance values are shown only for sub global models associated with each hypothesis.

Model	K[a]	Deviance[b]	AIC[c]	ΔAIC[d]	w_i[e]	Hypothesis	Concordance
Predation sub global	**12**	**3030**	**3054**	**0**	**1.00**	**P**	**0.77**
Woody	2	3404	3408	354	0.00	P	
Woody*woody	2	3369	3373	319	0.00	P	
Woody*jdate	2	3421	3425	371	0.00	P	
Diswdp*jdate	2	3646	3650	596	0.00	P	
Tall*jdate	2	3648	3652	598	0.00	P	
Cindex*jdate	2	3612	3616	562	0.00	P	
Disper	2	3621	3625	571	0.00	P	
Disper*disper	2	3635	3639	585	0.00	P	
Diswdp	2	3649	3653	599	0.00	P	

Table 8. continued

Model	K[a]	Deviance[b]	AIC[c]	ΔAIC[d]	w_i[e]	Hypothesis	Concordance
Diswdp*diswdp	2	3611	3615	561	0.00	P	
Woody*cindex	2	3442	3446	392	0.00	P	
Woody*jdate + diswdp*jdate	3	3421	3427	373	0.00	P	
Starvation sub global	11	3279	3301	247	0.00	S	0.71
Forb + grass	3	3613	3619	565	0.00	S	
Forb*forb	2	3643	3647	593	0.00	S	
Grass*grass	2	3641	3645	591	0.00	S	
Litter +grass +forb	4	3594	3602	548	0.00	S	
Jdate*forb +jdate*grass + Jdate*litter	4	3578	3586	532	0.00	S	
Jdate*time*tsf	2	3457	3461	407	0.00	S	
Jdate*sibs	2	3340	3344	290	0.00	S	
Tsf*sibs	2	3345	3349	295	0.00	S	

Table 8. continued

Model	K[a]	Deviance[b]	AIC[c]	ΔAIC[d]	w_i[e]	Hypothesis	Concordance
Thermal Refuge sub global	8	3242	3258	204	0.00	T	0.73
Woody*temp	2	3408	3412	358	0.00	T	
Woody*jdate	2	3421	3425	371	0.00	T	
Jdate*temp	2	3625	3629	575	0.00	T	
Jdate*time*woody	2	3649	3653	599	0.00	T	
Jdate*time*cindex	2	3632	3636	582	0.00	T	
Woody*diswdp*temp	2	3649	3653	599	0.00	T	
Tall*temp	2	3616	3620	566	0.00	T	
Woody*cindex*jdate	2	3455	3459	405	0.00	T	
Null[f]	1	3651	3653	599	0.00		

[a] Number of parameters in the model including one for the intercept.

[b] Deviance = - 2 (log likelihood)

[c] Akaike's Information Criteria (AIC) = Deviance + 2K (Allison 1999)

[d] Change in AIC is the difference between the model with the lowest AIC and the model of interest.

[e] Weight of evidence (w_i) is the likelihood estimate normalized to sum to 1 (Burnham and Anderson 2000).

[f] Comparing the null model to the predation sub global ($P < 0.01$), starvation sub global ($P < 0.01$), and thermal refuge sub global ($P < 0.01$) indicates model fit is adequate.

Table 9. Parameter coefficients (β), standard errors (SE), odds ratios (OR), and 95% confidence intervals (LCL, UCL) from the best model ($w_i = 1.0$) describing the resource selection function for juvenile dickcissels (*n* = 30) in southwestern Missouri in 2004. Variable codes are from Appendix 1.

Variable	β	SE	OR	LCL, UCL
Intercept	-2.0613	0.3647	0.1272	0.0622, 0.2601
Tall	8.6490	0.8037	5704.4390	1180.5650, 27563.6200
Woody	-3.6733	0.9635	0.0253	0.0038, 0.1678
Disperch	0.0059	0.0023	1.0059	1.0014, 1.0105
Diswdp	-0.0096	0.0100	0.9904	0.9711, 1.0100
Tall*jdate	-0.0418	0.0041	0.9589	0.9511, 0.9668
Woody^2	1.2381	0.1409	3.4490	2.616, 4.5460
Woody*jdate	0.0143	0.0050	1.0144	1.0044, 1.0245
Diswdp*jdate	0.0000	0.0001	1.0000	0.9999, 1.0000
Jdate*cindex	0.0049	0.0013	1.0049	1.0023, 1.0075
Disperch^2	-0.0000	0.0000	0.9999	0.9999, 0.9999
Diswdp^2	0.0000	0.0000	1.0000	1.0000, 1.0000
Woody*cindex	-0.9627	0.2362	0.3818	0.2403, 0.6066

Table 10. Final set of *a priori* candidate models to explain resource selection in eastern meadowlarks for 2002. Models are shown in decreasing rank according to their weight of evidence (w_i). Each candidate model is also shown with the letter code for the corresponding hypothesis for Predation (P) and Starvation (S). Concordance values are shown only for sub global models associated with each hypothesis.

Model	K[a]	Deviance[b]	AIC[c]	ΔAIC[d]	w_i[e]	Hypothesis	Concordance
Null[f]	1	1004	1006	0	0.20		
Grass*tall	2	1002	1006	0	0.18	P	
Woody	2	1002	1006	1	0.15	P	
Tall	2	1002	1006	1	0.15	P	
Grass	2	1003	1007	1	0.09	S	
Litter	2	1003	1007	1	0.08	S	
Forb	2	1004	1008	2	0.07	S	
Litter +grass	3	1003	1009	3	0.03	S	
Predation sub global	4	1000	1008	2	0.02	P	0.52
Starvation sub global	4	1003	1011	5	0.01	S	0.50

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[a] Number of parameters in the model including one for the intercept.

[b] Deviance = - 2 (log likelihood) from PROC GENMOD

[c] Akaike's Information Criteria (AIC) = Deviance + 2K (Allison 1999)

[d] Change in AIC is the difference between the model with the lowest AIC and the model of interest.

[e] Weight of evidence (w_i) is the likelihood estimate normalized to sum to 1 (Burnham and Anderson 2000).

[f] Comparing the null model to the predation sub global ($P = 0.48$) and to the starvation sub global ($P = 0.89$) indicates a lack of fit.

Table 11. Final set of *a priori* candidate models in relation to three hypotheses to explain resource selection in meadowlarks for 2003. Models are shown in decreasing rank according to their weight of evidence (w_i). Each candidate model is also shown with the letter code for the corresponding hypothesis for Predation (P), Starvation (S), or Thermal Refuge (T). Concordance values are shown only for sub global models associated with each hypothesis.

Model	K^a	$Deviance^b$	AIC^c	ΔAIC^d	w_i^e	Hypothesis	Concordance
Woody*temp	2	4134	4138	0	0.14	T	
Woody*jdate	2	4135	4139	1	0.09	T	
Woody	2	4136	4140	2	0.05	P	
Cindex	2	4136	4140	2	0.05	P	
$Null^f$	1	4138	4140	2	0.05		
Cindex*tsf	2	4136	4140	2	0.05	P	
Forb	2	4136	4140	2	0.05	S	
Jdate*grass	2	4136	4140	2	0.05	S	
Thermal Refuge sub global	9	4122	4140	2	0.05	T	0.52
Woody*diswdp	2	4137	4141	3	0.04	P	

Table 11. continued

Model	K^a	$Deviance^b$	AIC^c	ΔAIC^d	w_i^e	Hypothesis	Concordance
Litter + grass	3	4135	4141	3	0.04	S	
Jdate*forb	2	4137	4141	3	0.04	S	
Tall*temp	2	4137	4141	3	0.03	T	
Time*cindex	2	4137	4141	3	0.03	T	
Tall	2	4138	4142	4	0.02	P	
Disper	2	4138	4142	4	0.02	P	
Diswdp	2	4138	4142	4	0.02	P	
Woody*cindex	2	4138	4142	4	0.02	P	
Woody + disper	3	4136	4142	4	0.02	P	
Jdate*tsf	2	4138	4142	4	0.02	S	
Jdate*time	2	4138	4142	4	0.02	S	
Jdate*temp	2	4138	4142	4	0.02	T	
Tsf*temp	2	4138	4142	4	0.02	T	

Table 11. continued

Model	K^a	Deviance^b	AIC^c	ΔAIC^d	w_i^e	Hypothesis	Concordance
Starvation sub global	8	4130	4146	8	0.00	S	0.52
Predation sub global	11	4128	4150	12	0.00	P	0.51

[a] Number of parameters in the model including one for the intercept.

[b] Deviance = - 2 (log likelihood)

[c] Akaike's Information Criteria (AIC) = Deviance + 2K (Allison 1999)

[d] Change in AIC is the difference between the model with the lowest AIC and the model of interest.

[e] Weight of evidence (w_i) is the likelihood estimate normalized to sum to 1 (Burnham and Anderson 2000).

[f] Comparing the null model to the predation sub global ($P > 0.20$), starvation sub global ($P > 0.95$), and thermal refuge sub global ($P > 0.20$) indicates model fit is poor.

Table 12. Final set of *a priori* candidate models in relation to three hypotheses to explain resource selection in eastern meadowlarks for 2004. Models are shown in decreasing rank according to their weight of evidence (w_i) with the best model shown in bold. Each candidate model is also shown with the letter code for the corresponding hypothesis for Predation (P), Starvation (S), or Thermal Refuge (T). Concordance values are shown only for sub global models associated with each hypothesis.

Model	K[a]	Deviance[b]	AIC[c]	ΔAIC[d]	w_i[e]	Hypothesis	Concordance
Predation sub global	**12**	**3104**	**3128**	**0**	**1.00**	**P**	**0.65**
Woody	2	3465	3469	341	0.00	P	
Woody*woody	2	3466	3470	342	0.00	P	
Tall	2	3466	3470	342	0.00	P	
Woody*jdate	2	3469	3473	345	0.00	P	
Diswdp*jdate	2	3441	3445	317	0.00	P	
Tall*jdate	2	3432	3436	308	0.00	P	
Cindex*jdate	2	3470	3474	346	0.00	P	
Disper	2	3466	3470	342	0.00	P	
Disper*disper	2	3468	3472	344	0.00	P	

Table 12. continued

Model	K[a]	Deviance[b]	AIC[c]	ΔAIC[d]	w_i[e]	Hypothesis	Concordance
Diswdp	2	3447	3451	323	0.00	P	
Diswdp*diswdp	2	3453	3457	329	0.00	P	
Woody*cindex	2	3467	3471	343	0.00	P	
Woody*jdate + diswdp*jdate	3	3440	3446	318	0.00	P	
Starvation sub global	11	3140	3162	34	0.00	S	0.70
Forb + grass	3	3460	3466	338	0.00	S	
Forb*forb	2	3469	3473	345	0.00	S	
Litter*litter	2	3451	3455	327	0.00	S	
Grass*grass	2	3459	3463	335	0.00	S	
Litter +grass +forb	4	3444	3452	324	0.00	S	
Jdate*forb +jdate*grass + Jdate*litter	4	3462	3470	342	0.00	S	
Jdate*time*tsf	2	3275	3279	151	0.00	S	

Table 12. continued

Model	K^a	$Deviance^b$	AIC^c	ΔAIC^d	w_i^e	Hypothesis	Concordance
Jdate*sibs	2	3159	3163	35	0.00	S	
Tsf*sibs	2	3166	3170	42	0.00	S	
Thermal Refuge sub global	8	3334	3350	222	0.00	T	0.62
Woody*temp	2	3464	3468	340	0.00	T	
Woody*jdate	2	3469	3473	345	0.00	T	
Jdate*temp	2	3453	3457	329	0.00	T	
Jdate*time*woody	2	3469	3473	345	0.00	T	
Jdate*time*cindex	2	3469	3473	345	0.00	T	
Woody*diswdp*temp	2	3470	3474	346	0.00	T	
Tall*temp	2	3467	3471	343	0.00	T	
Woody*cindex*jdate	2	3470	3474	346	0.00	T	
$Null^f$	1	3470	3472	344	0.00		

[a] Number of parameters in the model including one for the intercept.

[b] Deviance = - 2 (log likelihood)

[c] Akaike's Information Criteria (AIC) = Deviance + 2K (Allison 1999)

[d] Change in AIC is the difference between the model with the lowest AIC and the model of interest.

[e] Weight of evidence (w_i) is the likelihood estimate normalized to sum to 1 (Burnham and Anderson 2000).

[f] Comparing the null model to the predation sub global ($P < 0.01$), starvation sub global ($P < 0.01$), and thermal refuge sub global ($P < 0.01$) indicates model fit is adequate.

Table 13. Parameter coefficients (β), standard errors (SE), odds ratios (OR), and 95% confidence intervals (LCL, UCL) from the best model ($w_i = 1.0$) describing the resource selection function for juvenile meadowlarks (*n* = 26) in southwestern Missouri in 2004. Variable codes are from Appendix 1.

Variable	β	SE	OR	LCL, UCL
Intercept	0.1828	0.4250	1.3255	0.5762, 3.0489
Tall	7.3895	0.5890	1618.8960	510.3312, 5135.5390
Woody	2.0194	0.8124	7.5338	1.5328, 37.0291
Woody^2	0.2226	0.1612	1.2493	0.9108, 1.7135
Disperch	0.0017	0.0017	1.0017	0.9982, 1.0053
Disperch^2	-0.0000	0.0000	0.9999	0.9999, 1.0000
Diswdp	-0.0326	0.0078	0.9679	0.9530, 0.9829
Diswdp^2	0.0000	0.0000	1.0000	1.0000, 1.0000
Jdate*tall	-0.0411	0.0031	0.9597	0.9537, 0.9657
Jdate*woody	-0.0112	0.0045	0.9887	0.9800, 0.9975
Jdate*diswdp	0.0001	0.0000	1.0000	1.0000, 1.0000
Jdate*cindex	0.0001	0.0013	1.0000	0.9974, 1.0027
Woody*cindex	-0.5184	0.3253	0.5954	0.3147, 1.1265

Appendix 1. Variable codes and descriptions for fixed effects used to predict resource selection for juvenile dickcissels and eastern meadowlarks in southwestern Missouri, 2002 – 2004.

Variable Code	Description
TSF	Time since fledging (days) where the first day out of the nest is day zero.
SIBS	Number of siblings that fledged in a brood
TIME	Time of the day (hours) that bird was radio tracked
JDATE	Julian date
TEMP	Temperature (°C) at the bird location or random location
GRASS	Grass cover (%) in a 20 x 50 cm Daubenmire quadrat
LITTER	Litter cover (%) in a 20 x 50 cm Daubenmire quadrat
FORB	Forb cover (%) in a 20 x 50 cm Daubenmire quadrat
WOODY	Woody cover (%) in a 20 x 50 cm Daubenmire quadrat
TALL	Height (cm) of vegetation in a 20 x 50 cm Daubenmire quadrat
CONC	Concealment class from above the bird or random location
SHADE	Shading class from above the bird of random location
CINDEX	CONC + SHADE
DISPER	Distance to a perch (m) that was at least 2 m tall and could support the body weight of a raptor
DISWDP	Distance to a woody patch (m) ≥ 3m in diameter that could

conceal a den, burrow, or predator

Appendix 2. Candidate model descriptions, structures, and expected results related to two hypotheses for predicting resource use in juvenile dickcissels in southwestern Missouri, 2002. The Thermal Refuge Hypothesis was added in 2003 and therefore not included.

Description	Structure	Expected Result
Predation Hypothesis		
Predation sub global	$\beta_o + \beta_1(\text{woody}) + \beta_2(\text{tall}) + \beta_3(\text{grass*tall}) + \beta_4(\text{woody*tall})$	$\beta_1 < 0, \beta_2 > 0, \beta_3 > 0, \beta_4 < 0$
Effect of woody cover	$\beta_o + \beta_1(\text{woody})$	$\beta_1 < 0$
Effect of tall vegetation	$\beta_o + \beta_1(\text{tall})$	$\beta_1 > 0$
Effect of grass cover and vegetation height	$\beta_o + \beta_1(\text{grass*tall})$	$\beta_1 > 0$
Effect of woody cover and vegetation height	$\beta_o + \beta_1(\text{woody*tall})$	$\beta_1 < 0$
Starvation Hypothesis		
Starvation sub global	$\beta_o + \beta_1(\text{forb}) + \beta_2(\text{litter}) + \beta_3(\text{grass})$	$\beta_1 > 0, \beta_2 > 0, \beta_3 > 0$
Effect of forb food sources	$\beta_o + \beta_1(\text{forb})$	$\beta_1 > 0$
Effect of litter food sources	$\beta_o + \beta_1(\text{litter})$	$\beta_1 > 0$

Appendix 2. continued.

Description	Structure	Expected Result
Effect of grass food sources	$\beta_o + \beta_1(\text{grass})$	$\beta_1 > 0$
Effect of litter and grass food sources	$\beta_o + \beta_1(\text{litter}) + \beta_2(\text{grass})$	$\beta_1 > 0, \beta_2 > 0$

Appendix 3. Description, structure, and expected results for *a priori* candidate models in relation to three hypotheses for explaining resource selection in dickcissels in 2003.

Description	Structure	Expected Result
Predation Hypothesis		
Predation sub global	$\beta_o + \beta_1(\text{woody}) + \beta_2(\text{tall}) + \beta_3(\text{cindex}) + \beta_4(\text{disper}) + \beta_5(\text{diswdp}) + \beta_6(\text{woody*cindex}) + \beta_7(\text{woody*diswdp}) + \beta_8(\text{woody*disper}) + \beta_8(\text{cindex*tsf})$	$\beta_1 < 0, \beta_2 > 0, \beta_3 > 0, \beta_4 > 0, \beta_5 > 0, \beta_6 > 0, \beta_7 > 0, \beta_8 > 0, \beta_9 > 0$
Effect of woody cover	$\beta_o + \beta_1(\text{woody})$	$\beta_1 < 0$
Effect of tall vegetation	$\beta_o + \beta_1(\text{tall})$	$\beta_1 > 0$
Effect of cover and shade	$\beta_o + \beta_1(\text{cindex})$	$\beta_1 > 0$
Effect of proximity to raptor perches	$\beta_o + \beta_1(\text{disper})$	$\beta_1 > 0$
Effect of proximity to woody patches	$\beta_o + \beta_1(\text{diswdp})$	$\beta_1 > 0$
Effects of all cover types	$\beta_o + \beta_1(\text{woody*cindex})$	$\beta_1 > 0$
Effects of woody cover and proximity to woody patches	$\beta_o + \beta_1(\text{woody*diswdp})$	$\beta_1 > 0$

Appendix 3. continued.

Description	Structure	Expected Result
Effect of woody cover and proximity to raptor perches	$\beta_o + \beta_1(\text{woody*disper})$	$\beta_1 > 0$
Effect of cover and age	$\beta_o + \beta_1(\text{cindex*tsf})$	$\beta_1 > 0$
Starvation Hypothesis		
Starvation sub global	$\beta_o + \beta_1(\text{forb}) + \beta_2(\text{litter}) + \beta_3(\text{grass}) + \beta_4(\text{jdate*forb}) + \beta_5(\text{jdate*grass}) + \beta_6(\text{jdate*tsf}) + \beta_7(\text{jdate*time})$	$\beta_1 > 0, \beta_2 > 0, \beta_3 > 0$ $\beta_4 > 0, \beta_5 > 0, \beta_6 < 0,$ $\beta_7 < 0$
Effect of forb food source availability	$\beta_o + \beta_1(\text{forb})$	$\beta_1 > 0$
Effect of litter and grass food Source availability	$\beta_o + \beta_1(\text{litter}) + \beta_2\,(\text{grass})$	$\beta_1 > 0, \beta_2 > 0$
Seasonal effect of forb food source availability	$\beta_o + \beta_1(\text{forb*jdate})$	$\beta_1 > 0$
Seasonal effect of grass food source availability	$\beta_o + \beta_1(\text{grass*jdate})$	$\beta_1 > 0$
Seasonal and age effects	$\beta_o + \beta_1(\text{jdate*tsf})$	$\beta_1 < 0$

Appendix 3. continued.

Description	Structure	Expected Result
Seasonal and temporal effects	$\beta_o + \beta_1(\text{jdate*time})$	$\beta_1 < 0$
Thermal Refuge Hypothesis		
Thermal Refuge sub global	$\beta_o + \beta_1(\text{woody*temp}) + \beta_2(\text{jdate*woody}) + \beta_3(\text{jdate*temp})$ $+ \beta_4(\text{tall*temp}) + \beta_5(\text{time*cindex}) + \beta_6(\text{tsf*temp})$	$\beta_1 > 0, \beta_2 > 0, \beta_3 < 0$ $\beta_4 < 0, \beta_5 > 0, \beta_6 < 0$
Effects of woody cover and temperature	$\beta_o + \beta_1(\text{woody*temp})$	$\beta_1 > 0$
Seasonal effects of woody cover	$\beta_o + \beta_1(\text{jdate*woody})$	$\beta_1 > 0$
Seasonal effects of temperature	$\beta_o + \beta_1(\text{jdate*temp})$	$\beta_1 < 0$
Effects of vegetation height and temperature	$\beta_o + \beta_1(\text{tall*temp})$	$\beta_1 < 0$
Temporal effects of cover	$\beta_o + \beta_1(\text{time*cindex})$	$\beta_1 > 0$
Effects of age and temperature	$\beta_o + \beta_1(\text{tsf*temp})$	$\beta_1 < 0$

Appendix 4. Description, structure, and expected results for *a priori* candidate models in relation to three hypotheses for explaining resource selection in dickcissels in 2004.

Description	Structure	Expected Result
Predation Hypothesis		
Predation sub global	$\beta_o + \beta_1(\text{woody}) + \beta_2(\text{woody})^2 + \beta_3(\text{disper}) + \beta_4(\text{disper})^2$ $+ \beta_5(\text{diswdp}) + \beta_6(\text{diswdp})^2 + \beta_7(\text{tall*jdate}) + \beta_8(\text{woody*jdate})$ $+ \beta_9(\text{diswdp*jdate}) + \beta_{10}(\text{cindex*jdate}) + \beta_{11}(\text{woody*cindex})$	$\beta_1 > 0$, $\beta_2 > 0$, $\beta_3 > 0$, $\beta_4 > 0$, $\beta_5 > 0$, $\beta_6 > 0$, $\beta_7 > 0$, $\beta_8 > 0$, $\beta_9 < 0$ $\beta_{10} > 0$, $\beta_{11} > 0$
Effect of woody cover	$\beta_o + \beta_1(\text{woody})$	$\beta_1 > 0$
Threshold effect of woody cover	$\beta_o + \beta_1(\text{woody})^2$	$\beta_1 > 0$
Effect of proximity to raptor perches	$\beta_o + \beta_1(\text{disper})$	$\beta_1 > 0$
Threshold effect of proximity to raptor perches	$\beta_o + \beta_1(\text{disper})^2$	$\beta_1 > 0$
Effect of proximity to woody patches	$\beta_o + \beta_1(\text{diswdp})$	$\beta_1 > 0$
Threshold effect of proximity to woody patches	$\beta_o + \beta_1(\text{diswdp})^2$	$\beta_1 > 0$

Appendix 4. continued.

Description	Structure	Expected Result
Effects of all cover types	$\beta_o + \beta_1(\text{woody*cindex})$	$\beta_1 > 0$
Effect of woody cover and proximity to raptor perches	$\beta_o + \beta_1(\text{woody*disper})$	$\beta_1 > 0$
Seasonal effect of vegetation height	$\beta_o + \beta_1(\text{jdate*tall})$	$\beta_1 > 0$
Seasonal effect of woody cover	$\beta_o + \beta_1(\text{jdate*woody})$	$\beta_1 > 0$
Seasonal effect of proximity to woody patches	$\beta_o + \beta_1(\text{jdate*diswdp})$	$\beta_1 < 0$
Seasonal effect of cover and shade	$\beta_o + \beta_1(\text{jdate*cindex})$	$\beta_1 > 0$
Seasonal effect of woody cover and proximity to woody patches	$\beta_o + \beta_1(\text{jdate*woody}) + \beta_2(\text{jdate*diswdp})$	$\beta_1 > 0, \beta_1 < 0$

Appendix 4. continued.

Description	Structure	Expected Result
Starvation Hypothesis		
Starvation sub global	$\beta_o + \beta_1(\text{forb}) + \beta_2(\text{forb})^2 + \beta_3(\text{grass}) + \beta_4(\text{grass})^2 + \beta_5(\text{litter}) + \beta_6(\text{jdate*forb}) + \beta_7(\text{jdate*grass}) + \beta_8(\text{jdate*litter}) + \beta_9(\text{jdate*time*tsf}) + \beta_{10}(\text{jdate*sibs}) + \beta_{11}(\text{tsf*sibs})$	$\beta_1 > 0, \beta_2 > 0, \beta_3 > 0$ $\beta_4 > 0, \beta_5 > 0, \beta_6 > 0,$ $\beta_7 > 0, \beta_8 > 0, \beta_9 < 0,$ $\beta_{10} < 0, \beta_{11} < 0$
Effect of forb and litter food source availability	$\beta_o + \beta_1(\text{forb}) + \beta_2(\text{litter})$	$\beta_1 > 0, \beta_2 > 0$
Threshold effect of forb food source availability	$\beta_o + \beta_1(\text{forb})^2$	$\beta_1 > 0$
Threshold effect of grass food source availability	$\beta_o + \beta_1(\text{grass})^2$	$\beta_1 > 0$
Effect of all food source availability	$_o + \beta_1(\text{grass}) + \beta_2(\text{forb}) + \beta_3(\text{litter})$	$\beta_1 > 0, \beta_2 > 0, \beta_3 > 0$
Seasonal effects of all food source availability	$\beta_o + \beta_1(\text{jdate*grass}) + \beta_2(\text{jdate*forb}) + \beta_3(\text{jdate*litter})$	$\beta_1 > 0, \beta_2 > 0, \beta_3 > 0$
Seasonal, temporal, and age effects	$\beta_o + \beta_1(\text{jdate*time*tsf})$	$\beta_1 < 0$

Appendix 4. continued.

Description	Structure	Expected Result
Seasonal effects of brood size	$\beta_o + \beta_1(\text{jdate*sibs})$	$\beta_1 < 0$
Age and brood size effects	$\beta_o + \beta_1(\text{tsf*sibs})$	$\beta_1 < 0$
Thermal Refuge Hypothesis		
Thermal Refuge sub global	$\beta_o + \beta_1(\text{woody*temp}) + \beta_2(\text{jdate*woody}) + \beta_3(\text{jdate*temp}) + \beta_4(\text{tall*temp}) + \beta_5(\text{jdate*time*cindex}) + \beta_6(\text{jdate*time*woody}) + \beta_7(\text{temp*woody*diswdp}) + \beta_8(\text{jdate*woody*cindex})$	$\beta_1 > 0, \beta_2 > 0, \beta_3 < 0$ $\beta_4 < 0, \beta_5 > 0, \beta_6 > 0$ $\beta_7 > 0, \beta_8 > 0$
Effects of woody cover and temperature	$\beta_o + \beta_1(\text{woody*temp})$	$\beta_1 > 0$
Seasonal effects of woody cover	$\beta_o + \beta_1(\text{jdate*woody})$	$\beta_1 > 0$
Seasonal effects of temperature	$\beta_o + \beta_1(\text{jdate*temp})$	$\beta_1 < 0$
Effects of vegetation height and temperature	$\beta_o + \beta_1(\text{tall*temp})$	$\beta_1 < 0$
Seasonal and temporal effects of cover	$\beta_o + \beta_1(\text{jdate*time*cindex})$	$\beta_1 > 0$
Seasonal and temporal effects of woody cover	$\beta_o + \beta_1(\text{jdate*time*woody})$	$\beta_1 > 0$

Appendix 4. continued.

Description	Structure	Expected Result
Effects of temperature, woody cover, and proximity to woody patches	$\beta_o + \beta_1(temp*woody*diswdp)$	$\beta_1 > 0$
Seasonal effects of woody cover, shade, and concealment	$\beta_o + \beta_1(jdate*woody*cindex)$	$\beta_1 > 0$

Appendix 5. Description, structure, and expected results for *a priori* candidate models in relation to two hypotheses explaining resource selection in eastern meadowlarks in 2002.

Description	Structure	Expected Result
Predation Hypothesis		
Predation sub global	$\beta_o + \beta_1(\text{woody}) + \beta_2(\text{tall}) + \beta_3(\text{grass*tall})$	$\beta_1 < 0, \beta_2 < 0, \beta_3 < 0$
Effect of woody cover	$\beta_o + \beta_1(\text{woody})$	$\beta_1 < 0$
Effect of tall vegetation	$\beta_o + \beta_1(\text{tall})$	$\beta_1 < 0$
Effect of grass cover and vegetation height	$\beta_o + \beta_1(\text{grass*tall})$	$\beta_1 < 0$
Starvation Hypothesis		
Starvation sub global	$\beta_o + \beta_1(\text{forb}) + \beta_2(\text{litter}) + \beta_3(\text{grass})$	$\beta_1 > 0, \beta_2 > 0, \beta_3 > 0$
Effect of forb food sources	$\beta_o + \beta_1(\text{forb})$	$\beta_1 > 0$
Effect of litter food sources	$\beta_o + \beta_1(\text{litter})$	$\beta_1 > 0$
Effect of grass food sources	$\beta_o + \beta_1(\text{grass})$	$\beta_1 > 0$

Appendix 5. continued.

Description	Structure	Expected Result
Effect of litter and grass food sources	$\beta_o + \beta_1(\text{litter}) + \beta_2(\text{grass})$	$\beta_1 > 0, \beta_2 > 0$

Appendix 6. Description, structure, and expected results for *a priori* candidate models in relation to three hypotheses for explaining resource selection in eastern meadowlarks in 2003.

Description	Structure	Expected Result
Predation Hypothesis		
Predation sub global	$\beta_o + \beta_1(\text{woody}) + \beta_2(\text{tall}) + \beta_3(\text{cindex}) + \beta_4(\text{disper}) + \beta_5(\text{diswdp}) + \beta_6(\text{woody*cindex}) + \beta_7(\text{woody*diswdp}) + \beta_8(\text{woody*disper}) + \beta_9(\text{cindex*tsf})$	$\beta_1 < 0$, $\beta_2 < 0$, $\beta_3 < 0$, $\beta_4 > 0$, $\beta_5 > 0$, $\beta_6 < 0$, $\beta_7 > 0$, $\beta_8 > 0$, $\beta_9 < 0$
Effect of woody cover	$\beta_o + \beta_1(\text{woody})$	$\beta_1 < 0$
Effect of tall vegetation	$\beta_o + \beta_1(\text{tall})$	$\beta_1 < 0$
Effect of cover and shade	$\beta_o + \beta_1(\text{cindex})$	$\beta_1 < 0$
Effect of proximity to raptor perches	$\beta_o + \beta_1(\text{disper})$	$\beta_1 > 0$
Effect of proximity to woody patches	$\beta_o + \beta_1(\text{diswdp})$	$\beta_1 > 0$
Effects of all cover types	$\beta_o + \beta_1(\text{woody*cindex})$	$\beta_1 < 0$
Effects of woody cover and proximity to woody patches	$\beta_o + \beta_1(\text{woody*diswdp})$	$\beta_1 > 0$

Appendix 6. continued.

Description	Structure	Expected Result
Effect of woody cover and proximity to raptor perches	$\beta_o + \beta_1$(woody*disper)	$\beta_1 > 0$
Effect of cover and age	$\beta_o + \beta_1$(cindex*tsf)	$\beta_1 < 0$
Starvation Hypothesis		
Starvation sub global	$\beta_o + \beta_1$(forb) + β_2(litter) + β_3(grass) + β_4(jdate*forb β_5(jdate*grass) + β_6(jdate*tsf) + β_7(jdate*time)	$\beta_1 > 0$, $\beta_2 > 0$, $\beta_3 > 0$ $\beta_4 > 0$, $\beta_5 > 0$, $\beta_6 < 0$, $\beta_7 < 0$
Effect of forb food source availability	$\beta_o + \beta_1$(forb)	$\beta_1 > 0$
Effect of litter and grass food Source availability	$\beta_o + \beta_1$(litter) + β_2 (grass)	$\beta_1 > 0$, $\beta_2 > 0$
Seasonal effect of forb food source availability	$\beta_o + \beta_1$(forb*jdate)	$\beta_1 > 0$
Seasonal effect of grass food source availability	$\beta_o + \beta_1$(grass*jdate)	$\beta_1 > 0$
Seasonal and age effects	$\beta_o + \beta_1$(jdate*tsf)	$\beta_1 < 0$

Appendix 6. continued.

Description	Structure	Expected Result
Seasonal and temporal effects	$\beta_o + \beta_1(\text{jdate*time})$	$\beta_1 < 0$
Thermal Refuge Hypothesis		
Thermal Refuge sub global	$\beta_o + \beta_1(\text{woody*temp}) + \beta_2(\text{jdate*woody}) + \beta_3(\text{jdate*temp})$ $\beta_4(\text{tall*temp}) + \beta_5(\text{time*cindex}) + \beta_6(\text{tsf*temp})$	$\beta_1 > 0, \beta_2 > 0, \beta_3 < 0$ $\beta_4 < 0, \beta_5 < 0, \beta_6 < 0$
Effects of woody cover and temperature	$\beta_o + \beta_1(\text{woody*temp})$	$\beta_1 > 0$
Seasonal effects of woody cover	$\beta_o + \beta_1(\text{jdate*woody})$	$\beta_1 > 0$
Seasonal effects of temperature	$\beta_o + \beta_1(\text{jdate*temp})$	$\beta_1 < 0$
Effects of vegetation height and temperature	$\beta_o + \beta_1(\text{tall*temp})$	$\beta_1 < 0$
Temporal effects of cover	$\beta_o + \beta_1(\text{time*cindex})$	$\beta_1 < 0$
Effects of age and temperature	$\beta_o + \beta_1(\text{tsf*temp})$	$\beta_1 < 0$

Appendix 7. Description, structure, and expected results for *a priori* candidate models in relation to three hypotheses explaining resource selection in eastern meadowlarks in 2004.

Description	Structure	Expected Result
Predation Hypothesis		
Predation sub global	$\beta_o + \beta_1(\text{woody}) + \beta_2(\text{woody})^2 + \beta_3(\text{disper}) + \beta_4(\text{disper})^2 + \beta_5(\text{diswdp}) + \beta_6(\text{diswdp})^2 + \beta_7(\text{tall*jdate}) + \beta_8(\text{woody*jdate}) + \beta_9(\text{diswdp*jdate}) + \beta_{10}(\text{cindex*jdate}) + \beta_{11}(\text{woody*cindex})$	$\beta_1 < 0, \beta_2 < 0, \beta_3 > 0, \beta_4 > 0, \beta_5 > 0, \beta_6 > 0, \beta_7 < 0, \beta_8 > 0, \beta_9 > 0$ $\beta_{10} < 0, \beta_{11} < 0$
Effect of woody cover	$\beta_o + \beta_1(\text{woody})$	$\beta_1 < 0$
Threshold effect of woody cover	$\beta_o + \beta_1(\text{woody})^2$	$\beta_1 < 0$
Effect of proximity to raptor perches	$\beta_o + \beta_1(\text{disper})$	$\beta_1 > 0$
Threshold effect of proximity to raptor perches	$\beta_o + \beta_1(\text{disper})^2$	$\beta_1 > 0$
Effect of proximity to woody patches	$\beta_o + \beta_1(\text{diswdp})$	$\beta_1 > 0$
Threshold effect of proximity to woody patches	$\beta_o + \beta_1(\text{diswdp})^2$	$\beta_1 > 0$

Appendix 7. continued.

Description	Structure	Expected Result
Effects of all cover types	$\beta_o + \beta_1(\text{woody*cindex})$	$\beta_1 < 0$
Effect of woody cover and proximity to raptor perches	$\beta_o + \beta_1(\text{woody*disper})$	$\beta_1 > 0$
Seasonal effect of vegetation height	$\beta_o + \beta_1(\text{jdate*tall})$	$\beta_1 < 0$
Seasonal effect of woody cover	$\beta_o + \beta_1(\text{jdate*woody})$	$\beta_1 > 0$
Seasonal effect of proximity to woody patches	$\beta_o + \beta_1(\text{jdate*diswdp})$	$\beta_1 > 0$
Seasonal effect of cover and shade	$\beta_o + \beta_1(\text{jdate*cindex})$	$\beta_1 < 0$
Seasonal effect of woody cover and proximity to woody patches	$\beta_o + \beta_1(\text{jdate*woody}) + \beta_2(\text{jdate*diswdp})$	$\beta_1 > 0$, $\beta_1 < 0$

Appendix 7. continued.

Description	Structure	Expected Result
Starvation Hypothesis		
Starvation sub global	$\beta_o + \beta_1(\text{forb}) + \beta_2(\text{forb})^2 + \beta_3(\text{grass}) + \beta_4(\text{grass})^2 + \beta_5(\text{litter}) + \beta_6(\text{jdate*forb}) + \beta_7(\text{jdate*grass}) + \beta_8(\text{jdate*litter}) + \beta_9(\text{jdate*time*tsf}) + \beta_{10}(\text{jdate*sibs}) + \beta_{11}(\text{tsf*sibs})$	$\beta_1 > 0$, $\beta_2 > 0$, $\beta_3 > 0$ $\beta_4 > 0$, $\beta_5 < 0$, $\beta_6 > 0$, $\beta_7 > 0$, $\beta_8 < 0$, $\beta_9 < 0$, $\beta_{10} < 0$, $\beta_{11} < 0$
Effect of forb and litter food source availability	$\beta_o + \beta_1(\text{forb}) + \beta_2(\text{litter})$	$\beta_1 > 0$, $\beta_2 < 0$
Threshold effect of forb food source availability	$\beta_o + \beta_1(\text{forb})^2$	$\beta_1 > 0$
Threshold effect of grass food source availability	$\beta_o + \beta_1(\text{grass})^2$	$\beta_1 > 0$
Effect of all food source availability	$\beta_o + \beta_1(\text{grass}) + \beta_2(\text{forb}) + \beta_3(\text{litter})$	$\beta_1 > 0$, $\beta_2 > 0$, $\beta_3 < 0$
Seasonal effects of all food source availability	$\beta_o + \beta_1(\text{jdate*grass}) + \beta_2(\text{jdate*forb}) + \beta_3(\text{jdate*litter})$	$\beta_1 > 0$, $\beta_2 > 0$, $\beta_3 < 0$
Seasonal, temporal, and age effects	$\beta_o + \beta_1(\text{jdate*time*tsf})$	$\beta_1 < 0$

Appendix 7. continued.

Description	Structure	Expected Result
Seasonal effects of brood size	$\beta_o + \beta_1$(jdate*sibs)	$\beta_1 < 0$
Age and brood size effects	$\beta_o + \beta_1$(tsf*sibs)	$\beta_1 < 0$
Thermal Refuge Hypothesis		
Thermal Refuge sub global	$\beta_o + \beta_1$(woody*temp) + β_2(jdate*woody) + β_3(jdate*temp) + β_4(tall*temp) + β_5(jdate*time*cindex) + β_6(jdate*time*woody) β_7(temp*woody*diswdp) + β_8(jdate*woody*cindex)	$\beta_1 > 0$, $\beta_2 > 0$, $\beta_3 < 0$ $\beta_4 < 0$, $\beta_5 < 0$, $\beta_6 > 0$ $\beta_7 > 0$, $\beta_8 > 0$
Effects of woody cover and temperature	$\beta_o + \beta_1$(woody*temp)	$\beta_1 > 0$
Seasonal effects of woody cover	$\beta_o + \beta_1$(jdate*woody)	$\beta_1 > 0$
Seasonal effects of temperature	$\beta_o + \beta_1$(jdate*temp)	$\beta_1 < 0$
Effects of vegetation height and temperature	$\beta_o + \beta_1$(tall*temp)	$\beta_1 < 0$
Seasonal and temporal effects of cover	$\beta_o + \beta_1$(jdate*time*cindex)	$\beta_1 < 0$

Appendix 7. continued.

Description	Structure	Expected Result
Seasonal and temporal effects of woody cover	$\beta_o + \beta_1$(jdate*time*woody)	$\beta_1 > 0$
Effects of temperature, woody cover, and proximity to woody patches	$\beta_o + \beta_1$(temp*woody*diswdp)	$\beta_1 > 0$
Seasonal effects of woody cover, shade, and concealment	$\beta_o + \beta_1$(jdate*woody*cindex)	$\beta_1 > 0$

CHAPTER 2

LANDSCAPE RESOURCE SELECTION OF POST-FLEDGING GRASSLAND BIRDS IN MISSOURI

ABSTRACT

Landscape composition and structure have been associated with resource selection patterns in many terrestrial vertebrates, including grassland birds. Grassland conservation strategies often make recommendations regarding reserve design that incorporate minimum sizes and landscape compositions to benefit broad communities of birds. However, there is little information about the resource selection patterns of post-fledging grassland birds and how those requirements may differ from the adults which are used in conservation strategies. Our goal was to characterize landscape-scale resource selection patterns of two species of post-fledging grassland birds in Missouri. Specifically, we investigated landscape-scale resource use for dickcissels (*Spiza americana*) and eastern meadowlarks (*Sturnella magna*) in southwestern Missouri from 2003 to 2004. We used a matched-case control logistic regression to develop resource selection functions from radio telemetry data for individuals with > 30 detections. At one site, the presence of juvenile dickcissels was negatively related to shrubby prairie, distance to natural water sources, and distance to country roads. However, there was individual variability in the direction of the relationship for the latter two variables. At a second site with a different landscape configuration, the presence of juvenile dickcissels was negatively related to shrubby prairie, distance to forests, and distance to grazing and positively related to Ultisol soil types. There was individual variation in the relationship

between dickcissel presence and distance to forest and distance to grazing. At the first site, the presence of juvenile meadowlarks was negatively related to distance to county roads and positively related to crops and distances to artificial water sources. At the second site, meadowlark presence was positively related to pasture, distance to natural water sources and negatively related to distance to grazing. There was individual variation in the relationship between meadowlark presence and distance to water sources and grazing. The importance of forests, roads, and water were common themes across species, although the direction of the relationship between landscape features and species differed. Our results show that post-fledging resource use differs from adult resource use and that a balance between the two stages is needed in developing prairie conservation strategies.

INTRODUCTION

The influence of landscape composition and structure on resource selection in grassland and shrubland birds has received increasing attention over the last decade (e.g., Knick and Rotenberry 1995, Helzer and Jelinski 1999, Johnson and Igl 2001, Bakker et al. 2002). Although habitat loss, especially in tallgrass prairies, has often been associated with declines in grassland bird populations (Knopf 1994, Samson and Knopf 1994), several authors have suggested that fragmentation and habitat degradation were additional sources of resource alteration for grassland birds (Herkert and Knopf 1998, Johnson and Igl 2001). Researchers have suggested that three types of fragmentation effects including patch size, edge, and isolation effects could alter resources for grassland birds (Faaborg et al. 1993, Johnson and Winter 1999, Johnson 2001). In addition, habitat degradation may affect resource use by grassland birds if management or lack of management alters the functional group composition of the plant community, thereby altering or hastening successional patterns. Land ownership patterns, anthropogenic features, and management practices may affect resource use for grassland birds by causing habitat degradation or one of the three types of fragmentation effects listed above. In this study, we compare resource selection patterns of two species of grassland birds in landscapes with different suites of management practices and anthropogenic influences.

Altered disturbance regimes, lack of management leading to succession, and invasion by woody plants or non-native species are symptoms of habitat degradation) that may effect landscape composition and structure for grassland birds (Johnson and Igl 2001). For example, increased disturbance in shrub steppe communities has been associated with conversion to annual grasslands that remove the small shrub component used by grassland species such as the western meadowlark (*Sturnella neglecta*) for song perches (Knick and Rotenberry 1995). Increased grazing pressure associated with pasture is another example of an altered disturbance regime that has been negatively associated with population trends of grassland birds due to altered landscape composition (Murphy 2003). Evidence from the post-fledging literature also documents the importance of maintaining frequent disturbance in forests to create early successional communities that are critical for juvenile birds during the post-fledging dispersal period (Anders et al. 1997 Vega Rivera et al. 1998, Lang et al. 2002, Fink 2003).

The effects of patch size and edges on landscape composition and structure also have been documented for grassland birds (e.g., Herkert 1994, Helzer and Jelinski 1999, Winter et al. 2000b, Johnson and Igl 2001, Bakker et al. 2002). Samson (1980) and

Herkert (1994) provided the first evidence of area sensitivity in grassland birds by relating species richness and occurrence to fragment sizes in the Midwest. Winter and Faaborg (1999) expanded current area sensitivity concepts by showing some species, such as the dickcissel (*Spiza americana*), may only demonstrate area sensitivity at the demographic level and not at the census level. Both studies also addressed recent criticisms of fragmentation studies including passive sampling that have led to inconsistent results (Johnson 2001). Edge effects related to distances from woody features also have been documented with mixed results (Winter et al. 2000a, b; Ribic and Sample 2001, Bakker et al. 2002). For example, Winter et al. (2000b) reported that nesting success of dickcissels and Henslow's sparrows (*Ammodramus henslowii*) was lower within 50 m of shrubby edges, but not related to distance to forested areas. In South Dakota, four of the seven species studied had negative associations with the distance of woody patch edges divided by the total edge of the patch (Bakker et al. 2002). In Wisconsin, the density of grassland bird species of management concern as a group was positively associated with distance to woodlots, but the density of grasshopper sparrows (*Ammodramus savannarum*) and bobolinks (*Dolichonyx oryzivorus*) was negatively associated with woodlot area (grasshopper sparrow) and distance to hedgerows (bobolink) (Ribic and Sample 2001).

Although existing research suggests that landscape composition and structure affect resource selection of adult grassland birds, information about potential effects on post-fledging juveniles is limited (Kershner 2001). Information about natal dispersal is scant for all bird species (Part 1990, Morton 1991, Baker 1993) but best described for wood thrushes (*Hylocichla mustelina*) in North America (Anders et al. 1997, Vega Rivera et al. 1998, Powell et al. 2000, Lang et al. 2002, Fink 2003). Understanding potential influences of landscape composition and structure on post-fledging resource use is critical for questions about optimal reserve design and restoration. Current models for grassland bird conservation center around the Bird Conservation Area concept proposed by the Midwest Working Group of Partners in Flight (PIF) that is based on requirements for breeding adults (Pashley and Fitzgerald 1996). Under the model, identifying landscapes that minimize woody composition and have a minimum grass component are prioritized. Preliminary tests of the concept have produced mixed results (Winter et al. 2000a) suggesting further evaluation is warranted. As a result, our goal was to investigate resource selection patterns of post-fledging dickcissels and eastern meadowlarks (*Sturnella magna*) at the landscape scale in southwestern Missouri. Both species have shown significant declines in the U.S. between 1966 and 2003 (Sauer et al. 2004). In addition, dickcissels are currently listed on the PIF Continental Watchlist as a species with declines or high threats and in need of management (Rich et al. 2004).

METHODS

Study Sites

We conducted this study at Taberville Conservation Area (38° N, 93° W) and Wah'Kon-Tah Prairie (37°N, 94° W) in Cedar and St. Clair Counties in southwestern Missouri. Taberville Conservation area is a 680-ha prairie owned and managed by the Missouri Department of Conservation (MDC) and embedded in an agricultural matrix of crops (wheat, soybeans, and corn) and private land. Wah'Kon-Tah Prairie is a 1930-ha prairie owned by the Missouri Chapter of The Nature Conservancy (TNC) and jointly managed by MDC and TNC located at the northern periphery of El Dorado Springs, Missouri (population ~ 4,000 people) and primarily surrounded by a forest matrix with some private land interspersed. Both sites are part of a network of focal areas targeting grassland bird conservation in Missouri. Dominant land management practices included livestock grazing, prescribed burning, seed harvesting, and haying. Although grazing occurs at Taberville, it is not a dominant management practice and only a minor component of the private land in the surrounding matrix. At Wah'Kon-Tah, grazing is more prevalent on the study site and is the dominant land use in the private land surrounding the site. Woody removal is a minor management practice at both sites. At Taberville, woody removal primarily occurs along wooded draws. At Wah'Kon-Tah, woody removal occurs along woody draws, fence lines, and pasture borders with the help of local volunteer groups. The study sites are divided into management units that receive some sort of management practice (primarily prescribed burning or haying) at least once every three years. Dominant vegetation was composed of bluestem grasses and included big bluestem (*Andropogon gerardi*), little bluestem (*Schizachyrium scoparium*), and indian grass (*Sorghastrum nutans*). Forb species included coneflowers (*Echinacea* spp.), white wild indigo (*Baptisa alba*), blazing star (*Liatrus* spp.), compass plant (*Silphium laciniatum*), milkweeds (*Aesclepias* spp), and sunflowers (*Helianthus* spp.). Dominant, native woody species include smooth sumac (*Rhus glabra*), persimmon (*Diospyros virginiana*), blackberry (*Rubus* spp.), and roses (*Rosa* spp.). Throughout this chapter unless specifically stated otherwise, we use the term woody plants to refer to this group of small, shrubs that does not include invasive or encroaching woody plants or those plants for other human uses.

Bird Capture and Handling

We located nests of both species using systematic searches and haphazard walks from 0600 to 1400 hours each day between the third week of April and the second week of August from 2003 to 2004. When possible, we used behavioral cues of the parents to indicate the presence of a nest. After locating each nest, we recorded the GPS coordinates and marked the location by placing colored flagging at least 5 m away. At each nest we recorded the species, content, parental activity, and presence of any non-host eggs. If the nest contained nestlings, we attempted to age the nestlings using the presence of down, whether the eyes were open or not, the extent of pin feather development, or the presence of a full complement of feathers based on our observations (K. Suedkamp Wells, unpubl. data). We were usually successful at aging nestlings within two days of their true age depending on growth rates and weather conditions. We monitored each nest every three to four days until just prior to fledging and then switched to daily nest checks. Two to three days prior to fledging, we attached a metal USFWS band to the left leg and a unique combination of plastic, UV-resistant Darvic bands (Avinet, Dryden, New York) to the right leg and weighed each individual.

Following a modification of the Rappole and Tipton method (1991) previously described (Suedkamp Wells et al. 2003), we attached 0.7-gram transmitters with a 10-cm whip antennae (Biotrack, Dorset, United Kingdom) to the back of each bird using a leg harness. Battery life for each transmitter was expected to range between 55 and 60 days. We constructed the leg harness from cotton, elastic beading cord to allow room for growth. Using super glue (Duro, Avon, Ohio), we secured the bottom of the transmitter to the back of the bird. After attaching transmitters to each bird, we placed the brood back in the nest. Handling and processing time usually was between 2 and 5 minutes per bird.

Radiotracking

Using telemetry, we began tracking birds using homing for visual confirmation (Mech 1983) the day after attaching transmitters. If the brood remained in the nest the day after attachment, we returned each subsequent morning and began tracking when at least one brood member fledged. We tracked each bird twice daily in non-consecutive time blocks because our sampling goal was to obtain at least 50 detections per individual on a minimum of 25 individuals of both species (Garton et al. 2001). The four tracking blocks were early morning (0600 to 0930 hours), mid-morning (0930 to 1230 hours), afternoon (1230 to 1700 hours), and evening (1700 to 2130 hours). We grouped the time blocks to reflect biological activity and environmental constraints such as hot temperatures when activity is reduced. The first day of tracking for an individual occurred in the morning and afternoon time periods. On the second day tracking occurred in the mid-morning and evening time periods to capture locations representative of all diurnal activities typical of tracking studies (Garton et al. 2001). We alternated between the two daily schedules for subsequent tracking days. We avoided tracking before 0600 hours and after 2130 hours to reduce the risk of mortality when juvenile birds could not be visually located. After reaching 50 detections, each individual was tracked once daily alternating between the first two and last two periods of the day until the bird died, the transmitter was recovered, or the study period ended.

Immediately after being unable to locate an individual, we performed extensive searches of the immediate area on foot with a team of assistants. If we were unable to locate the individual, we broadened the search to include all roads within a 3.2 km of the last known location using an omni antennae mounted on the roof of a truck. We continued to search for missing individuals twice daily for one week after their disappearance. At the start of the second week, we reduced search time to one attempt per day. We also attempted to locate missing birds by flying at least 5 km strips over the study area in a helicopter twice monthly between 1 June and 30 August each year. If we recovered a transmitter, we recorded a description of the recovery site (e.g. burrow or pond), condition of the transmitter (presence of teeth marks or snake feces), and any other information that could be used to identify the potential predator or cause of death.

Resource selection measurements

At each bird location, we recorded the GPS coordinates and dominant habitat type (crop, draw, forest, pasture, shrubby prairie, or prairie). In Appendix 1, we define each habitat type designation. To characterize availability for each bird location, we used the random number generator in Excel to select five paired, random points for each telemetry location for further analysis. We accounted for potential differences in resource availability as a result of increasing flight ability by using the maximum distance between used and available locations based on our observations of movement during key developmental periods. For example, during the first two weeks post-fledge, juvenile birds are learning to fly and becoming nutritionally independent so we used the maximum movements observed for each species during that time period (139.7 m for dickcissels and 142.1 m for meadowlarks). Between weeks two and four when juveniles begin showing longer flights, we used the average maximum distances of 201.6 m and 279.4 m, respectively. After six weeks when juveniles were generally not interacting with their parents or siblings, we used the average maximum distances of 877.2 m and 1323.0 m, respectively.

We selected candidate variables and biologically interpretable interactions around two factors (starvation and predation) that have been associated with post-fledging resource selection (Anders et al. 1998, Vega Rivera et al. 1998). The first factor is the Starvation Hypothesis that states that post-fledging grassland birds select resources at the landscape scale to provide foraging opportunities. The second factor is the Predation Hypothesis that states that post-fledging grassland birds select resources at the landscape scale to avoid risk of predation. Variables associated with the Starvation Hypothesis included crop and prairie habitat types, Mollisol and Alfisol soils, and distances to hay roads, grazed units, and crops. Variables associated with the Predation Hypothesis included draw, forest, and shrubby habitat types, Inceptisol and Ultisol soils, the perimeter to area ratio (PAR), and distances to forests, draws, water sources, country roads, and unit boundaries. We associated Mollisol and Alfisol soils with the Starvation Hypothesis because they generally support prairies and crops, respectively (Miller and Donahue 1995). Likewise, we associated Inceptisol and Ultisol soils with the Predation Hypothesis because they are usually associated with brushy cover or forested areas (Miller and Donahue 1995) that may be more likely to support predators.

We used Arc View 3.3 (Environmental Systems Research Institute, Redlands, California) to overlay themes containing habitat type, soil type, and landscape feature information. We used infrared imagery completed in 2003 from the National Agricultural Imagery Program (NAIP) available through Missouri Spatial Data Information Service (MSDIS) web site to digitize all habitat types, woody draws, and interior hay roads. We obtained all other information regarding landscape features such as soil types, ponds, streams, and country roads from existing themes on the MSDIS site. For forest cover, we modified the coverage available from the MSDIS site from 1983 to include the recent extent of mature trees around both sites. We used the X Tools extension (Oregon Department of Forestry, Salem, Oregon) in Arc View to calculate the perimeter to area ratio (PAR) of each management unit or private property parcel and the distance to each landscape feature described in Appendix 1. Distances to artificial water sources (disawater) usually included stock ponds or man-made lakes while distances to water (diswater) included naturally occurring water features such as streams. Country roads were either paved or gravel roads with one lane in either direction but hay roads were generally interior dirt roads used to manage and navigate within the sites. We calculated distance to the nearest unit boundary (disub) based on distance to the nearest management unit on public land or distance to the nearest field or parcel border on private land. In general, Taberville had fewer soil types, less pasture and shrubby prairie, longer distances to artificial water sources (disawater), shorter distances to naturally occurring water sources (diswater), and longer distances to county roads compared to Wah'Kon-Tah (Table 1).

Resource Selection Analysis

We used matched-case control, logistic regression (Vierkant et al. 1998, Allison 1999, Hosmer and Lemeshow 2000) with a conditional logit model to estimate resource selection in juvenile dickcissels and eastern meadowlarks on two study sites from 2003 – 2004. We elected to model resource selection on the two sites separately for this chapter because of differences in dominant management practices and the surrounding matrix (agriculture versus forest). Although we could have used an Information Theoretic approach similar to the methods we used in Chapter 1, we were unsatisfied with the data loss (20% to 40%) that would have resulted from combining unequal sample sizes of individuals across sites and years and potential differences in resource availability. Although not including site as a random effect limits our ability to make inferences to other sites, we feel our approach was justified given the reasons described above. We only included those individuals with ≥ 30 detections in the analysis because they were most likely to have survived past the parental dependence phase and this sample size corresponded to the minimum sample size recommended for movement analysis described in Chapter 3.

We used PROC PHREG in SAS 8.0 (SAS Institute Cary, North Carolina) to maintain the paired structure between bird locations (cases) and a subsample of paired random points (controls) to appropriately define availability for each individual. We fit multivariate resource selection models containing all the single variable predictors and biologically interpretable interactions (e.g., distance to country roads * distance to hay roads) to data from each individual separately. To illustrate, the multivariate model fit to every individual (assuming each individual was located on a site where all variables were present) included ten habitat or soil types (crop, draw, forest, pasture, shrubby prairie, prairie, alfisols, inceptisols, ultisols, and mollisols), the perimeter to area ratio of the patch or management unit, nine measures of distance to landscape features (distances to forests, draws, artificial water sources, natural water sources, county roads, hay roads, unit boundaries, crops, and grazing), and four interactions habitat or soil types and distance measures (edge to area ratio, roads, wooded, and water). We used *t*-ratio tests to assess the significance of each variable or interaction and retained those predictors with *P* values < 0.05 (Hosmer and Lemeshow 2000). To control for potential autocorrelation among multiple brood members in the sample, we randomly selected one individual per brood for inclusion in further analysis. To generate population-level coefficients, we used a bootstrapping approach (Mooney and Duval 1993) to estimate parameter coefficients and their 95% confidence intervals as suggested by Millspaugh et al. (2005). We generated 1,000 bootstrap replications (Mooney and Duval 1993) for each parameter in the individual resource selection function using the sample size of individuals where that parameter was significant in SYSTAT (SPSS 1999). For example, if 12 individuals had crop as a significant variable in their resource selection, we drew 1,000 bootstrap replications using a sample size of 12. We obtained the 95% confidence intervals for each coefficient by sorting the bootstrap replicates and extracting the 25th estimate (lower 95% confidence interval) and the 975th estimate (upper 95% confidence interval). Finally, we used the population-level parameter coefficients to calculate odds ratios and the 95% confidence interval around each odds ratio (Allison 1999). We interpret results for predictors where the 95% confidence interval around the odds ratio did not include one. We evaluated model fit by comparing the global model to the null model for each species and site combination and assessed the percent correct classification using concordance (Allison 1999).

RESULTS

We attached transmitters to 198 and 135 individual dickcissels and eastern meadowlarks, respectively between 2003 and 2004. The subset that survived for ≥ 30 detections was 50 dickcissels from 31 broods and 56 meadowlarks from 27 broods, respectively. Predation was the main cause of death for individuals with < 30 detections resulting in 75% and 59% mortality, respectively for dickcissels and meadowlarks during that period. Snakes were the dominant predator type observed taking juveniles including bullsnakes (*Pituophus catenifer*), speckled king snakes (*Lampropeltis getula*), and prairie king snakes (*Lampropeltis callagaster*).

Resource selection patterns by juvenile dickcissels were variable and largely dependent on differences in landscape context between the two sites (Table 1). Model fit and predictive power of population-level resource selection models for dickcissels at Taberville were adequate ($X^2 = 193$, $P < 0.001$; concordance = 63%). Shrubby prairies, distance to natural water sources, and distance to country roads were important predictors of the presence of juvenile dickcissels at Taberville (i.e., 95% confidence intervals around the odds ratio did not include one) (Table 2). Landscapes with smaller amounts of shrubby prairie (Table 3, Fig. 1), and shorter distances to natural water sources and county roads (Table 3, Fig. 2) were associated with an increase in dickcissel presence. Interpreting the odds ratio for the distance to natural water sources (diswater) indicates increasing the distance to a natural water source by 1 m would decrease the odds of dickcissel presence by 16%. Similarly, increasing the distance to a country road by 1 m would decrease the odds of dickcissel presence by 3%. However, examining the number of individuals that were significantly positive or negatively related to either variable (Table 3) shows individual variation in resource selection.

Model fit and predictive power of population-level resource selection models for dickcissels at Wah'Kon-Tah were adequate ($X^2 = 92.9$, $P < 0.001$; concordance = 70%). Shrubby prairie, Ultisol soils, the distance to forests, and the distance to grazing were important predictors of dickcissel presence (Table 2). Smaller amounts of shrubby prairie (Table 4, Fig. 3), and shorter distances to forests and grazing (Table 4, Fig. 2) were associated with an increase in dickcissel presence. Interpreting odds ratios show that increasing the distance to a forest by 1 m would decrease the odds of dickcissel presence by 35%. Increasing the distance to grazed units by 1 m would also decrease the odds of dickcissel presence by 5%. Individual variability in resource selection patterns also was evident at this site in relation to distances to forests and grazing (Table 4).

Patterns of resource selection also reflected differences in landscape context for juvenile eastern meadowlarks (Table 5). Model fit and predictive power of population-level resource selection models for meadowlarks at Taberville were adequate ($X^2 = 67$, $P < 0.001$; concordance = 69%). Crops, the distance to artificial water sources, distance to county roads, and the roads interaction term at Taberville were all important predictors of meadowlark presence. Larger amounts of crops (Table 6, Fig. 4), longer distances to artificial water sources, shorter distances to county roads, and shorter distances to the combination of county and hay roads were associated with an increase in meadowlark presence (Table 6, Fig. 5). Interpreting odds ratios indicated that increasing the distance to artificial water sources by 1 m would increase the odds of meadowlark presence by 4% and that increasing the distance to country roads by 1 m would decrease the odds of meadowlark presence by 2%. However, individual variability was less as indicated by the large number of individuals associated with the coefficient in the same direction as the population level resource selection pattern (Table 6).

Model fit and predictive power of population-level resource selection models for meadowlarks at Wah'Kon-Tah were adequate ($X^2 = 74$, $P < 0.001$; concordance = 65%).

Pasture availability, distance to natural water sources, and distance to grazing were important predictors of the presence of meadowlarks (Table 5). Larger amounts of pasture (Table 7, Fig. 6), longer distances to natural water sources, and shorter distances to grazing were associated with an increase in meadowlark presence (Table 7, Fig. 5). Increasing the distance to an artificial water source by 1 m would increase the odds of meadowlark presence by 18% but increasing the distance to grazing by 1 m would decrease the odds of meadowlark presence by 3%. Individual variability in resource selection patterns was particularly evident for distance to grazing because four individuals had positive associations and eight individuals had negative associations (Table 7).

Overall, our results demonstrate three common themes across species. First, differences in landscape context between the sites related to the surrounding matrix and dominant management practices, were reflected in resource selection patterns. For example, crops were much more available at Taberville compared to Wah'Kon-Tah and were preferentially selected. Second, although researchers often ignore individual variability by generating population-level resource selection functions, individual variability within species was apparent. Finally, woody habitat components (forests and shrubby prairie), water sources, and roads played important roles for resource selection in both species, although the direction of the relationship often differed between species.

DISCUSSION

Our results demonstrate the importance of landscape context in terms of the surrounding matrix and dominant management practices for two species of post-fledging grassland birds. Although both of our sites are located within 20 km of each other in the tallgrass prairies of southwestern Missouri and essentially managed as identical landscapes, our results illustrate how differences in landscape composition and structure influence resource selection patterns of post-fledging grassland birds. In general, the most prevalent differences between the two landscapes relate to the dominant land uses (grazing versus crops) and their effects on woody vegetation, the availability of water sources (both natural and artificial), and proximity to roads.

Landscape composition

The negative association between dickcissel resource selection and shrubby cover across both sites parallels other landscape studies with this species (Hughes et al. 1999, McCoy 2000, Walk and Warner 2000, Winter et al. 2000b) and is consistent with our Predation Hypothesis. For example, dickcissel abundance was negatively related to the percent woody cover within 800 m and the percent wooded perimeter of Conservation Reserve Program (CRP) fields in northeastern Kansas (Hughes et al. 1999). Presence of dickcissels also was negatively related to the amount of woody edge in CRP fields of northeastern Missouri (McCoy 2000). The frequency of encountering dickcissels also was highest in grazed and mowed warm-season grass habitats targeting woody reduction in the tallgrass prairies of Illinois (Walk and Warner 1999). The frequency of nest placement, nest success, and brood parasitism on dickcissels in a nearby county of southwestern Missouri were also lower within 50 m of shrubland and forest habitats (Winter et al. 2000b). Although increases in mammalian predators in edge habitats between grasslands and wooded areas have been implicated as the responsible mechanism (Burger et al. 1994, Winter et al. 2000b), evidence from video cameras has shown that snakes are the dominant predator of songbird nests in fields of Missouri

(Thompson et al. 1999) and that the dominant predator may shift from small mammals in northern grasslands (Pietz and Granfors 2000, Renfrew and Ribic 2003) to snakes in southern grasslands (K. Suedkamp Wells, unpublished manuscript, L. Wolfenbarger, pers. comm.).

The importance of crops for post-fledging eastern meadowlarks also is consistent with existing information on resource selection by this species and other grassland birds (McKee 1995, Walk and Warner 2000, Kershner 2001, Ribic and Sample 2001). The positive relationship between juvenile meadowlarks and crops at Taberville is likely a result of the agricultural matrix of private land surrounding this site where we frequently observed female meadowlarks and their young foraging. In the only other study on juvenile meadowlarks that we are aware of, Kershner (2001) reported a preference for soybean crops but avoidance of corn fields in Illinois. A study on greater prairie chickens (*Tympanuchus cupido*) (McKee 1995) on the same sites reported similar observations and also suggested that crops were an important food source. In a recent analysis of the effects of changing farmland structure on grassland bird population trends, Murphy (2003) showed that declines in harvestable crops were associated with decreasing population trends in grassland birds, especially short distance migrants. The compilation of existing evidence suggests that the Starvation Hypothesis needs further evaluation and may explain the mechanism behind the selection of agricultural crops.

The importance of pasture habitat types on the landscape for juvenile meadowlarks is also consistent with other information on resource selection for this species (Walk and Warner 2000, Ribic and Sample 2001) but not consistent with the only other study on juvenile meadowlarks (Kershner 2001). Although pasture was only a significant predictor of resource selection at one site (Wah'Kon-Tah), it was the most common habitat type used in comparison to the prairie reference type at both sites. In contrast, juvenile meadowlarks in Illinois used pastures in equal proportions to their availability (Kershner 2001). However, the frequency of encountering adult eastern meadowlarks was highest in hayed and grazed cool-season grass habitats on the same site (Walk and Warner 2000). In addition, the density of breeding pairs of meadowlarks in Wisconsin also was highest in dry pasture compared to hay fields, cool-season grass, and dry prairie (Ribic and Sample 2001). If the availability of habitat types required by post-fledging birds affects nest-site selection for breeding adults, then associations of adult birds with pasture habitat types may reflect post-fledging requirements. Although Taberville and the site described by Kershner (2001) are both dominated by similar crop types, we suggest other landscape characteristics such as patch size, management rotations, and intensity of production may explain the difference in results. Given the variability of results in similar Midwestern states, future research should continue to focus on elucidating the role of landscape composition and structure on post-fledging resource use.

Landscape structure

The four landscape features that were significant predictors of dickcissel resource use were distance to forests, distance to natural water sources, distance to county roads, and distance to grazed areas. The negative association between dickcissel resource use and distance to forests at Wah'Kon-Tah is interesting because it contrasts with existing data documenting negative relationships with woody cover (Hughes et al. 1999, McCoy 2000, Winter et al. 2000b). However, we suggest this result may be an artifact of the surrounding landscape because forest cover is the primary habitat type surrounding Wak'Kon-Tah and may mean juvenile birds are required to disperse through it to reach other suitable habitat types. Another possibility is that forested areas offer cooler temperatures during the summer which would be consistent with our results from another resource selection analysis at the local scale indicating support for the Thermal Refuge Hypothesis in one year (see Chapter 1). Further work evaluating the impacts of forest cover on juvenile resource selection is needed to determine whether it is safe to generalize from adult resource selection patterns.

The negative relationship between distance to water sources (primarily streams) and distance to country roads is somewhat counterintuitive because of the existing literature documenting the effects of edges and roads on this species (Winter et al. 2000b). However, we suggest that streams were favorable because they provided cooler temperatures during hot periods and that roads were favorable because they are associated with song perches. We frequently observed juvenile dickcissels moving into woody draws lining stream corridors during hot periods, especially during July and August (K. Suedkamp Wells, unpubl. data). Juvenile birds that moved into woody draws often used them as corridors to travel from one patch to another and often only entered interior grassland patches during the morning for foraging (K. Suedkamp Wells, unpubl. data). The influence of microclimate on other aspects of resource selection for grassland birds also has been documented (With and Webb 1993, Nelson and Martin 1999, Suedkamp 2000, Lusk et al. 2003). We also note that individual variability is important to understand because four individuals had positive relationships and 9 had negative relationships with distance to water sources. Likewise, individual variability could partly explain the negative relationship between distance to county roads and juvenile dickcissel resource selection. In this case, 3 individuals had positive relationships and eight individuals had negative relationships with distance to country roads. We suggest that individual differences are due to differential availability of song and feeding perches for juveniles because the individuals positively related to distances from county roads were located in areas with more trees, power lines, and fence posts (K. Suedkamp Wells, unpublished data). Therefore, the association with roads is likely the result of structures associated with roads (like the features mentioned above) instead of with the roads themselves.

Distance to grazed habitat types at Wah'Kon-Tah was likely important because of the associated changes in vegetation. In contrast to Taberville, grazed pastures at Wah'Kon-Tah were frequently dominated by shrubs and had minimal grass cover which provided suitable nesting substrates and minimized thick vegetation cover that may have concealed predators. Although this result is consistent with another analysis of ours examining local resource selection by juveniles of this species (see Chapter 1), it is not consistent with other research showing negative relationships between woody components and adult resource selection (Hughes et al. 1999, McCoy 2000, Winter et al. 2000b). However, we suggest requirements for post-fledging juveniles are different from breeding adults and that the former need additional research for effective grassland bird conservation strategies.

Increasing distances to natural (primarily streams) and artificial water sources (primarily livestock ponds) were likely positively associated with the presence of juvenile meadowlarks because they are attractive to predators, which is consistent with our

Predation Hypothesis. For example, Dijak et al. (2000) reported that the abundance of raccoons (*Procyon lotor*) and opossums (*Didelphis virginiana*) were positively related to stream density in Missouri forests. The movement of striped skunks (*Mephitis mephitis*) has also been associated with wetland edges in Iowa grasslands (Phillips et al. 2004). River corridors also have been associated with the dominant predator species on our site including the bullsnake for burrowing and foraging (Kissner and Nicholson 2003). Habitat edges also have been associated with providing important thermoregulation opportunities for black rat snakes, which are considered dietary generalists and opportunistic hunters of birds and their nests (Blouin-Demers and Weatherhead 2001a, b, Weatherhead et al. 2003). Although we observed prairie king snakes consuming juvenile birds of both species, Olson and Warner (2001) were only able to document small mammals in their diet. In Missouri, prairie king snakes were observed depredating 6 songbird nests in old fields compared to 1 nest in forests (Thompson and Burhans 2003). Speckled snakes were observed depredating equal numbers of nests (one each) in old fields and forests in the same study.

The negative relationship between distance to county roads and distance to grazed areas with meadowlark presence reflects the importance of elevated perches. County roads at our study sites are usually associated with woody fence rows, power lines, and telephone poles that we frequently observed meadowlarks using for perches. In addition, we observed more fence lines and shrubs in grazed areas, which were frequently used as perches by juveniles and adults (K. Suedkamp Wells, unpublished data). Both sexes of meadowlarks often had favorite shrub perches that were the last place they landed prior to jumping down on the ground to walk in and deliver food to their young. Knick and

Rotenberry (1995) also documented the importance of shrubs for their congener, the western meadowlark (*Sturnella neglecta*) in shrub steppe habitats of the Intermountain West.

Our results also demonstrate the importance of individual variability in resource selection patterns within and across species. Typically, researchers pool data across individuals, which obscures individual differences in behavior and resource patterns. However, recent improvements in resource selection studies have provided methods for scaling up from the individual to population level to address this issue (Marzluff et al. 2004, Millspaugh et al. 2005). In this study, individual variability in resource use was evident for both species in relation to the importance of forests, water sources, and roads. In several cases, roads were a significant predictor of the presence of both species at the individual level when adequate perching sites were lacking. As a result, we frequently observed parents associated with fledglings of both species using power lines and fence poles as perches for food delivery in the absence of small shrubs or other elevated perch sites (K. Suedkamp Wells, unpubl. data). Steller's jays (*Cyanocitta stelleri*) also have shown individual variability in patterns of resource use (Marzluff et al. 2004). In the Olympic peninsula, jays located closer to human activity used high-contrast edges more than jays located farther from human activities. As a result, the authors recommend accounting for individual variability in resource selection prior to using population-level resource selection projections on a landscape.

CONSERVATION IMPLICATIONS

To identify conservation strategies that balance post-fledging and breeding requirements, we recommend addressing two research needs. First, researchers need to invest more resources to understand the differences in threats to grassland conservation from fragmentation and habitat degradation. Documentation of area effects on grassland birds (Herkert and Knopf 1998, Vickery and Herkert 2001) has been the primary impetus behind conservation strategies that emphasize protecting large blocks of grassland habitats. However, we believe threats due to habitat degradation may rival those of fragmentation and have received comparatively little attention.

Woody encroachment, invasion by non-native grasses, and inappropriate management practices are symptoms of habitat degradation that have been identified as issues of high conservation concern for the Prairie Avifaunal Biome according to PIF (Rich et al. 2004). Our results demonstrate the negative effects of large, homogenous shrubby habitats. However, prior work (see Chapter 1) shows that micro-scale woody features, including small shrubs, are actually beneficial. To combat woody encroachment while maintaining a small component of woody features, we recommend that grassland conservation strategies target the creation of heterogeneous plant communities where possible. To obtain a mix of vegetation structure and composition, we suggest that managers combine strategies such as prescribed burning and grazing. In addition, we recommend that managers experiment with the timing and intensity of management to achieve a balance between nesting and post-fledging requirements as logistical planning allows. In addition, successful models of management should be shared among grassland managers and researchers to demonstrate possible solutions. One solution being pursued by The Nature Conservancy is based on the model from the Tallgrass Prairie Preserve in northeastern Oklahoma. At this site, managers and researchers have been using combinations of patch burning and grazing techniques to achieve variability, or heterogeneity in vegetation structure and composition (Fuhlendorf and Engle 2001). Although we are aware that our site and the model site described above are large prairies (>1,000 ha), we suggest that managers strive for a mix of management practices on prairies of reasonable size (> 50 ha) with the recognition that planning efforts may prioritize conservation strategies on larger fragments where economies of scale make intensive management more palatable.

The importance of anthropogenic features in our results including natural water sources, forests, and roads are also indicators of habitat degradation. For example, the positive relationship between the presence of juvenile birds and county roads is most likely a result of missing habitat features that are associated with county roads, such as elevated perch sites. Although adjacent prairie patches often appeared to meet conservation goals based on the dominance of grass and forb components, the distribution of adequate woody features was confined to fence lines, the sides of roads, or woody draws, which probably does not reflect historical patterns of woody plant distribution. For sites with similar symptoms of habitat degradation, we recommend evaluating conservation practices to insure that habitat features needed during the nesting and post-fledging periods are satisfied on relevant scales for grassland bird species.

The second research need is to improve our basic knowledge of resource use during the post-fledging period. Although an increasing number of authors have recognized the importance of this critical life history stage for bird populations (Part 1990, Morton et al. 1991, Baker 1993, Anders et al. 1997, Vega Rivera et al. 1998, Powell et al. 2000, Lang et al. 2002, Fink 2003), we lack information on basic life history requirements and resource selection patterns during the post-fledging phase for many species. Coincidentally, many of those same species are listed on the PIF Continental Watchlist (Rich et al. 2004). Where possible, we recommend that researchers consider collecting information on basic life history and resource selection for post-fledging birds in conjunction with breeding studies. Research on post-fledging birds in grasslands, western shrublands, shrub/successional, and riparian communities would be particularly beneficial because they represent high proportions of birds species listed on the PIF Watchlist.

LITERATURE CITED

Allison, P. D. 1999. Logistic regression using the SAS system. SAS Institute, Cary, North Carolina, USA.

Anders, A. D., D. C. Dearborn, J. Faaborg, and F. R. Thompson III. 1997. Juvenile survival in a population of Neotropical Migrant birds. Conservation Biology 11:698-707.

Baker, R. R. 1993. The function of post-fledging exploration: a pilot study of three species of passerines ringed in Britain. Ornis Scandinavica 24:71-79.

Bakker, K. K., D. E. Naugle, and K. F. Higgins. 2002. Incorporating landscape attributes into models for migratory grassland bird conservation. Conservation Biology 16:1638-1646.

Blouin-Demers, G., and P. J. Weatherhead. 2001a. An experimental test of the link Between foraging, habitat selection, and thermoregulation in black rat snakes *Elaphe obsolete obsoleta*. Journal of Animal Ecology 70:1006-1013.

Blouin-Demers, G., and P. J. Weatherhead. 2001b. Habitat use by black rat snakes (*Elaphe obsoleta obsoleta*) in fragmented forests. Ecology 82:2882-2896.

Burger, L. D., L. W. Burger, and J. Faaborg. 1994. Effects of prairie fragments on predation of artificial nests. Journal of Wildlife Management 58:249-254.

Dijak, W. D., and F. R. Thompson. 2000. Landscape and edge effects on the distribution of mammalian predators in Missouri. Journal of Wildlife Management 64:209-216.

Faaborg, J., M. Brittingham, T. Donovan, and J. Blake. 1993. Habitat fragmentation in the temperate zone: a perspective for managers. *In* Status and management of Neotropical migratory birds. D. M. Finch and P. W. Stangel, editors. U. S. Forest

Service General Technical Report RM-229.

Fink, M. L. 2003. Post-fledging ecology of juvenile Wood thrush in fragmented and contiguous landscapes. Ph.D. Dissertation, University of Missouri, Columbia, Missouri, USA.

Fuhlendorf, S. D., and D. M. Engle. 2001. Restoring heterogeneity on rangelands: ecosystem management based on evolutionary grazing patterns. BioScience 51:625-632.

Garton, E. O., M. J. Wisdom, F. A. Leban, and B. K. Johnson. 2001. Experimental design for radiotelemetry studies. Pages 16 - 44 in J. J. Millspaugh and J. M. Marzluff, editors. Wildlife Radiotelemetry: Design and Analysis. Academic Press, Inc. San Diego, California, USA.

Helzer, C. J., and D. E. Jelinski. 1999. The relative importance of patch area and perimeter-area ratio to grassland breeding birds. Ecological Applications 9:1448-1458.

Herkert, J. R. 1994. The effects of habitat fragmentation on midwestern grassland bird communities. Ecological Applications 4:461-471.

Herkert, J. R. and F. L. Knopf. 1998. Research needs for grassland bird conservation. *In* Avian conservation: research and management. J. M. Marzluff and R. Sallabanks, editors. Island Press, Washington, D.C., USA.

Hosmer, D. W. and S. Lemeshow. 2000. Applied logistic regression. Second edition. Wiley & Sons, New York, New York, USA.

Hughes, J. P., R. J. Robel, K. E. Kemp, and J. L. Zimmerman. 1999. Effects of habitat on dickcissel abundance and nest success in Conservation Reserve Program fields in Kansas. Journal of Wildlife Management 63:523-529.

Johnson, D. H. 2001. Habitat fragmentation effects on birds in grassland and wetlands: a critique of our knowledge. Great Plains Research 11:211-231.

Johnson, D. H., and M. Winter. 1999. Reserve design for grasslands: considerations for bird populations. Proceedings of the George Wright Society Biennial Conference. 10:391-396.

Johnson, D. H., and L. D. Igl. 2001. Area requirements of grassland birds: a regional perspective. Auk 118:24-34.

Johnson, H. G., and S. A. Temple. 1990. Nest predation and brood parasitism of tallgrass prairie birds. Journal of Wildlife Management 54:106-111.

Kershner, E. L. 2001. Conservation of grassland birds in an agricultural landscape: the importance of habitat availability and demography. Ph.D. Dissertation, University of Illinois at Urbana-Champaign., Urbana, Illinois, USA.

Kissner, K. J., and J. Nicholson. 2003. Bullsnakes (*Pituophis catenifer sayi*) in Alberta: literature review and data compilation. Alberta Fish and Wildlife Division. Alberta Species at Risk Reports, Number 62. Edmonton, Alberta, Canada.

Knick, S. T., and J. T. Rotenberry. 1995. Landscape characteristics of fragmented shrub steppe habitats and breeding passerine birds. Conservation Biology 9:1059-1071.

Knopf, F. L. 1994. Avian assemblages on altered grasslands. Studies in Avian Biology 15:247-257.

Lang, J. D., L. A. Powell, D. G. Krementz, and M. J. Conroy. 2002. Wood thrush movements and habitat use: effects of forest management for red-cockaded woodpeckers. Auk 119:109-124.

Lusk, J. J., K. M. Suedkamp Wells, F. S. Guthery, and S. D. Fuhlendorf. 2003. Lark sparrows (*Chondestes grammacus*) nest-site selection and success in a mixed-grass prairie. Auk 120:120-129.

Marzluff, J. M., J. J. Millspaugh, P. Hurvitz, and M. S. Handcock. 2004. Relating resources to a probabilistic measure of space use: forest fragments and Steller's jays. Ecology 85:1411-1427.

McCoy, T. D. 2000. Effects of landscape composition and multi-scale habitat characteristics on the grassland bird community. Unpublished PhD dissertation. University of Missouri, Columbia, Missouri, USA.

McKee, G. 1995. Ecology of Greater prairie chickens in relation to habitat characteristics in southwestern Missouri. Unpublished M.S. thesis. University of Missouri, Columbia, Missouri, USA.

Mech, L. D. 1983. Handbook of animal radio-tracking. University of Minnesota Press, Minneapolis, Minnesota, USA.

Miller, R. W., and R. L. Donahue. 1995. Soils in our environment. Seventh edition. Prentice- Hall Inc., Englewood Cliffs, New Jersey, USA.

Millspaugh, J. J., R. M. Nielson, L. McDonald, J. M. Marzluff, R. A. Gitzen, C. D. Rittenhouse, M. W. Hubbard, and S. L. Sheriff. 2005. Analysis of resource selection using utilization distributions. Journal of Wildlife Management *In Press.*

Mooney, C. Z., and R. D. Duval. 1993. Bootstrapping: a nonparametric approach to statistical inference. Sage University Paper Series. Quantitative Applications in the Social Sciences, Number 95.

Morton, M. L., M. W. Wakamatsu, M.E. Pereyra, and G. A Morton. 1991. Postfledging dispersal, habitat imprinting, and philopatry in a montane, migratory sparrow. Ornis Scandinavica 22:98-106.

Murphy, M. T. 2003. Avian population trends within the evolving agricultural landscapes of eastern and central United States. Auk 120:20-34.

Olson, D. J., and R. E. Warner. 2001. Grassland snakes. Diet. Herpetological Review 32:186-187.

Nelson, K. J., and K. Martin. 1999. Thermal aspects of nest-site location for Vesper Sparrows and Horned Larks in British Columbia. Studies in Avian Biology 19:137-143.

Part, T. 1990. Natal dispersal in the collared flycatcher: possible causes and reproductive consequences. Ornis Scandinavica 21:83-88.

Pashley, D. and J. Fitzgerald. 1996. Results of Prairie Pothole Joint Venture/Partners in Flight meeting, July 11 – 12. Unpublished memorandum.

Phillips, M. L., W. L. Clark, S. M. Nusser, M. A. Sovado, and R. J. Greenwood. 2004. Analysis of predator movements in prairie landscapes with contrasting prairie composition. Journal of Mammalogy 85:187-195.

Pietz, P. J., and D. A. Granfors. 2000. Identifying predators and fates of grassland Passerine nests using miniature video cameras. Journal of Wildlife Management 64:71-87.

Powell, L. A., J. D. Lang, M. J. Conroy, and D. G. Krementz. 2000. Effects of forest management on density, survival, and population growth of Wood thrushes. Journal of Wildlife Management 64:11-23.

Rappole, J. H., and A. R. Tipton. 1991. New harness design for attachment of radio transmitters to small passerines. Journal of Field Ornithology 62:335-337.

Ribic, C. A., and D. W. Sample. 2001. Associations of grassland birds with landscape factors in southern Wisconsin. American Midland Naturalist 146:105-121.

Rich, T. D., C. J. Beardmore, H. Berlanga, P. J. Blancher, M. S. W. Bradstreet, G. S. Butcher, D. W. Demarest, E. H. Dunn, W. C. Hunter, E. E. Inigo-Elias, J. A. Kennedy, A. M. Martell, A. O. Panjabi, D. N. Pashley, K. V. Rosenberg, C. W. Rustay, J. S. Wendt, and T. C. Will. 2004. Partners in Flight North American Landbird Conservation Plan. Cornell Lab or Ornithology, Ithaca, New York, USA.

Samson, F. B. 1980. Island biogeography and the conservation of prairie birds. Proceedings of the North American Prairie Conference 7:293-299.

____, and F. Knopf. 1994. Prairie conservation in North America. BioScience 44:418-421.

SAS Institute. 2001. SAS, version 8.0. SAS Institute, Cary, North Carolina, USA.

Sauer, J. R., J. E. Hines, and J. Fallon. 2004. The North American Breeding Bird Survey, Results and Analysis 1966 - 2003. Version 2004.1. USGS Patuxent Wildlife Research Center, Laurel, Maryland, USA.

SPSS. 1999. SYSTAT, version 9.0. SPSS, Chicago, Illinois, USA.

Suedkamp, K. M. 2000. Effects of temperature on nest-site selection of ground-nesting grassland birds. M.S. Thesis, Oklahoma State University, Stillwater, Oklahoma, USA.

Suedkamp Wells, K. M., B. E. Washburn, J. J. Millspaugh, M. R. Ryan, and M. W. Hubbard. 2003. Effects of radio-transmitters on fecal glucocorticoid levels in captive dickcissels. Condor 105:805-810.

Thompson, F. R. III, W. Dijak, and D. E. Burhans. 1999. Video identification of predators at songbird nests in old fields. Auk 116:259-264.

_____, and D. E. Burhans. 2003. Predation of songbird nests differs by predator and between field and forest habitats. Journal of Wildlife Management 67:408-416.

Vega Rivera, J. H., J.H. Rappole, W. J. McShea, and C. A. Haas. 1998. Wood thrush postfledging movements and habitat use in northern Virginia. Condor 100:69-78.

Vickery, P. D., and J. R. Herkert. 2001. Recent advances in grassland bird research: where do we go from here? Auk 118:11-15.

Vierkant, R. A., T. M. Therneau, J. L. Kosanke, and J. M. Naessens. 1997. A SAS macro to analyze data from a matched or finely stratified case-control design. Proceedings of the SAS Users Group International Conference, Paper 285. Miami Beach, Florida, USA. February 11 – 14.

Walk, J. W., and R. E. Warner. 2000. Grassland management for the conservation of Songbirds in the Midwestern USA. Biological Conservation 94:165-172.

Weatherhead, P. J., G. Blouin-Demers, and K. M. Cavey. 2003. Seasonal and prey-size dietary patterns of black ratsnakes (*Elaphe obsoleta obsoleta*). American Midland Naturalist 150:275-281.

Winter, M., and J. Faaborg. 1999. Patterns of area sensitivity in grassland-nesting birds. Conservation Biology 13:1424-1436.

Winter, M., D. H. Johnson, T. M. Donovan, and W. D. Svedarsky. 2000a. Evaluation of the Bird Conservation Area Concept in the Northern Tallgrass Prairie. Annual Report: 2000. Northern Prairie Wildlife Research Center, U.S. Geological Survey, Jamestown, ND: Northern Prairie Wildlife Research Center Home Page. http://www.npwrc.usgs.gov/resource/2001/bca2000/bca2000.htm (Version 26 JAN 2001).

Winter, M., D. H. Johnson, and J. Faaborg. 2000b. Evidence for edge effects on multiple levels in tallgrass prairie. Condor 102:256-266.

With, K. A., and D. R. Webb. 1993. Microclimate of ground nests: the relative importance of radiative cover and wind breaks for three grassland species. Condor 95:401 - 413.

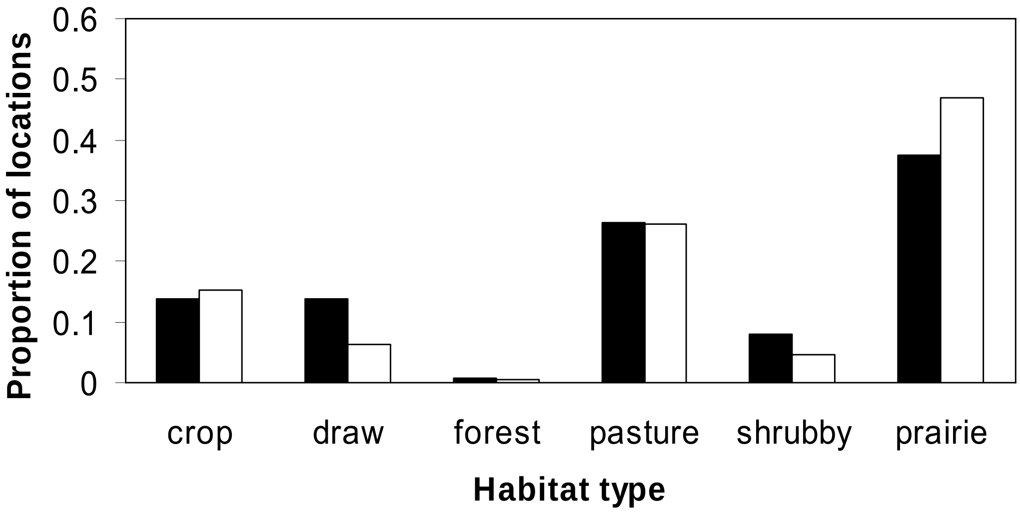

Figure 1. Proportion of telemetry locations in each habitat type at used (black bars; *n* = 1208) and available (white bars, *n* = 6059) points for juvenile dickcissels at Taberville Conservation Area in southwestern Missouri, 2003 to 2004.

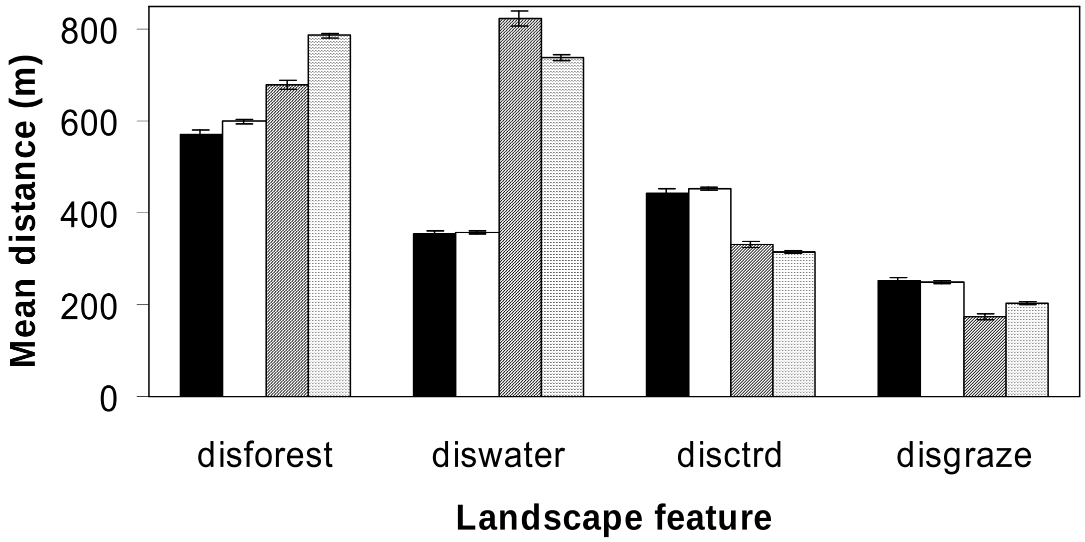

Figure 2. Mean distance (m) to significant landscape features (see Table 2) from population-level resource selection patterns by juvenile dickcissels at two sites in southwestern Missouri, 2003 to 2004. Mean distances at Taberville are shown for used (black bars; *n* = 1208) and available (white bars; *n* = 6059) points. Mean distances at Wah'Kon-Tah are shown for used (bars with upward horizontal lines; *n* = 1619) and available (bars with horizontal dashed lines; *n* = 8109) points. Variable codes are from Appendix 1. All means are shown ± one standard error.

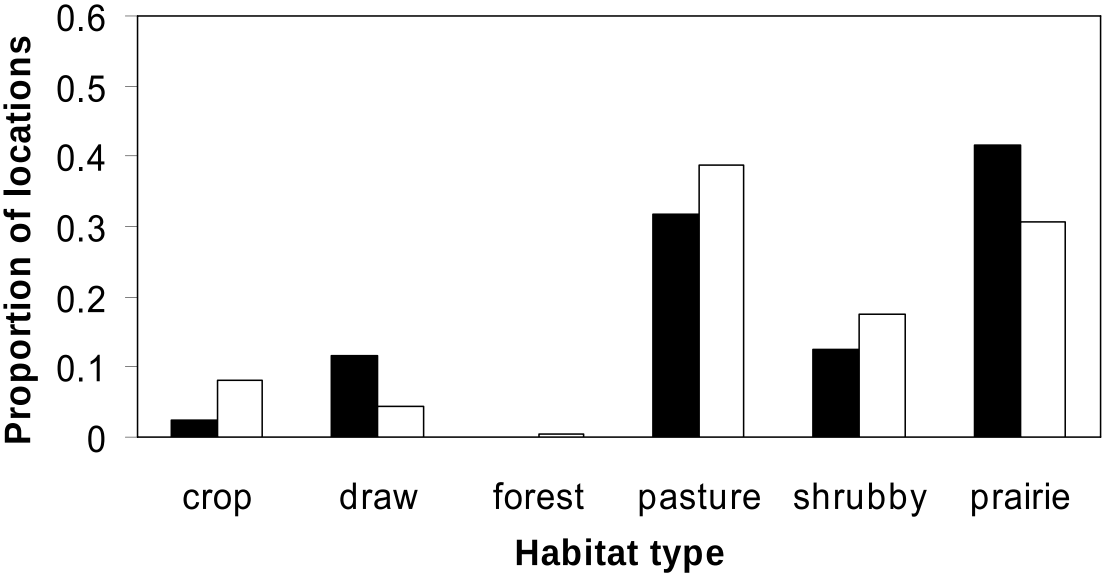

Figure 3. Proportion of telemetry locations in each habitat type at used (black bars; *n* = 1619) and available (white bars; *n* = 8109) points for juvenile dickcissels at Wah'Kon-Tah Prairie in southwestern Missouri, 2003 to 2004.

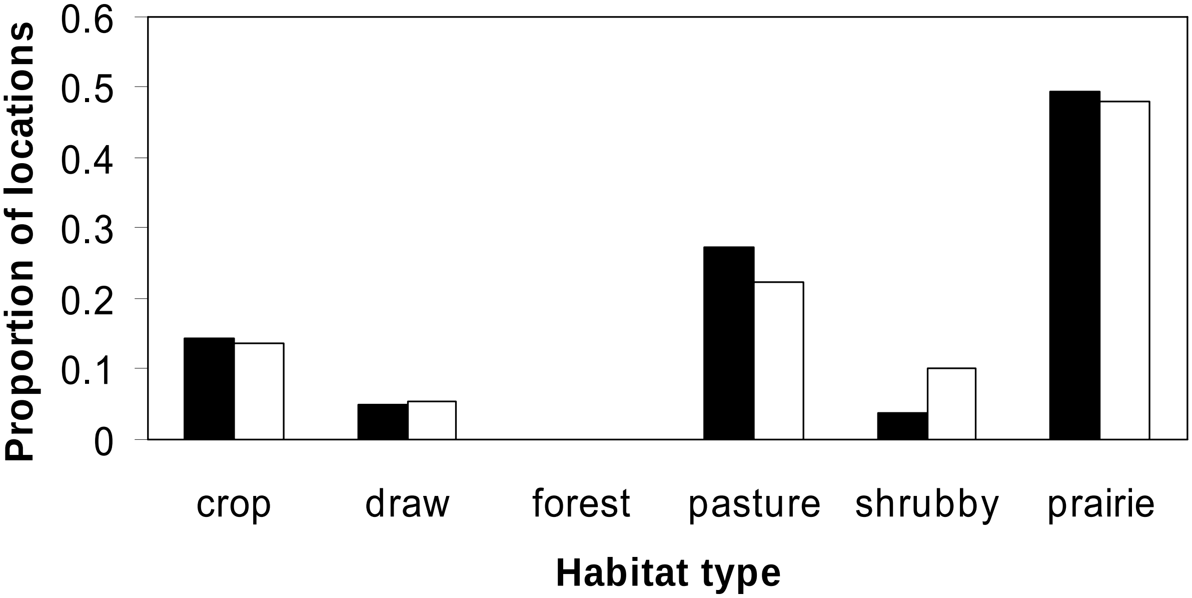

Figure 4. Proportion of telemetry locations in each habitat type at used (black bars; *n* = 1181) and available (white bars; *n* = 5924) points for juvenile eastern meadowlarks at Taberville Conservation Area in southwestern Missouri, 2003 to 2004.

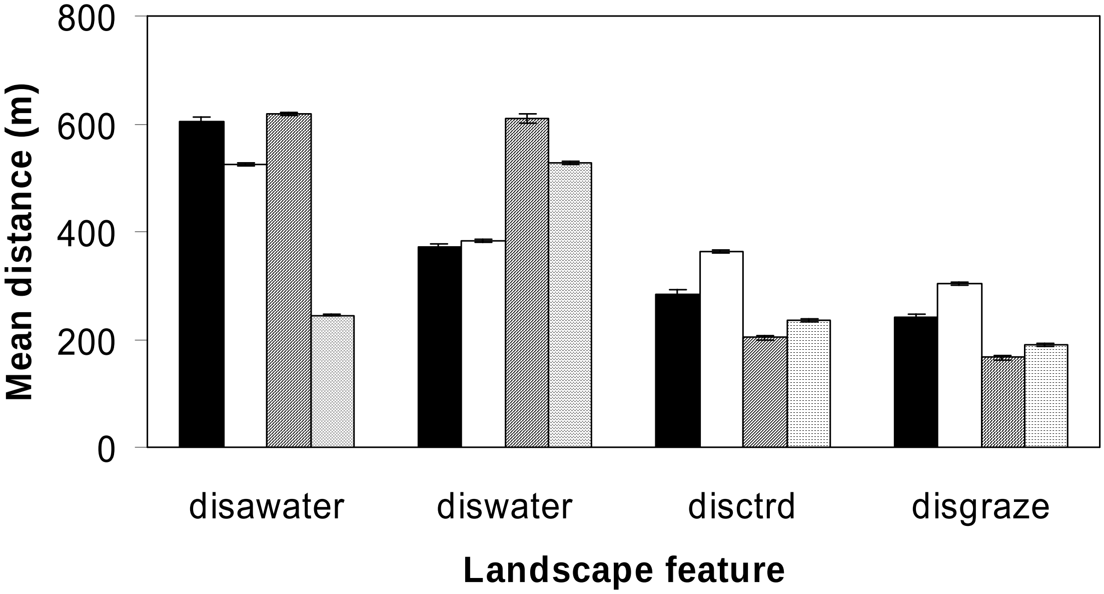

Figure 5. Mean distance (m) to significant landscape features (see Table 5) from population-level resource selection patterns in juvenile eastern meadowlarks at two sites in southwestern Missouri, 2003 to 2004. Mean distances at Taberville are shown for used (black bars; *n* = 1181) and available (white bars; *n* = 5924) points. Mean distances at Wah'Kon-Tah are shown for used (bars with upward horizontal lines; *n* = 2124) and available (bars with dashed horizontal lines; *n* = 10639) points. Variable codes are from Appendix 1. All means are shown ± one standard error.

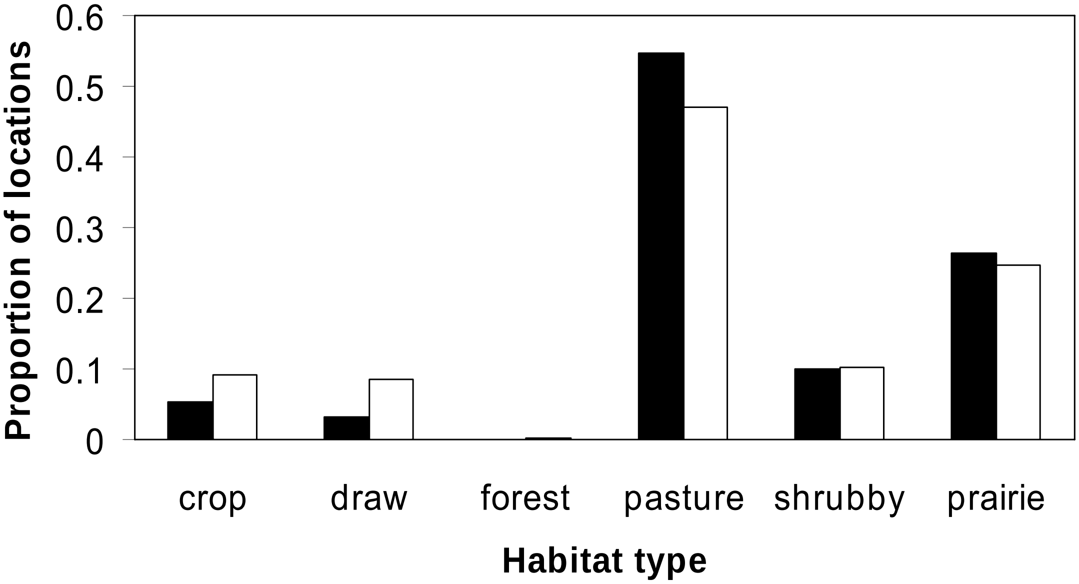

Figure 6. Proportion of telemetry locations in each habitat type at used (black bars; *n* = 2124) and available (white bars; *n* = 10639) points for juvenile eastern meadowlarks at Wah'Kon-Tah Prairie in southwestern Missouri, 2003 to 2004.

Table 1. Availability of each habitat type, soil type, and distance (m) to each landscape feature based on paired, random points associated with used points from telemetry data for juvenile dickcissels and eastern meadowlarks at Taberville Conservation Area and Wah'Kon-Tah Prairie in southwestern Missouri, 2003 – 2004. Variable codes are from Appendix 1. Mean distances are shown ± one standard error.

	Site	
Variable	Taberville	Wah'-Kon'Tah
Crop[a]	1743	1637
Draw[a]	715	1266
Forest[a]	33	72
Pasture[a]	2919	8151
Shrubby prairie[a]	881	2508
Prairie[a]	5700	5119
Alfisol[b]	8910	1822
Inceptisol[b]	------[c]	12
Utilsol[b]	------[c]	4045
Mollisol[b]	3084	12877
PAR	0.009 (0.004)	0.007 (0.009)
Disforest	640.277 (327.882)	710.353 (377.068)
Disdraw	120.572 (98.538)	117.790 (101.431)
Disawater	485.721 (287.593)	249.311 (123.451)
Diswater	369.144 (208.914)	616.691 (449.787)

Table 1. continued.

Variable	Site Taberville	 Wah’-Kon’Tah
Disctrd	408.485 (283.036)	268.960 (229.232)
Dishay	322.483 (310.052)	421.506 (370.369)
Disgraze	---------[d]	195.948 (223.606)
Discrop	335.086 (257.385)	---------[d]
Disub	79.316 (67.707)	100.397 (88.680)

[a] Habitat types coded as categorical variables leaving prairie out as the reference type.

[b] Soil order types coded as categorical variables leaving Mollisols as the reference type because they are the dominant prairie soil (Miller and Donahue 1995).

[c] Not present at Taberville.

[d] Available in small amounts but not used by enough individual birds for inclusion (see Methods text).

Table 2. Odds ratios and 95% confidence intervals from population-level resource selection analyses at the landscape scale for juvenile dickcissels at Taberville Conservation Area and Wah'Kon-Tah Prairie in southwestern Missouri, 2003 – 2004. Variables in bold had 95% confidence intervals that did not include one.

	Site			
	Taberville		Wah'kon-tah	
Variable[a]	Odds Ratio	LCL, UCL	Odds Ratio	LCL, UCL
Crop[b]	1.086×10^{4}	$0.060, 7.297 \times 10^{10}$	41.280	$0.189, 3.189 \times 10^{5}$
Draw[b]	3.744	$5.83 \times 10^{-6}, -2.406 \times 10^{6}$	0.198	$9.330 \times 10^{-6}, -1.787 \times 10^{3}$
Forest[b]	30.000	$8.570 \times 10^{-12}, -4.598 \times 10^{3}$	1.653×10^{5}	$6.030 \times 10^{-6}, 4.120 \times 10^{15}$
Pasture[b]	0.040	$3.110 \times 10^{-7}, 5.369 \times 10^{3}$	1.004	$0.000, 2.614 \times 10^{3}$
Shrubby Prairie[b]	**2.360×10^{-5}**	**$6.380 \times 10^{-8}, 0.009$**	**1.650×10^{-7}**	**$1.090 \times 10^{-12}, 0.022$**
Alfisol[c]	6.956	$0.499, 2.298 \times 10^{2}$	2.950×10^{-5}	$5.940 \times 10^{-8}, 0.014$
Inceptisol[c]	--------------------------------[d]		NA[e]	NA

Table 2. continued.

	Site			
	Taberville		Wah'kon-tah	
Variable	Odds Ratio	LCL, UCL	Odds Ratio	LCL, UCL
Ultisol[c]	--------------------------------- [d]		**5.566 x 10^4**	**80.856, 2.480 x 10^7**
PAR	6.150 x 10^{21}	4.800 x 10^{-156}, 1.300 x 10^3	0.000	0.000, 0.000
Disforest	0.995	0.959, 1.035	**0.650**	**0.408, 0.949**
Disdraw	1.020	0.990, 1.060	1.000	0.988, 1.012
Disawater	0.995	0.977, 1.014	1.020	0.977, 1.077
Diswater	**0.976**	**0.954, 0.996**	1.025	0.998, 1.050
Disctrd	**0.849**	**0.631, 0.999**	0.906	0.740, 1.065
Dishay	0.993	0.975, 1.010	0.546	0.1851, 1.012
Disub	1.001	0.981, 1.023	1.003	0.989, 1.016

Table 2. continued.

	Site			
	Taberville		Wah'kon-tah	
Variable	Odds Ratio	LCL, UCL	Odds Ratio	LCL, UCL
Discrop	0.980	0.947, 1.016	--------------------------------[f]	
Disgraze	------------------------------[f]		**0.956**	**0.915, 0.990**
AE	0.499	0.022, 8.489	2.647	0.214, 28.633
Roads	0.999	0.999, 1.000	0.999	0.999, 1.000
Wooded	0.999	0.999, 1.000	1.000	0.999, 1.000
Water	0.999	0.999, 1.000	1.000	0.999, 1.000

[a] Variables codes are from Appendix 1.

[b] All habitat types are compared to prairie as the reference type.

[c] All soil orders are compared to Mollisols as the reference order.

[d] Dashed lines indicate that variable was not present at that site.

[e] 95% confidence intervals not calculated because $n = 1$.

[f] Available but not used by enough individual birds for inclusion.

Table 3. Population-level resource selection coefficients (β), standard errors (SE), and the number of individuals significantly related to predictors of the presence of juvenile dickcissels at Taberville Conservation Area in southwestern Missouri, 2003 – 2004. Variables where 95% confidence intervals around the odds ratio are shown in bold (from Table 2).

			No. individuals associated with variable	
Variable[a]	Mean β	SE	+	-
Crop[b]	9.29	0.22	5	3
Draw[b]	1.32	0.20	4	2
Forest[b]	-8.91	0.33	1	2
Pasture[b]	-3.21	0.19	3	3
Shrubby Prairie[b]	**-10.65**	**0.12**	**1**	**2**
Alfisol[c]	1.93	0.04	8	4
PAR	50.17	6.71	6	6
Disforest	-4.32×10^{-3}	6.10×10^{-4}	4	7
Disdraw	0.02	5.65×10^{-4}	5	7
Disawater	-4.55×10^{-3}	2.95×10^{-4}	5	7
Diswater	**-0.02**	**3.57×10^{-4}**	**4**	**9**
Disctrd	**-0.16**	**4.36×10^{-3}**	**3**	**8**
Dishay	-6.79×10^{-3}	2.83×10^{-4}	5	8
Disub	1.96×10^{3}	3.32×10^{-4}	6	6

Table 3. continued.

			No. individuals associated with variable	
Variable	Mean β	SE	+	-
Discrop	-0.01	5.37×10^{-4}	5	6
AE	-0.69	0.04	5	5
Roads	-2.30×10^{-5}	-5.18×10^{-7}	4	8
Wooded	-1.60×10^{-5}	4.93×10^{-7}	7	6
Water	-2.00×10^{-5}	6.53×10^{-7}	6	5

[a] Variables codes are from Appendix 1.

[b] All habitat types are compared to prairie as the reference type.

[c] All soil orders are compared to Mollisols as the reference order.

Table 4. Population-level resource selection coefficients (β), standard errors (SE), and the number of individuals significantly related to predictors of the presence of juvenile dickcissels at Wah'Kon-Tah Prairie in southwestern Missouri, 2003 – 2004. Variables where 95% confidence intervals around the odds ratio are shown in bold (from Table 2).

			No. individuals associated with variable	
Variable[a]	Mean β	SE	+	-
Crop[b]	3.72	0.11	2	2
Draw[b]	-1.61	0.14	7	5
Forest[b]	12.01	0.45	1	1
Pasture[b]	0.00	0.14	6	5
Shrubby Prairie[b]	**-15.61**	**0.19**	**1**	**8**
Alfisol[c]	-10.43	0.10	1	4
Inceptisol[c]	17.01	NA[d]	1	0
Ultisol[c]	**10.92**	**0.10**	**5**	**2**
PAR	1.66×10^{3}	35.46	11	5
Disforest	**-0.43**	**6.90 x 10**	**4**	**14**
Disdraw	4.30×10^{-4}	1.92×10^{-4}	8	9
Disawater	0.02	8.36×10^{-4}	9	7
Diswater	0.02	4.20×10^{-4}	13	4
Disctrd	-0.09	2.90×10^{-3}	14	4

Table 4. continued.

			No. individuals associated with variable	
Variable	Mean β	SE	+	-
Dishay	-0.60	0.01	6	11
Disub	3.09×10^{-3}	2.10×10^{-4}	10	7
Disgraze	**-0.04**	**6.42×10^{-3}**	**3**	**7**
AE	0.97	0.03	10	6
Roads	-7.90×10^{-6}	9.39×10^{-7}	7	8
Wooded	1.17×10^{-6}	3.79×10^{-7}	7	9
Water	1.16×10^{-5}	5.91×10^{-7}	9	6

[a] Variables codes are from Appendix 1.

[b] All habitat types are compared to prairie as the reference type.

[c] All soil orders are compared to Mollisols as the reference order.

[d] Indicates *n* = 1.

Table 5. Odds ratios and 95% confidence intervals from population-level resource selection analyses at the landscape scale for juvenile eastern meadowlarks at Taberville Conservation Area and Wah'Kon-Tah Prairie in southwestern Missouri, 2003 – 2004. Variables in bold had 95% confidence intervals that did not include one.

	Site			
	Taberville		Wah'kon-tah	
Variable[a]	Odds Ratio	LCL, UCL	Odds Ratio	LCL, UCL
Crop[b]	**1.699×10^4**	**$24.093, 7.617 \times 10^6$**	3.569	$0.005, 1.085 \times 10^4$
Draw[b]	0.032	$5.730 \times 10^5, 42.161$	0.123	$0.000, 1.934 \times 10^2$
Forest[b]	0.021	0.000, 1.908	313.272	$0.000, 1.640 \times 10^9$
Pasture[b]	0.027	$2.410 \times 10^{-5}, 55.225$	**1.345×10^3**	**$21.045, 1.020 \times 10^5$**
Shrubby Prairie[b]	1.525×10^3	$0.114, 1.560 \times 10^{10}$	0.006	$6.840 \times 10^{-6}, 12.655$
Alfisol[c]	0.985	0.003, 202.551	0.012	$5.140 \times 10^{-5}, 1.750$
Inceptisol[c]	------------------------------[d]		NA[e]	NA

Table 5. continued.

	Site			
	Taberville		Wah'kon-tah	
Variable	Odds Ratio	LCL, UCL	Odds Ratio	LCL, UCL
Ultisol[c]	--------------------------------[d]		14.152	0.008, 2.183×10^{4}
PAR	$5.140 \times 10^{4-77}$	8.500×10^{-218}, 6.370×10^{51}	1.090×10^{12}	4.100×10^{-106}, 2.200×10^{127}
Disforest	1.007	0.970, 1.069	0.989	0.978, 1.002
Disdraw	1.005	0.987, 1.020	1.003	0.989, 1.015
Dispond	**1.046**	**1.012, 1.090**	0.993	0.982, 1.001
Disstream	0.995	0.973, 1.013	**1.186**	**1.032, 1.398**
Disctrd	**0.980**	**0.963, 0.994**	0.993	0.983, 1.002
Dishay	1.008	0.969, 1.056	1.001	0.987, 1.015
Disub	0.990	0.978, 1.001	1.001	0.991, 1.010
Discrop	1.009	0.982, 1.040	--------------------------------[f]	

Table 5. continued.

	Site			
	Taberville		Wah’kon-tah	
Variable	Odds Ratio	LCL, UCL	Odds Ratio	LCL, UCL
Disgraze	-------------------------------[f]		**0.971**	**0.945, 0.991**
AE	0.331	0.056, 1.580	0.743	0.243, 2.194
Roads	**0.999**	**0.999, 0.999**	1.000	0.999, 1.000
Wooded	0.999	0.999, 1.000	1.000	1.000, 1.000
Water	0.999	0.999, 1.000	1.000	1.000, 1.000

[a] Variables codes are from Appendix 1.

[b] All habitat types are compared to prairie as the reference type.

[c] All soil orders are compared to Mollisols as the reference order.

[d] Not present at that site.

[e] 95% confidence intervals not calculated because $n = 1$.

[f] Available but not used by enough individual birds for inclusion.

Table 6. Population-level resource selection coefficients (β), standard errors (SE), and the number of individuals significantly related to predictors of juvenile meadowlarks on the landscape at Taberville Conservation Area in southwestern Missouri, 2003 – 2004. Variables where 95% confidence intervals around the odds ratio are shown in bold (from Table 5).

			No. individuals associated with variable	
Variable[a]	Mean β	SE	+	-
Crop[b]	**9.74**	**0.10**	**5**	**2**
Draw[b]	-3.42	0.11	2	6
Forest[b]	-3.82	0.08	1	2
Pasture[b]	-3.59	0.11	4	5
Shrubby Prairie[b]	7.32	0.20	3	2
Alfisol[c]	-0.01	0.08	5	4
PAR	-1.75×10^{2}	5.00	6	5
Disforest	$7.02\ 10^{-3}$	$8.63\ 10^{-4}$	2	9
Disdraw	$5.34\ 10^{-3}$	2.61×10^{-4}	7	4
Disawater	**0.04**	**$5.91\ 10^{-4}$**	**9**	**2**
Diswater	$-4.57\ 10^{-3}$	3.25×10^{-4}	5	6
Disctrd	**-0.02**	**2.56×10^{-4}**	**2**	**8**
Dishay	8.37×10^{-3}	7.23×10^{-4}	4	6
Disub	$-9.27\ 10^{-3}$	1.94×10^{-4}	4	7

Table 6. continued.

			No. individuals associated with variable	
Variable	Mean β	SE	+	-
Discrop	$9.88\ 10^{-3}$	4.49×10^{-4}	6	3
AE	-1.10	0.02	3	7
Roads	**-5.8×10^{-5}**	**8.36×10^{-7}**	**2**	**8**
Wooded	-7.2×10^{-6}	4.95×10^{-7}	6	5
Water	6.99×10^{-5}	1.23×10^{-6}	8	2

[a] Variables codes are from Appendix 1.

[b] All habitat types are compared to prairie as the reference type.

[c] All soil orders are compared to Mollisols as the reference order.

Table 7. Population-level resource selection coefficients (β), standard errors (SE), and the number of individuals significantly related to predictors of juvenile meadowlarks on the landscape at Wah'Kon-Tah Prairie in southwestern Missouri, 2003 – 2004. Variables where 95% confidence intervals around the odds ratio are shown in bold (from Table 5).

			No. individuals associated with variable	
Variable[a]	Mean β	SE	+	-
Crop[b]	1.27	0.12	3	8
Draw[b]	-2.09	0.11	4	8
Forest[b]	5.74	0.24	3	3
Pasture[b]	**7.20**	**0.06**	**10**	**2**
Shrubby Prairie[b]	-5.08	0.11	3	8
Alfisol[c]	-4.41	0.08	5	5
Inceptisol[c]	-12.66	NA[d]	0	1
Ultisol[c]	2.64	0.12	5	3
PAR	27.71	4.18	8	7
Disforest	-0.01	1.99×10^{-4}	7	8
Disdraw	3.17×10^{-3}	2.04×10^{-4}	9	5
Disawater	-6.77×10^{-3}	1.54×10^{-4}	8	7
Diswater	**0.17**	**2.59×10^{-3}**	**11**	**4**
Disctrd	-6.45×10^{-3}	1.57×10^{-4}	8	7

Table 7. continued.

			No. individuals associated with variable	
Variable	Mean β	SE	+	-
Dishay	1.15×10^{-3}	2.26×10^{-4}	6	9
Disub	1.93×10^{-3}	1.54×10^{-4}	10	5
Disgraze	**-0.02**	**3.93×10^{-4}**	**4**	**8**
AE	-0.29	0.01	8	5
Roads	3.87×10^{-5}	9.78×10^{-7}	8	7
Wooded	2.34×10^{-5}	3.61×10^{-7}	9	3
Water	2.17×10^{-5}	6.23×10^{-7}	10	5

[a] Variables codes are from Appendix 1.

[b] All habitat types are compared to prairie as the reference type.

[c] All soil orders are compared to Mollisols as the reference order.

[d] Indicates $n = 1$.

Appendix 1. Variable codes and descriptions for landscape resource selection analysis with juvenile dickcissels and eastern meadowlarks in southwestern Missouri, 2003 – 2004.

Variable Code	Variable description
Crop	Lands in production including soybeans, corn, and wheat
Draw	Narrow woody corridors originating around stream beds
Forest	Mature forests habitat types classified from land cover data (MORAP 1983) and modified to include current extent
Pasture	Habitat types grazed by cattle or goats (only at Wah'kon-tah)
Shrubby prairie	Habitat types dominated by shrubs and rank vegetation generally ≥ 1 m that have not been as a whole unit for at least two years (includes isolated woody reduction)
Prairie	Grassland habitat types receiving management practices at least once every three years such that grasses are the dominant functional group and woody species are a minor component with spare and isolated distributions
Alifsols[a]	Soils in the Alfisol order that are fertile, have a clay horizon, and are either moist or dry during the growing season
Inceptisols[a]	Soils in the Inceptisol order that are usually moist, often weathered or altered by human factors, and recently developed
Ultisols[a]	Soils in the Ultisol order are often weathered and have clay accumulation
Mollisols[a]	Soils in the Mollisol order usually associated with prairies
PAR	Perimeter (m) to area (m^2) ratio
Disforest	Distance (m) to the nearest forest block (see definition above)
Disdraw	Distance (m) to the nearest woody draw (see definition above)
Disawater	Distance (m) to artificial water sources including ponds and lakes

Appendix 1. continued.

Variable Code	Variable description
Diswater	Distance (m) to naturally occurring water sources like streams
Disctrd	Distance (m) to county roads
Dishay	Distance (m) to interior hay roads that are composed of dirt or gravel roads used for management purposes and navigation within sites
Disub	Distance (m) to the nearest management unit boundary (on state land) or property boundary (private land)
Discrop[b]	Distance (m) to the nearest crop field (soybean, corn, or wheat)
Disgraze[c]	Distance (m) to the nearest grazed pasture (cattle or goats)
AE	Interaction between area (m^2) of the management unit or private property parcel and distance (m) to the nearest edge of that management unit or private property parcel (see definition of Disub above)
Roads	Interaction between disctrd and dishay (see definitions above)
Wooded	Interaction between disdraw and disforest (see definitions above)
Water	Interaction between disawater and diswater (see definitions above)

[a] According to Miller and Donahue (1995)

[b] Available and used by a sufficient number of individual birds for inclusion.

[c] Available and used by sufficient number of individual birds for inclusion.

CHAPTER 3

MOVEMENT PATTERNS OF POST-FLEDGING GRASSLAND BIRDS IN MISSOURI

ABSTRACT

We quantified dispersal movements and home range size of two species of grassland birds during the post-fledging period in southwestern Missouri, from 2002 to 2004. Understanding movement patterns during the post-fledging period is critical for developing effective conservation strategies for grassland birds. Using radio telemetry, we obtained ≥ 30 locations for 74 juvenile dickcissels (*Spiza americana*) and 64 juvenile eastern meadowlarks (*Sturnella magna*) during the post-fledging period. We calculated dispersal movements and then used fixed-kernel estimators to calculate home range size based on utilization distributions. We used an iterative, exploratory process with Information Theoretic methods to model home range size as a function of biological, spatial, and temporal factors. Dickcissels showed earlier initiation of dispersal movements but a shorter period of large dispersal movements compared to meadowlarks. Core (50%) home range sizes were similar between species, but 95% home range contours were 25% larger for meadowlarks than dickcissels, which is not surprising given the difference in body size (mean juvenile weight 15.3 ± 3.0 g for dickcissels compared to 44.8 ± 6.6 g for meadowlarks). Home range patterns were mostly non-linear and categorized as central or exploratory in contrast to other post-fledging studies. Across years, biological factors were the best predictors of home range size. In general, home range size decreased with increasing clutch size and fledging order in relation to other brood mates for both species. Heavier birds at fledging also were associated with larger

home range sizes for both species. Our results indicate grassland birds show different movement patterns than previously documented for other species. In addition, our results emphasize the need for expanding our definitions and concepts of suitable breeding habitats to accommodate nesting and post-fledging requirements.

INTRODUCTION

The post-fledging period is a crucial life history stage for juvenile birds as they complete the prebasic molt and begin to build fat reserves for migration while trying to avoid predators (Moore 1993, Vega Rivera et al. 1998). Despite the importance of the post-fledging period, it is widely regarded as the least understood part of the avian life cycle (Part 1990, Morton 1991, Baker 1993, King and Belthoff 2001). Although several authors have provided information on post-fledging movement patterns in passerines (Morton et al. 1991, Anders et al. 1998, Vega Rivera et al. 1998, Fink 2003, White 2005), limited information about grassland passerines is available (Kershner 2001, Yackel Adams 2001). However, existing information on wood thrush (*Hylochichla mustelina*) and our two target species, dickcissels (*Spiza americana*) and eastern meadowlarks (*Sturnella magna*), indicates that movements are driven by the proximity and juxtaposition of multiple resources during the post-fledging period (Anders et al. 1998, Fink 2003, Suedkamp Wells 2005, see chapters 1 and 2).

Understanding the movement patterns of juvenile grassland birds is critical for effective conservation and management strategies. Current grassland conservation models, such as the Bird Conservation Area concept proposed by the Midwest Working Group of Partners in Flight (PIF) (Pashley and Fitzgerald 1996), strive to protect large areas in hopes of providing the suite of resources needed by all grassland birds during their life cycle. If movement patterns are driven by the proximity and juxtaposition of required resources, then grassland conservation strategies that maximize the availability of these resources are most likely to be effective. In ecosystems like grasslands that are maintained by disturbance and characterized by high spatial and temporal variability (Knopf 1994, Samson and Knopf 1994), lack of suitable resources during the post-fledging dispersal process may limit population growth by negatively affecting juvenile birds prior to recruitment. Post-fledging movement patterns are most likely driven by the need to avoid predators and locate foraging opportunities (Anders et al. 1998). However, other components of our study also indicate that juvenile bird movements may be affected by adverse microclimates that force young birds to seek thermal refuge (Suedkamp Wells 2005, Chapter 1).

Behavioral interactions, such as dominance patterns among siblings, and differences in movement patterns (stationary or drifting) due to parental care strategies, may affect the optimal size of grassland reserves and the distribution of resources within those reserves required for successful recruitment of juveniles. For example, dominance hierarchies among juvenile western screech owls (*Otus kennicottii*) influenced the timing of dispersal but not the total dispersal distance (Ellsworth and Belthoff 1999). However, juvenile golden eagles (*Aquila chrysaetos*) dispersed together and showed no effects of aggression or dominance on post-fledging dispersal (O'Toole et al. 1999). Differences in post-fledging movements as a consequence of parental care patterns may also be different across ecosystem types. For example, wood thrush display stationary and drifting home ranges in Missouri forests (Anders et al. 1998). Post-fledging movement patterns in wood thrush and Swainson's thrush (*Catharus ustulatus*) also have been categorized as having multiple dispersal sites after leaving the natal site (Vega Rivera et al. 1998, Fink 2003, White 2005). For Swainson's thrushes, White (2005) associated these different movement types with parental care strategies. To our knowledge, the only information available on the post-fledging movements of grassland birds relates to the distances moved during the pre-independence period by lark buntings (*Calamospiza melanocorys*) and total dispersal distances of juvenile eastern meadowlarks (*Sturnella magna*) (Kershner 2001). As a result, researchers need information on movement patterns in grassland birds to determine if they show movement patterns similar to those documented for thrush species. If movement patterns are influenced by parental strategies (stationary or drifting), then differences between grassland and forests birds may necessitate alternative conservation strategies that address the size and distribution of resources on reserves to minimize hazards for juvenile birds during dispersal.

Our goal was to determine the type of movement pattern displayed by two species of grassland birds and to determine the best predictors of home range size. We focused on dickcissels (*Spiza americana*) and eastern meadowlarks because they were common grassland species on our site, had a large enough body size (> 20 g) to facilitate wearing transmitters for a sufficient time period, and have shown declining population trends nationwide between 1966 and 2003 (Sauer et al. 2004). In addition, dickcissels are currently listed on the PIF Continental Watchlist as a species with declines or high threats and in need of management (Rich et al. 2004).

METHODS

Study Sites

We conducted this study at Taberville Conservation Area (38° N, 93° W) and Wah'Kon-Tah Prairie (37°N, 94° W) in Cedar and St. Clair Counties in southwestern Missouri. Taberville Conservation area is a 680-ha prairie owned and managed by the Missouri Department of Conservation (MDC) and embedded in an agricultural matrix of crops (wheat, soybeans, and corn) and other private land uses. Wah'Kon-Tah Prairie is a 1930-ha prairie owned by the Missouri Chapter of The Nature Conservancy (TNC) and jointly managed by MDC and TNC located at the northern periphery of El Dorado Springs, Missouri (population ~ 4,000 people). Both sites are part of a network of focal areas targeting grassland bird conservation in Missouri. Dominant land management practices included livestock grazing, prescribed burning, seed harvesting, and haying. Removal of invasive and encroaching woody species occurs along woody draws, fence lines, and pasture borders. The study sites are divided into management units that receive some type of management practice (primarily prescribed burning or haying) at least once every three years. Dominant vegetation was composed of bluestem grasses and included big bluestem (*Andropogon gerardi*), little bluestem (*Schizachyrium scoparium*), and indian grass (*Sorghastrum nutans*). Forb species included coneflowers (*Echinacea* spp.), white wild indigo (*Baptisia alba*), blazing star (*Liatrus* spp.), compass plant (*Silphium laciniatum*), milkweeds (*Aesclepias* spp), and sunflowers (*Helianthus* spp.). Dominant, native woody species on these sites include smooth sumac (*Rhus glabra*), persimmon (*Diospyros virginiana*), blackberry (*Rubus* spp.), and roses (*Rosa* spp.). Throughout this chapter unless specifically stated otherwise, we will refer to woody plants as these small, shrub species that were native and not planted for human uses.

Bird Capture and Handling

We located nests of both species using systematic searches and haphazard walks from 0600 to 1400 hours each day between the third week of April and the second week of August from 2002 to 2004. When possible, we used behavioral cues of the parents to indicate the presence of a nest nearby. After locating each nest, we recorded the GPS coordinates and marked the location by placing colored flagging tape at least 5 m away. At each nest we recorded the species, content, parental activity, and presence of any non-host eggs. If the nest contained nestlings, we attempted to age the nestlings using the presence of down, whether the eyes were open or not, the extent of pin feather development, or the presence of a full complement of feathers (K. Suedkamp Wells, pers. obs.). Based on our observations, we were usually successful at aging nestlings within two days of their true age depending on growth rates and weather conditions. We monitored each nest every three to four days until just prior to fledging and then switched to daily nest checks. Two to three days prior to fledging, we attached a metal USFWS band to the left leg and a unique combination of plastic, UV-resistant Darvic bands (Avinet, Dryden, New York) to the right leg and weighed each individual.

We began processing each bird by weighing it to the nearest gram using a spring scale (Avinet, Dryden, New York). Following a modification of the Rappole and Tipton method (1991) previously evaluated (Suedkamp Wells et al. 2003), we attached 0.7-gram transmitters with a 10-cm whip antennae (Biotrack, Dorset, United Kingdom) to the back of each bird using a leg harness. Battery life for each transmitter was expected to range between 55 and 60 days. We constructed the leg harness from cotton, elastic beading cord to allow room for growth. Using super glue (Duro, Avon, Ohio), we secured the bottom of the transmitter to the back of the bird. After attaching transmitters to each bird, we placed the brood back in the nest. Handling and processing time usually was between 2 and 5 minutes per bird.

Radiotracking

Using telemetry, we began tracking birds using homing for visual confirmation (Mech 1983) the day after attaching transmitters. If the brood remained in the nest the day after attachment, we returned each subsequent morning and began tracking when at least one brood member fledged. We tracked each bird twice daily in non-consecutive time blocks for a minimum of 50 detections per individual for a total of 25 individuals of both species (Garton et al. 2001). The four tracking blocks were early morning (0600 to 0930 hours), mid-morning (0930 to 1230 hours), afternoon (1230 to 1700 hours), and evening (1700 to 2130 hours). We grouped the time blocks to reflect biological activity and environmental constraints such as hot temperatures when activity is reduced. The first day of tracking for an individual occurred in the early morning and afternoon periods. On the second day, tracking occurred in the mid-morning and evening hours. We continued to alternate tracking days on this schedule to capture locations representative of all diurnal activities typical of tracking studies (Garton et al. 2001). We avoided tracking before 0600 hours and after 2130 hours to reduce the risk of stepping on juvenile birds when they could not be visually located. After reaching 50 detections, each individual was tracked once daily alternating between the first two and last two periods of the day until the bird died, the transmitter was recovered, or the study period ended.

We tracked each individual until we recovered the transmitter or dead bird through the end of August in each field season. Immediately after being unable to locate an individual, we performed extensive searches of the immediate area on foot with a team of assistants. If we were unable to locate the individual, we broadened the search to include all roads within 3.2 km of the last known location using an omni antennae mounted on the roof of a truck. We continued to search for missing individuals twice daily for one week after their disappearance. At the start of the second week, we reduced search time to one attempt per day. We also attempted to locate missing birds by flying at least 5 km strips over the study area in a helicopter twice monthly between 1 June and 30 August each year. Transmitter failure was minimal and < 2% of all transmitters attached showed signs of failure.

Movement and Home Range Calculations

We restricted our sample to those individuals with ≥ 30 detections because simulation research has indicated that is the minimum sample size required for stable home range estimates using kernel estimators (Seaman et al. 1999). We calculated average daily movement from GPS coordinates for each individual using the Animal Movements extension in Arc View 3.3 (Environmental Systems Research Institute, Redlands, California). We calculated 50% and 95% home range contours using a fixed-kernel estimator (Worton 1987, 1989) in Matlab (version 5.3, Mathworks 1999). We used kernel estimators because they have been ranked as the best method for estimating home range size and are capable of calculating utilization distributions (Kernohan et al.

2001). In a review of home range estimators, kernels were superior to several other methods based on several criteria including robustness with autocorrelated data, ability to handle multiple centers of use, comparability with other estimators, and sensitivity to outliers (Kernohan et al. 2001). To determine the smoothing method for the fixed-kernel estimator, we used the "plug in" method and smoothed the x and y coordinates independently (Wand and Jones 1995, Jones et al. 1996). We started the smoothing process using a pilot bandwidth near zero and then iteratively scaled up until the mean square error was minimized for each bird. Finally, we used a grid cell size of 400 x 400 m for smoothing and set the evaluation boundaries at the minimum and maximum for each coordinate ± 5 times the bandwidth for each coordinate.

Modeling Home Range Size

We used an iterative, exploratory approach to predict 95% home range size as a function of several biological, temporal, and spatial variables we selected prior to data collection. Biological variables included weight at the time of transmitter attachment (g), maximum observed clutch size, the number of siblings that fledged, and the order of fledging (1, 2, or 3). We used variables for year and Julian date to represent temporal variables and site as a spatial variable. Prior to analysis, but after data collection, we developed several candidate models to predict home range size based on the variables we selected before data collection. Candidate models composed of single variables and two-variable interactions represented temporal and spatial patterns, site quality, parental quality, food demands, thresholds of weight and number of siblings, and combinations of these factors. Prior to model fitting, we checked for multicollinearity in SAS (SAS Institute 2001) using the tolerance option in PROC REG (Allison 1999). We used likelihood ratio tests to assess goodness-of-fit and the Pearson correlation coefficient to assess predictive power (Littell et al. 1996). We began by fitting the three-year pooled data set for each species with mixed models using PROC MIXED (SAS Institute 1997) in SAS. We included multiple members per brood, so we treated broods as random effects to correctly model the correlation between brood members and the resulting standard errors (Littell et al. 1996, 1998). We used AICc to rank the models because our sample size divided by the number of parameters was < 40 (Burnham and Anderson 1998). Then, we calculated the weight of evidence (w_i) for each model following Burnham and Anderson (2002). If there was model uncertainty ($w_i < 0.90$), we used model averaging over all models in the candidate set to calculate parameter coefficients and standard errors (Burnham and Anderson 2002). If model fit or predictive power were poor, we used summary statistics to evaluate potential causes. For example, we evaluated home range size in different years to see if average trends were similar. If average home range size was highly variable across years, we partitioned the data set into similar years (based on the mean and variability) and repeated the process to improve model fit and predictive power. If model fit and predictive power were adequate, we present the parameter coefficients and standard errors from the best model or model-averaged estimates. If model fit was adequate and we were unable to improve predictive power, we only present relative importance values for each variable.

RESULTS

We attached transmitters to 248 juvenile dickcissels and 164 meadowlarks between 2002 and 2004 representing 94 and 46 broods, respectively. Of the subset that survived, we obtained a minimum of 30 detections on 74 individual dickcissels and 64 individual eastern meadowlarks, representing 45 and 32 broods, respectively. Mean weight at the time of transmitter attachment was 15.3 ± 3.0 g for dickcissels and 44.8 ± 6.6 g for meadowlarks. Mean number of telemetry locations per individual was 53 ± 1 (range 34 – 75) for dickcissels and 58 ± 1 (range 30 – 88) for meadowlarks.

Although both of our target species share some ecological characteristics, they displayed differences in basic dispersal biology. For example, juvenile dickcissels were highly mobile immediately after leaving the nest. We frequently observed juvenile dickcissels hopping out of their nests and making short jumps into small shrubs or grass clumps during the first week post-fledge. When located using homing, juvenile dickcissels also were usually perched at the top of grass clumps or small shrubs waiting for food. In contrast, juvenile meadowlarks remained relatively motionless and often concealed themselves in thick clumps of grass or litter. We never observed juvenile meadowlarks jumping or hopping to navigate or attempting short flights (< 10 m) prior to the second week post-fledge. Another difference between our two focal species was the timing and pattern of departure from the nest at fledging. Juvenile dickcissels most frequently fledged within 24 hours of each other and often departed in pairs or whole groups (K. Suedkamp Wells, pers. obs.). In contrast, juvenile meadowlarks often staggered their departure from the nest as much as one to three days and usually radiated out independently in a star-shaped pattern. In addition, individuals from larger clutches appeared more similar in weight and condition among dickcissels; whereas, meadowlark juveniles were often drastically different in condition with the heaviest juvenile weighing 5 g or more (8 to 15% of total body weight) than the lightest juvenile.

Average distance moved per day increased slowly during the first 29 days post-fledging and was similar between species (Fig. 1). The period of peak dispersal movements began earlier and was shorter for dickcissels compared to eastern meadowlarks. The peak period of dispersal movements started at 33 days post-fledge and appeared to level off around 45 days post-fledge for dickcissels. In comparison, peak dispersal movements for eastern meadowlarks started at 34 days post-fledge and appeared to continue up to at least 60 days post-fledge. Average distance moved per week appeared similar for both species during the first seven weeks post-fledge but variability of dickcissel movements seemed larger after five weeks post-fledging compared to meadowlarks (Fig. 2).

Home range sizes were similar across species for the 50% contour but 95% contours were nearly twice as large for juvenile meadowlarks compared to juvenile dickcissels (Fig. 3). Mean 50% contour home range sizes were 3.9 ± 0.5 ha for dickcissels and 4.6 ± 0.7 ha for meadowlarks. Mean 95% contour home range sizes were 51.2 ± 8.8 ha for dickcissels and 80.9 ± 13.9 ha for meadowlarks. The distribution of 95% home ranges for dickcissels was mostly concentrated on the left side of the distribution in categories < 40 ha and fell off gradually with one exception of a peak in the right tail in the 60 to 80 ha category (Fig. 4). In contrast, 95% home range sizes for meadowlarks were more unevenly distributed with a peak in the 160+ ha category (Fig. 4). In addition, both species displayed central and exploratory home range patterns (Figs. 5, 6). Central home ranges, such as the example shown for a dickcissel in the lower left corner of Fig. 5, were characterized by the clustering of points around the central, natal area with occasional trips away from the central area in a non-linear fashion. Birds showing central home range patterns may have shown long trips (> 500 m) away from the central area, but usually returned to the central area instead of continually dispersing away in sequential movements. In contrast, birds showing exploratory home ranges, such as the example shown by the dickcissel in the upper right corner of Fig. 5, were characterized by numerous long movements (> 500 m) that either resulted in irregular dispersal paths that sequentially moved away from the central area or immediately preceded dispersal from the site.

The final model predicting home range size in dickcissels showed adequate fit and predictive ability. Pooling all three years of data within species showed no evidence of multicollinearity for dickcissels (tolerance > 0.55) or eastern meadowlarks (tolerance > 0.80). Models predicting home range size for dickcissels showed adequate fit (LRT χ^2 = 235, $P < 0.001$) but low predictive power ($r^2 = 0.12$). However, average home range size for dickcissels in 2002 (76.9 ± 22.1 ha, range 25.9 to 127.9 ha) was twice as large and four times as variable compared to 2003 (31.3 ± 7.4 ha, range 13.7 to 48.9 ha) and 2004 (34.9 ± 7.4 ha, range 17.8 to 51.9 ha). Therefore, we partitioned the data, tested for multicollinearity (all tolerances > 0.70), and analyzed 2002 and 2003 plus 2004 separately, which improved predictive power (2002, $\chi^2 = 196$, $P < 0.001$, $r^2 = 0.34$; 2003 plus 2004, $\chi^2 = 170$, $P < 0.001$, $r^2 = 0.13$).

In 2002, the site quality candidate model containing site, clutch size, weight, and number of siblings at fledging was the best model ($w_i = 0.93$) for dickcissels (Table 1). Increasing clutch size was associated with smaller home ranges and increasing weight at fledging was associated with larger home range sizes (Table 2). For the combination of 2003 and 2004, there was model uncertainty ($w_i < 0.60$, Table 3), so we used model averaging to generate parameter coefficients and standard errors. Increasing clutch size and order of fledging were associated with smaller home ranges in addition to interactions between clutch size and number of siblings and order of fledging and weight (Table 4). Average trends across all three years show that dickcissels that were heavier at fledging were associated with larger home range sizes (Fig. 7), but dickcissels from nests with larger clutch sizes (Fig. 8) or that fledged later compared to brood mates (Fig. 9) were associated with smaller home ranges.

Models predicting home range size of eastern meadowlarks fit adequately but we were rarely able to improve predictive ability. For all three years of data pooled, model fit was adequate ($\chi^2 = 243$, $P < 0.001$) but predictive power was poor ($r^2 = 0.03$). Our sample size for 2002 was small (8 individuals), so we removed that year from further analysis. Mean home range sizes for meadowlarks in 2003 (82.7 ± 29.4 ha) were similar to 2004 (70.7 ± 11.6 ha) but 2003 was twice as variable as 2004. Therefore, we partitioned the data and analyzed 2003 and 2004 separately.

There was evidence of multicollinearity in 2003 (tolerance < 0.30), but removing Julian date from further analysis in this year only eliminated the problem (tolerance > 0.80). In 2003, models predicting home range size for meadowlarks showed adequate fit ($\chi^2 = 181$, $P < 0.001$) and improved predictive power ($r^2 = 0.12$). The main effects model including site, clutch size, weight, and number of siblings was the best model ($w_i = 0.99$; Table 5). Weight at fledging and order of fledging were the most important variables as indicated by relative importance values (Table 8).

In 2004, there was no evidence of multicollinearity (tolerance > 0.62) and models predicting home range size showed adequate fit ($\chi^2 = 166$, $P < 0.001$) and improved predictive power ($r^2 = 0.10$). The main effects model was the best model ($w_i = 0.97$; Table 7) and the order of fledgling, number of siblings, and weight were the most important variables (Table 8). Average trends across all three years show that heavier birds (Fig. 7) and birds that fledge earlier compared to their brood mates (Fig. 8) were associated with larger home range sizes and birds that came from nests with larger clutch sizes had smaller home ranges (Fig. 9).

DISCUSSION

Dispersal Distances

Daily movement distances shown by both of our study species fall in the middle of the range reported for forest and scrub species (Morton et al. 1991, Cohen and Lindell 2004, White 2005) and grassland birds (Yackel Adams et al. 2001). For example, mean distances traveled per day at one week post-fledging ranged from 83.1 m (dickcissels) to 85.4 m (meadowlarks). Our estimate is less than half the average distance moved per day reported for lark buntings (256 m) in shortgrass prairies of Colorado (Yackel Adams et al. 2001). However, our estimate for this period is much closer to the estimate for white-throated robins (*Turdus assimilis*) in agricultural landscapes in Costa Rica (35 to 50 m) and white-crowned sparrows (*Zonotrichia leucophrys*) in montane meadows (25 to 30 m). By three weeks post-fledge, our estimates (137 m for dickcissels, 157 m for meadowlarks) were close to those reported for white-crowned sparrows (range 110 to 150 m) (Morton et al. 1991), but still much smaller than those reported for lark buntings (238 m) (Yackel Adams et al. 2001). We believe the lack of similar movement patterns between grassland species is a result of different landscape matrices and the resulting effect on food availability. Yackel Adams et al. (2001) suggested that brood partitioning in lark buntings was due to food limitations and noted that the nearest agricultural field was at least 2 km away. In our study (see Chapter 2) and that of Kershner (2001), juvenile grassland birds have been associated with agricultural fields as a food source. If food was limiting for juvenile lark buntings, this may explain why the movement patterns reported by Yackel Adams et al. (2001) were much larger than the estimates for both grassland species in this study.

Home Range Size

Patterns of space use among juvenile dickcissels and eastern meadowlarks are different than those reported for wood thrush (Anders et al. 1998) and Swainson's thrush (White 2005). To date, the dominant pattern of post-fledging movement that has been described for thrushes characterizes home ranges as either stationary or drifting (Anders et al. 1998, White 2005). Anders et al. (1998) described stationary home ranges as having locations within the home range visited repeatedly up to dispersal and drifting as home ranges where individuals moved gradually away from the natal area. For both thrush species, telemetry locations clearly indicate post-fledging dispersal occurs in a series of linear movements away from the natal home range to one or more post-dispersal areas. In contrast, few individuals of either species we studied displayed this pattern. Instead, juvenile dickcissels and meadowlarks often engaged in exploratory movements away from a central area that they often returned to prior to making additional exploratory forays. Although some individuals showed small clusters of locations away from natal centers, they rarely moved sequentially from those clusters to other additional clusters or dispersal areas in a linear fashion. As a result, the convex polygon home ranges reported by Anders et al. (1998) and the fixed-kernel estimates reported by White

(2005) are most similar to the smallest home ranges we report for each grassland species (see Fig. 4).

We categorize the two types of home ranges shown in this study from fledging to the end of the study period as either central or exploratory in contrast to stationary and drifting (Anders et al. 1998). Birds displaying central home range patterns, such as the dickcissel in the lower left hand corner of Fig. 5 and the meadowlark in the lower left hand of Fig. 6 with dashed lines, were primarily located in the central natal area and showed a few long distance movements (> 500 m), but generally either returned to the central area or dispersed from the area after movement away from the central natal core. In contrast, exploratory home ranges, such as those shown by the dickcissel in the upper right corner of Fig. 5 and the meadowlark in the upper left corner of Fig. 6., generally showed repeated exploratory movements that exceeded 500 m in a non-linear fashion and did not show clustering or evidence of multiple dispersal areas as typically defined (Anders et al. 1998, Vega Rivera et al. 1998, Fink 2003, White 2005).

Predictors of Home Range Size

Biological covariates were the most important factors affecting home range size. In 2002, the combination of biological and temporal factors in the site quality model for dickcissels indicates spatial variability may be important. In a companion study on our site, we also have observed spatial variability in nest success such that certain locations appear to be hot spots for reproduction in this species (K. Suedkamp Wells, unpubl. data). However, biological factors such as weight at fledging and order of fledging were consistently important across years for both species. Our results showing the importance of weight at fledging are consistent with the hypothesis that foraging optimization drives post-fledging movements (Vega Rivera et al. 1998). Juvenile wood thrush in Virginia and Missouri forests were frequently observed moving into early successional habitats where fruit and invertebrate abundance were assumed to be higher compared to mature forest (Vega Rivera et al. 1998, Anders et al. 1998).

Weight at fledging also has been positively associated with survival during the post-fledging period (Krementz et al. 1989, Sullivan 1989, Naef-Daenzer et al. 2001). In addition to longer survival, heavier birds may be more likely to utilize a variety of habitat types during the post-fledging dispersal process. If sampling different habitats is advantageous, then the order of fledging may also help explain patterns in home range size because heavier birds are able to disperse more rapidly and survive adverse weather conditions. In juvenile western screech owls, social dominance affected the timing of dispersal but not the total dispersal distance (Ellsworth and Belthoff 1999). However, we never observed aggressive encounters between siblings, which is consistent with other passerine studies (Anders et al. 1998, Vega Rivera et al. 1998). However, we did observe stark differences in weight and body condition of juvenile meadowlarks that may reflect energetic demands on parents during incubation or competition for food among siblings in nests from larger clutches. As an example, we occasionally observed parents favoring one sibling over the others during feeding in broods with three or more individuals in both species. Location and begging calls sounded similar in these cases, so social dominance may be a factor affecting feeding rates in larger broods. If social behavior affects the order of dispersal in passerines, then birds that fledge earlier may be more likely to locate food resources or other important habitat components during the dispersal process.

CONSERVATION IMPLICATIONS

Our results concur with recommendations from other studies that demonstrate our working definitions and concepts of suitable breeding habitat need modification (Anders et al. 1998, Vega Rivera et al. 1998). Research on the post-fledging needs of forest and grassland birds all indicate that multiple resources (for foraging and avoiding predation) and different habitat types are important (Anders et al. 1998, Vega Rivera et al. 1998, Kershner 2001, Fink 2003, this study, chapter 2). As a result, the spatial and temporal application of traditional management practices needs to be re-evaluated in this context. In grassland ecosystems, we suggest management paradigms related to agricultural habitats and woody species need particular attention because of their association with resource use (see Chapters 1 and 2). If food sources are limiting, juvenile birds may be forced to increase their home range size to accommodate those needs. Previous research has demonstrated that agricultural habitats were important food sources (Kershner 2001, McKee 2003), contrary to prevailing dogma in grassland bird conservation models in the Midwest (Pashley and Fitzgerald 1996). Results from other parts of this study (see Chapters 1 and 2) indicate that resource use is affected by the availability of food resources such as crops and cover resources including woody habitat features (small shrubs and draws). Therefore, we suggest that conservation priority should not necessarily penalize potential sites with agricultural matrices.

In addition, certain woody features (small shrubs in Chapter 1; draws in Chapter 2) that have been associated with negative effects on reproductive success (Johnson and Temple 1990, Winter et al. 2000), clearly play a positive role for post-fledging grassland birds (Chapters 1 and 2). If juvenile birds are forced to expand their home ranges to find habitat components to provide shelter from predators or avoid adverse microclimates (using woody features as shelter), then lack of, or inadequate, management may make larger reserves necessary to accommodate post-fledging requirements. We suggest that future research on post-fledging grassland birds should focus on evaluating the utility of agricultural and woody features in other landscape contexts to determine whether the patterns we have documented reflect true selection or the best available options on our sites. In either situation, we need to alter our conservation message so it reflects the total suite of needs required by grassland birds during the breeding season.

Another implication of our results is that we need additional information on other bird species and groups to elucidate post-fledging requirements for designing appropriate conservation strategies. In contrast to previously reported work, primarily on thrush species (Anders et al. 1998, Vega Rivera et al. 1998, Fink 2003, Cohen and Lindell 2004, White 2005), grassland birds show different patterns of dispersal movements. In addition, patterns within grassland species across grassland types (shortgrass, Yackel Adams et al. 2001; tallgrass, Kershner 2001, this study,) appear different and may not be limited by the same factors. If landscape composition and structure impose barriers or risks for dispersing birds, conservation and management strategies at local scales may not be as effective. As a result, additional research is needed on other species during the post-fledging period to evaluate potential differences as a result of varying landscapes. In particular, we suggest that post-fledging research should be expanded to include birds with declining populations and restricted distributions such as shrubland, riparian, and wetland bird species. These same groups also contain the highest number of species of conservation concern listed on the Continental Watchlist by Partners in Flight (Rich et al. 2004).

LITERATURE CITED

Anders, A. D., J. Faaborg, and F. R. Thompson III. 1998. Postfledging dispersal, habitat use, and home-range size of juvenile wood thrushes. Auk 115:349-358.

Baker, R. R. 1993. The function of post-fledging exploration: a pilot study of three species of passerines ringed in Britain. Ornis Scandinavica 24:71-79.

Burnham, K. P. and D. R. Anderson. 1998. Model selection and inference. A practical Information-Theoretic Approach. Springer, New York, New York, USA.

_____, and _____. 2002. Model selection and multi-model inference. A practical Information-Theoretic Approach. Second edition. Springer, Norwell, Massachusetts, USA.

Cohen, E. B., and C. A. Lindell. 2004. Survival, habitat use, and movements of fledgling white-throated robins (*Turdus assimilis*) in a Costa Rican agricultural landscape. Auk 121:404-414.

Ellsworth, E. A., and J. R. Belthoff. 1999. Effects of social status on the dispersal behavior of juvenile western screech-owls. Animal Behavior 57:883-892.

Fink, M. L. 2003. Post-fledging ecology of juvenile Wood thrush in fragmented and contiguous landscapes. Ph.D. Dissertation, University of Missouri, Columbia, Missouri, USA.

Garton, E. O., M. J. Wisdom, F. A. Leban, and B. K. Johnson. 2001. Experimental design for radiotelemetry studies. Pages 16–44 in J. J. Millspaugh and J. M. Marzluff, editors. Radio tracking and animal populations. Academic Press, Inc. San Diego, California, USA.

Johnson, H. G., and S. A. Temple. 1990. Nest predation and brood parasitism of tallgrass prairie birds. Journal of Wildlife Management 54:106-111.

Jones, M. C., J. S. Marron, S. J. Sheather. 1996. A brief survey of bandwidth selection for density estimation. Journal of the American Statistical Association 91:401-407.

Kernohan, B. J., R. A. Gitzen, and J. J. Millspaugh. 2001. Analysis of animal space use and movements. Pages 126-168 in J. J. Millspaugh and J. M. Marzluff, editors. Wildlife Radiotelemetry: Design and Analysis. Academic Press, Inc. San Diego, California, USA.

Kershner, E. L. 2001. Conservation of grassland birds in an agricultural landscape: the importance of habitat availability and demography. Ph.D. Dissertation, University of Illinois at Urbana-Champaign., Urbana, Illinois, USA.

King, R. A. and J. R. Belthoff. 2001. Post-fledging dispersal of burrowing owls in southwestern Idaho: characterization of movements and use of satellite burrows. Condor 103:118-126.

Knopf, F. L. 1994. Avian assemblages on altered grasslands. Studies in Avian Biology 15:247-257.

Krementz, D. G., J. D. Nichols, J. E. Hines. 1989. Postfledging survival of European starlings. Ecology 70:646-655.

Littell, R. C., G. A. Milliken, W. W. Stroup, and R. D. Wolfinger. 1996. SAS system for mixed models. SAS Institute, Cary, North Carolina, USA.

_____P. R. Henry, and C. B. Ammerman. 1998. Statistical analysis of repeated measures data using SAS procedures. Journal of Animal Science 76:1216-1231.

Mathworks. 1999. Matlab version 5.3. Natick, Massachusetts, USA.

Mech, L. D. 1983. Handbook of animal radio-tracking. University of Minnesota Press, Minneapolis, Minnesota, USA.

Moore, F. R., S. A. Gauthreaux, Jr., P. Kerlinger, and T. R. Simmons. 1993. Stopover habitat: management implications and guidelines. Pages 58-69 *in* Status and management of Neotropical Migratory birds (D. M. Finch and P. W. Stangel, Eds.). US. Forest Service General Technical Report RM-229. Fort Collins, Colorado, USA.

Morton, M. L., M. W. Wakamatsu, M.E. Pereyra, and G. A Morton. 1991. Postfledging dispersal, habitat imprinting, and philopatry in a montane, migratory sparrow. Ornis Scandinavica 22:98-106.

Naef-Daenzer, B., F. Widmer, and M. Nuber. 2001. Differential post-fledging survival of great and coal tits in relation to their condition and fledging date. Journal of Animal Ecology 70:730-738.

O'Toole, L. T., P. L. Kennedy, R. L. Knight, and L. C. McEwen. 1999. Postfledging behavior of golden eagles. Wilson Bulletin 11:472-477.

Part, T. 1990. Natal dispersal in the collared flycatcher: possible causes and reproductive consequences. Ornis Scandinavica 21:83-88.

Pashley, D. and J. Fitzgerald. 1996. Results of Prairie Pothole Joint Venture/Partners in Flight meeting, July 11 – 12. Unpublished memorandum.

Rappole, J. H., and A. R. Tipton. 1991. New harness design for attachment of radio transmitters to small passerines. Journal of Field Ornithology 62:335-337.

Rich, T. D., C. J. Beardmore, H. Berlanga, P. J. Blancher, M. S. W. Bradstreet, G. S. Butcher, D. W. Demarest, E. H. Dunn, W. C. Hunter, E. E. Inigo-Elias, J. A. Kennedy, A. M. Martell, A. O. Panjabi, D. N. Pashley, K. V. Rosenberg, C. W. Rustay, J. S. Wendt, and T. C. Will. 2004. Partners in Flight North American Landbird Conservation Plan. Cornell Lab or Ornithology, Ithaca, New York, USA.

Samson, F., and F. Knopf. 1994. Prairie conservation. BioScience 44:418-421.

SAS Institute. 1997. SAS/STAT software: changes and enhancements through release 6.12. SAS Institute, Cary, North Carolina, USA.

SAS Institute. 2001. SAS, version 8.0. SAS Institute, Cary, North Carolina, USA.

Sauer, J. R., J. E. Hines, and J. Fallon. 2004. The North American Breeding Bird Survey, Results and Analysis 1966 - 2003. Version 2004.1. USGS Patuxent Wildlife Research Center, Laurel, Maryland, USA.

Seaman, D. E., J. J. Millspaugh, B. J. Kernohan, G. C. Brundige, K. J. Raedeke, and R. A. Gitzen. 1999. Effects of sample size on kernel home range estimates. Journal of Wildlife Management 63:739-747.

Suedkamp Wells, K. M., B. E. Washburn, J. J. Millspaugh, M. R. Ryan, and M. W. Hubbard. 2003. Effects of radio-transmitters on fecal glucocorticoid levels in captive dickcissels. Condor 105:805-810.

Sullivan, K. A. 1989. Predation and starvation: age-specific mortality in juvenile juncos (*Junco phasenotus*). Journal of Animal Ecology 58:275-286.

Vega Rivera, J. H., J.H. Rappole, W. J. McShea, and C. A. Haas. 1998. Wood thrush postfledging movements and habitat use in northern Virginia. Condor 100:69-78.

Wand, M. P., and M. C. Jones. 1995. Kernel smoothing. Chapman and Hall, London, United Kingdom.

White, J. D. 2005. Post-fledging survival, resource selection, and dispersal of juvenile Swainson's thrushes in central coastal California. Unpublished PhD Dissertation, University of Missouri, Columbia, Missouri, USA.

Winter, M., D. H. Johnson, and J. Faaborg. 2000. Evidence for edge effects on multiple levels in tallgrass prairie. Condor 102:256-266.

Worton, B. J. 1987. A review of models of home range for animal movement. Ecological Modelling 38:277-298.

_____. 1989. Kernel methods for estimating the utilization distribution in home-range studies. Ecology 70:164-168.

Yackel Adams, A., S. K. Skagen, and R. D. Adams. 2001. Movements and survival of lark bunting fledglings. Condor 103:643-647.

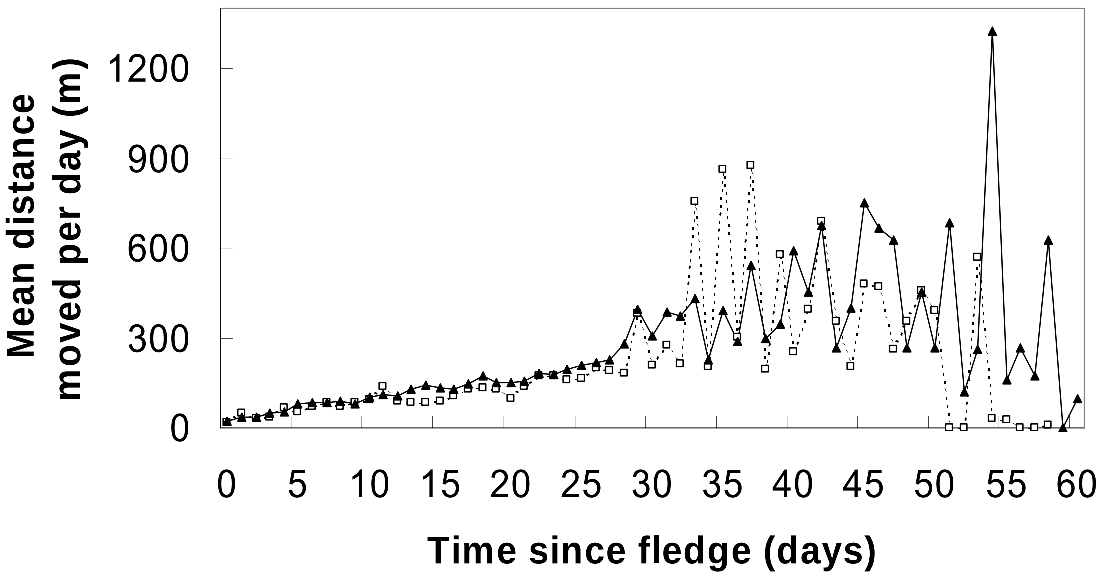

Fig. 1. Average distance moved per day (m) for juvenile dickcissels (*n* = 74; lines with filled triangles) and eastern meadowlarks (*n* = 64; dashed lines with open squares) with ≥ 30 detections in southwestern Missouri, 2002 to 2004.

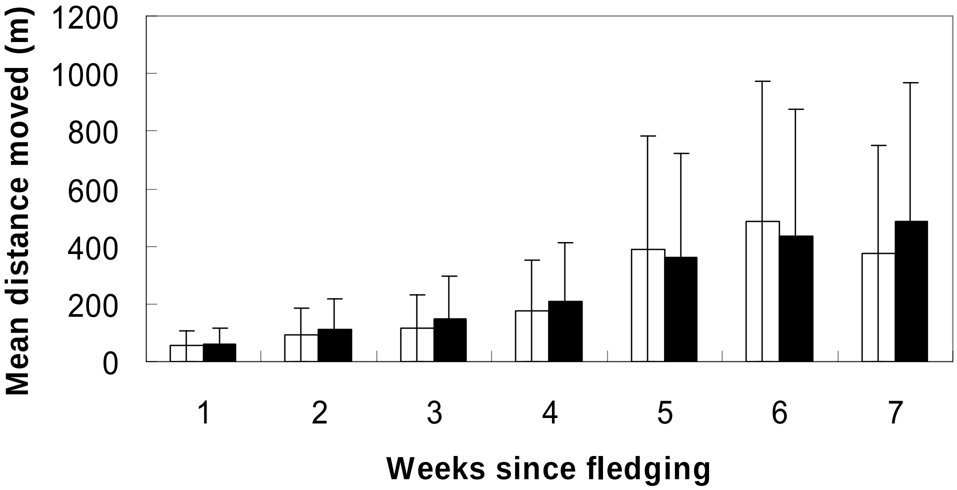

Fig. 2. Average distance moved per week (m) shown with one standard error for the first seven weeks post-fledge for juvenile dickcissels (n = 74; open bars) and eastern meadowlarks (n = 64; filled bars) with ≥ 30 detections in southwestern Missouri, 2002 to 2004.

Fig. 3. Average 50% and 95% home range contours (ha) for juvenile dickcissels (*n* = 74; bars with horizontal lines) and eastern meadowlarks (*n* = 64; open bars) with ≥ 30 detections in southwestern Missouri, 2002 to 2004.

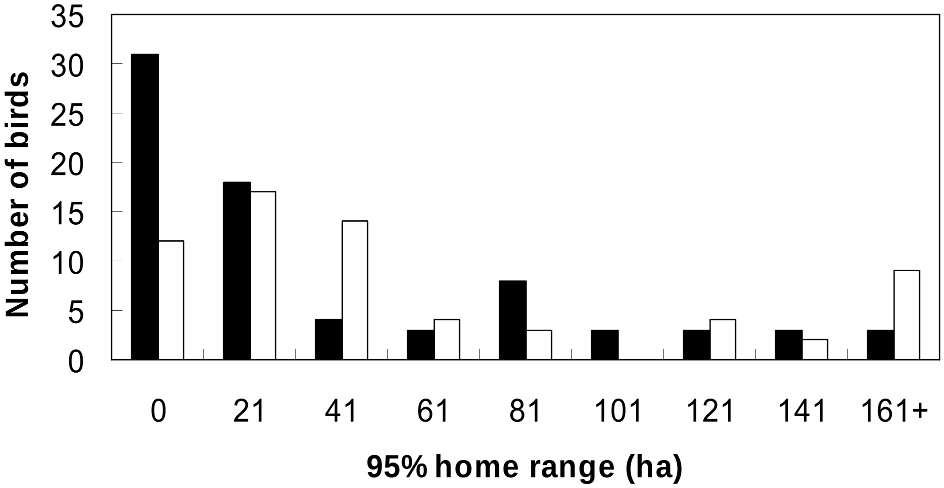

Fig. 4. Frequency histogram of 95% home range contours (ha) for juvenile dickcissels (*n* = 74; filled bars) and eastern meadowlarks (*n* = 64; open bars) with ≥ 30 detections in southwestern Missouri, 2002 to 2004.

Fig. 5. Example of a central (lower right) and exploratory (upper left) post-fledging home range (95% contour) for juvenile dickcissels in southwestern Missouri, 2002 to 2004. Points represent sequential telemetry locations for an individual connected with a movement path line. The outline of the study site boundary is also shown. Central home ranges displayed the majority of telemetry locations in the central natal area and showed a few long distance movements (> 500 m), but generally either returned to the central area or dispersed from the area after movement away from the central natal core. Exploratory home ranges displayed repeated exploratory movements > 500 long in a non-linear fashion and did not show clustering or evidence of multiple dispersal areas.

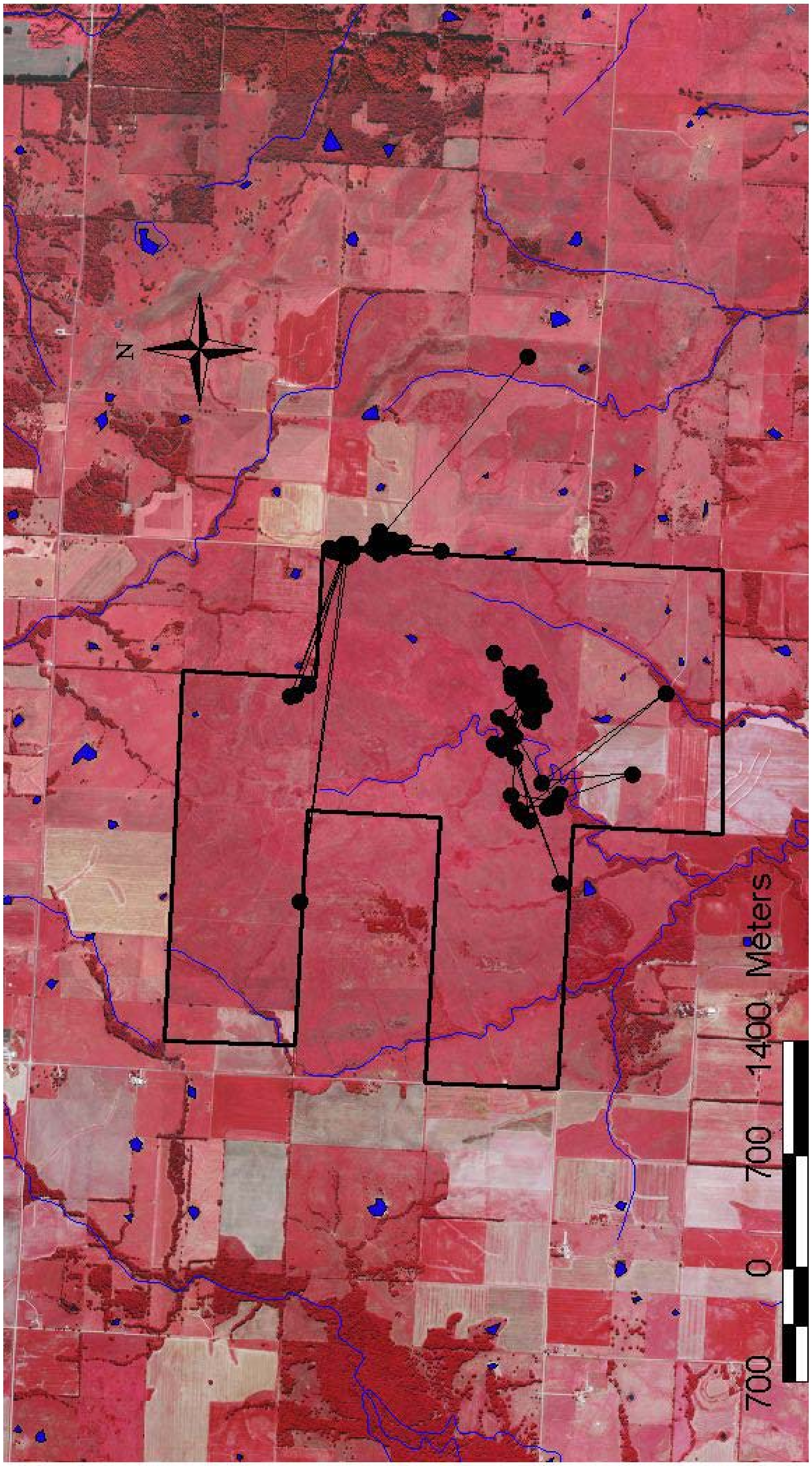
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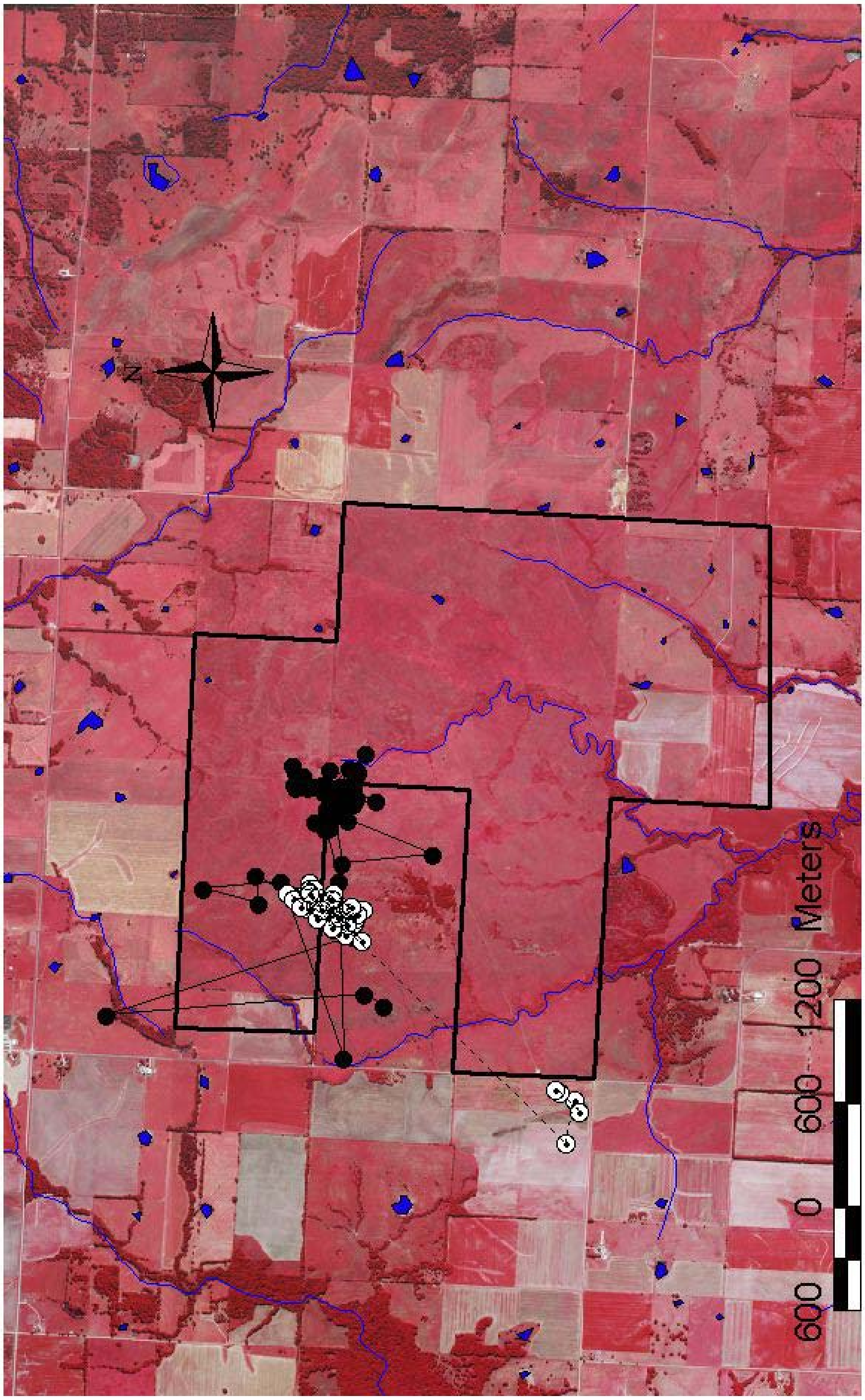
Fig. 6.   Example of a central (open circles with an inner dot and dashed lines) and exploratory (filled circles with lines) post-fledging home range (95% contour) for juvenile eastern meadowlarks in southwestern Missouri, 2002 to 2004. Points represent sequential locations for an individual connected with a movement path line. The outline of the study site boundary is also shown. Central home ranges displayed the majority of telemetry locations in the central natal area and showed a few long distance movements (> 500 m), but generally either returned to the central area or dispersed from the area after movement away from the central natal core. Exploratory home ranges displayed repeated exploratory movements > 500 long in a non-linear fashion and did not show clustering or evidence of multiple dispersal areas.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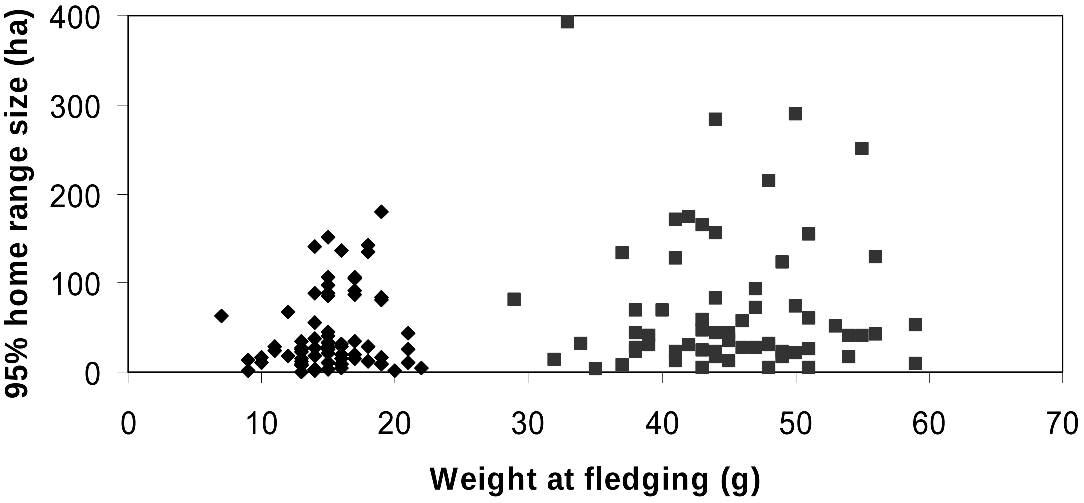

Fig. 7. Average 95% contour home range size (ha) as a function of weight at fledging (g) shown with one standard error for juvenile dickcissels (*n* = 74; filled bars) and eastern meadowlarks (*n* = 64; open bars) with ≥ 30 detections in southwestern Missouri, 2002 to 2004.

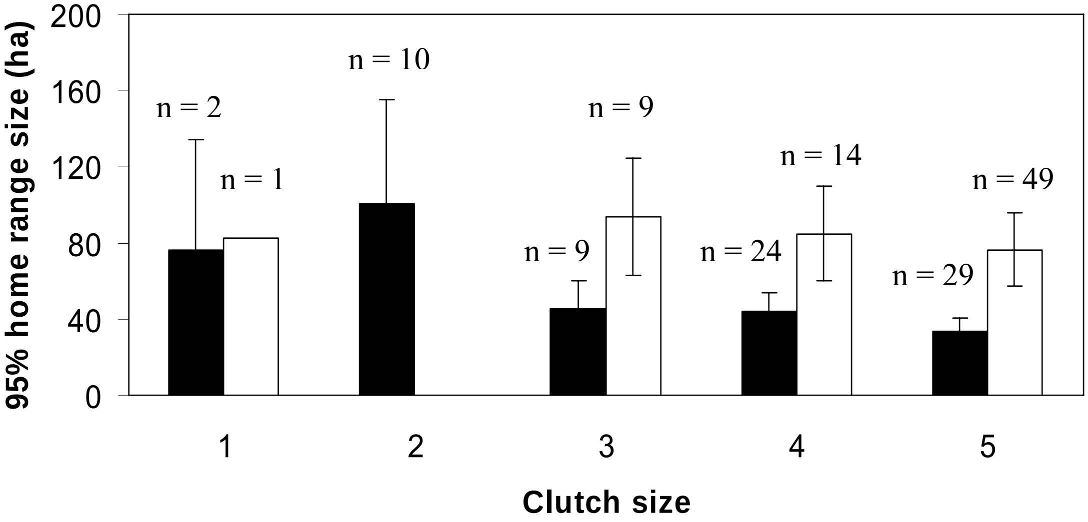

Fig. 8. Average 95% contour home range size (ha) shown with one standard error within each clutch size for juvenile dickcissels (*n* = 74; filled bars) and eastern meadowlarks (*n* = 64; open bars) with ≥ 30 detections in southwestern Missouri, 2002 to 2004. Sample sizes within each clutch size are shown above each bar.

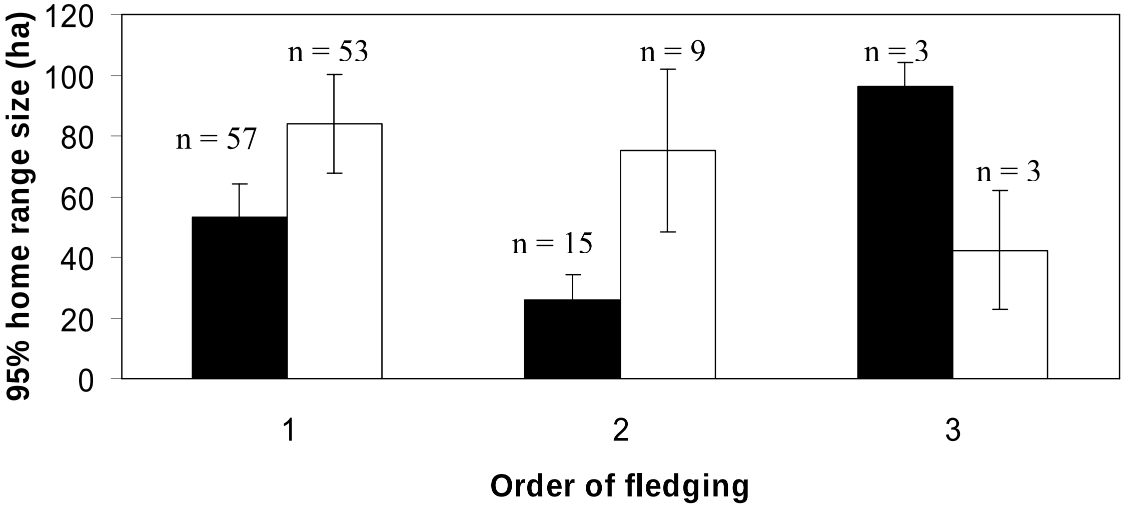

Fig. 9. Average 95% contour home range size (ha) shown with one standard error by order of fledging for juvenile dickcissels ($n = 74$; filled bars) and eastern meadowlarks ($n = 64$; open bars) with ≥ 30 detections in southwestern Missouri, 2002 to 2004. Sample sizes for the number of individual birds are shown above each bar.

Table 1. Model results for predicting home range size in juvenile dickcissels (*n* = 26) from southwestern Missouri in 2002. Models are listed in decreasing order of weight of evidence (w_i).

Model name	Model Structure	K^a	ΔAIC_c^b	w_i
Site quality	Intercept +site +clutch +weight + sibs	5	0.00	0.99
Main effects	Intercept + site +clutch + weight +sibs + Julian date + order	10	5.45	0.06
Age and food demands	Intercept + order*sibs + order*weight	7	8.60	0.01
Food demands	Intercept + sibs*clutch + weight	3	12.07	0.00
Weight	Intercept + weight	2	14.70	0.00
Age and parent quality	Intercept + order*sibs	4	17.88	0.00
Age and brood threshold	Intercept + weight*weight + sibs*sibs	3	18.67	0.00
Clutch size	Intercept + clutch size	2	19.20	0.00
Site quality and temporal	Intercept + order +site	6	19.90	0.00
Siblings	Intercept + siblings	2	20.20	0.00
Temporal and spatial	Intercept + site +Julian date	4	20.58	0.00

Table 1. continued.

Model name	Model Structure	K	ΔAIC_c	w_i
Age threshold	Intercept + weight*weight	2	20.60	0.00
Order of fledge	Intercept +order	4	23.78	0.00
Julian date	Intercept + Julian date	2	25.70	0.00
Null	Intercept	1	26.25	0.00
Temporal and age threshold	Intercept + Julian date *Julian date + weight *weight	3	32.47	0.00
Temporal threshold	Intercept + Julian date *Julian date	2	37.50	0.00
Global[c]	Intercept + site +clutch + weight +sibs + Julian date + order + weight *weight + Julian date *Julian date + sibs*sibs + order*sibs + sibs*clutch	20	134.08	0.00

[a] Number of parameters.

[b] For small sample sizes where *n* / K < 40 (Burnham and Anderson 1998).

[c] Model fit ($\chi^2 = 196$, $P < 0.001$) and predictive power ($r^2 = 0.34$) were adequate.

Table 2. Parameter coefficients (β) and standard errors (SE) from the best model (w_i = 0.99) for predicting home range size in juvenile dickcissels (*n* = 26) from southwestern Missouri in 2002.

Variable	β	SE
Intercept	-262.74	165.42
Site (Taberville)	-14.58	43.55
Clutch	-19.28	21.23
Siblings	0.20	21.91
Weight	24.54	8.71

Table 3. Model results for predicting home range size in juvenile dickcissels (*n* = 48) from southwestern Missouri in 2003 to 2004. Models are listed in order of decreasing weight of evidence (w_i).

Model name	Model Structure	K^a	$\Delta AIC_c{}^b$	w_i
Main effects	Intercept + site + clutch + weight + sibs + Julian date + order + year	12	0.00	0.56
Age and food demands	Intercept + order*sibs + order*weight	7	0.89	0.36
Site quality and temporal	Intercept + order + site +year	8	5.48	0.03
Age and parent quality	Intercept + order*sibs	4	7.32	0.01
Order of fledge	Intercept +order	4	8.02	0.01
Site quality	Intercept + site + clutch + weight + sibs	5	12.44	0.00
Siblings	Intercept + siblings	2	18.46	0.00
Clutch size	Intercept + clutch size	2	20.06	0.00
Temporal and spatial	Intercept + site + Julian date + year	6	20.84	0.00

Table 3. continued.

Model name	Model Structure	K	ΔAIC_c	w_i
Global[c]	Intercept + site + clutch + weight + sibs + year + Julian date + order + weight* weight + Julian date*Julian date + sibs*sibs + order*sibs + sibs*clutch + order*weight	22	20.97	0.00
Food demands	Intercept + sibs*clutch + weight	3	21.24	0.00
Year	Intercept +year	3	21.24	0.00
Weight	Intercept + weight	2	22.76	0.00
Null	Intercept	1	23.58	0.00
Julian date	Intercept + Julian date	2	25.86	0.00
Age and brood threshold	Intercept + weight*weight + sibs*sibs	3	26.34	0.00
Age threshold	Intercept + weight*weight	2	29.56	0.00
Temporal threshold	Intercept + Julian date*Julian date	2	37.56	0.00
Temporal and age threshold	Intercept + Julian date*Julian date + weight*weight	3	43.54	0.00

[a] Number of parameters.

[b] For small sample sizes where *n* / K < 40 (Burnham and Anderson 1998).

[c] Model fit ($\chi^2 = 170$, $P < 0.001$) was adequate but predictive ability was low ($r^2 = 0.13$).

Table 4. Model averaged parameter coefficients (β) and standard errors (SE) from models predicting home range size in juvenile dickcissels (*n* = 48) in southwestern Missouri, 2003 to 2004.

Parameter	β	SE
Intercept	-0.88	4.05
Siblings	1.55	0.86
Year (2003)	0.75	2.36
Weight	0.19	0.31
Site (Taberville)	0.02	1.39
Clutch size	-0.34	1.08
Order of fledge (1)	-7.05	5.15
Order of fledge (2)	-9.89	5.26
Julian date	0.09	0.15
Julian date*Julian date	5.00 x 10^{-4}	8.30 x 10^{-4}
Weight*weight	6.92 x 10^{-3}	0.04
Clutch*sibs	-0.01	0.04
Sibs*order (1)	0.65	0.64
Sibs*order (2)	1.17	1.44
Sibs*order (3)	0.32	9.47
Weight*order (1)	0.14	0.47
Weight*order (2)	-0.16	0.47
Weight*order (3)	5.30	16.64

Table 4. continued.

Parameter	β	SE
Siblings	0.82	0.37

Table 5. Results from the best model ($w_i = 0.99$) for predicting home range size of juvenile eastern meadowlarks ($n = 26$) in southwestern Missouri in 2003. Models are listed in order of decreasing weight of evidence (w_i).

Model name	Model structure	K[a]	ΔAIC_c[b]	w_i
Main effects	Intercept + site + clutch + weight + sibs + order	9	0.00	0.99
Site quality	Intercept + site + clutch + weight + sibs	6	14.37	0.00
Age and food demands	Intercept + order*sibs + order*weight	7	16.47	0.00
Site quality and temporal	Intercept + order +site	6	17.27	0.00
Age and parent quality	Intercept + order*sibs	4	20.75	0.00
Order of fledge	Intercept +order	4	23.95	0.00
Clutch size	Intercept + clutch size	2	31.77	0.00
Site	Intercept + site	3	33.74	0.00
Siblings	Intercept + siblings	2	34.27	0.00
Food demands	Intercept + sibs*clutch + weight	3	34.44	0.00

Table 5. continued.

Model name	Model Structure	K	ΔAIC_c	w_i
Null	Intercept	1	41.81	0.00
Age and brood threshold	Intercept + weight*weight + sibs*sibs	3	43.84	0.00
Weight	Intercept + weight	2	47.57	0.00
Age threshold	Intercept + weight*weight	2	47.57	0.00
Global[c]	Intercept + site + clutch + weight +sibs + order + weight*weight + sibs*sibs + order*sibs + sibs*clutch + order*weight	18	65.66	0.00

[a] Number of parameters.

[b] For small sample sizes where *n* / K < 40 (Burnham and Anderson 1998).

[c] Model fit ($\chi^2 = 181$, $P < 0.001$) was adequate but predictive power was low ($r^2 = 0.12$).

Table 6. Relative importance values for variables used to predict home range size in juvenile eastern meadowlarks ($n = 26$) in southwestern Missouri in 2003. Variables are shown in decreasing order of relative importance.

Variable	Relative importance
Weight	1.0000
Order of fledge	1.0000
Siblings	0.9998
Site	0.9997
Clutch size	0.9995

Table 7. Results from the best model ($w_i = 0.99$) for predicting home range size of juvenile eastern meadowlarks (*n* = 26) in southwestern Missouri in 2003. Models are listed in decreasing order of weight of evidence (w_i).

Model name	Model structure	K^a	ΔAIC_c^b	w_i
Main effects	Intercept + site + clutch + weight + sibs + Julian date + order	10	0.00	0.97
Site quality and temporal	Intercept + order +site	6	8.47	0.01
Age and food demands	Intercept + order*sibs + order*weight	7	10.51	0.00
Site quality	Intercept + site + clutch + weight + sibs	6	11.57	0.00
Age and parent quality	Intercept + order*sibs	4	13.02	0.00
Order of fledge	Intercept +order	4	16.02	0.00
Temporal and spatial	Intercept + site + Julian date	4	23.82	0.00
Site	Intercept + site	3	24.44	0.00
Siblings	Intercept + siblings	2	25.46	0.00
Clutch size	Intercept + clutch size	2	26.16	0.00
Food demands	Intercept + sibs*clutch + weight	3	26.94	0.00

Table 7. continued.

Model name	Model Structure	K	ΔAIC_c	w_i
Weight	Intercept + weight	2	29.46	0.00
Julian date	Intercept + Julian date	2	31.06	0.00
Null	Intercept	1	31.96	0.00
Age and brood threshold	Intercept + weight*weight + sibs*sibs	3	35.04	0.00
Age threshold	Intercept + weight*weight	2	38.46	0.00
Temporal threshold	Intercept +Julian date*Julian date	2	42.56	0.00
Age and temporal threshold	Intercept + weight*weight + Julian date* Julian date	3	49.34	0.00
Global[c]	Intercept + site + clutch + weight + sibs + Julian date + order + weight*weight + Julian date*Julian date + sibs*sibs + order*sibs + sibs*clutch	20	61.24	0.00

[a] Number of parameters.

[b] For small sample sizes where n / K < 40 (Burnham and Anderson 1998).

[c] Model fit ($\chi^2 = 166$, $P < 0.001$) was adequate but predictive power was low ($r^2 = 0.10$).

Table 8. Relative importance values for variables used to predict home range size in juvenile eastern meadowlarks ($n = 30$) in southwestern Missouri in 2004. Variables are shown in decreasing order of relative importance.

Variable	Relative importance
Order of fledge	0.998
Siblings	0.996
Weight	0.996
Site	0.994
Clutch size	0.991
Julian date	0.990

CHAPTER 4

SURVIVAL OF POST-FLEDGING GRASSLAND BIRDS IN MISSOURI

ABSTRACT

The post-fledging period is largely undescribed for most avian species, especially grassland birds. As a consequence, this important component of population demography is often estimated from adult survival rates of the same or related species. Our objective was to describe survival patterns during the post-fledging period for two species of grassland birds in Missouri. We used radio telemetry to follow 248 juvenile dickcissels (*Spiza americana*) and 164 juvenile eastern meadowlarks (*Sturnella magna*) in southwestern Missouri, from 2002 to 2004. We used Cox proportional hazards models to evaluate the effects of biological, spatial, and temporal covariates on survival. We also identified sources of mortality using visual observations, transmitter recoveries, and remains of juvenile birds. Models predicting survival without covariates were better than models with covariates for both species, so we used the null model to estimate survival. The cumulative probability of survival for dickcissels declined by 35% during the first 4 days post-fledging and remained at 0.547 from day 27 after fledging to the end of the study period at day 58 after fledging. The hazard function declined rapidly during the first 10 days after fledging and then leveled off after 16 days. The cumulative probability of survival declined more slowly for meadowlarks compared to dickcissels. From day 37 after fledging to the end of the study period at day 72 after fledging, the cumulative probability of survival was 0.607. The hazard function began lower and declined more quickly than for dickcissels, but leveled off at 15 days post-fledging. Our estimates of post-fledging survival were within the range reported by other studies, but our dickcissel

estimates were much larger than reported for a similar species in another prairie system. Snakes were the dominant predator of both species and had the highest rates of cause-mortality during the study period compared to other causes. We suggest that researchers consider documenting the dominant predator groups with avian survival studies, so that conservation strategies designed to reduce predation pressure by altering landscape features will target the appropriate predator groups.

INTRODUCTION

The post-fledging period is a crucial life history stage for juvenile birds as they complete the prebasic molt and begin to build fat reserves for migration while trying to avoid predators (Moore 1993, Vega Rivera et al. 1998). Despite the importance of the post-fledging period, it is widely regarded as the least understood part of the avian life cycle (Part 1990, Morton 1991, Baker 1993, King and Belthoff 2001). Although several authors have provided survival estimates for the post-fledging period (Woolfenden 1978, Sullivan 1989, Zann and Runciman 1994, Anders et al. 1997, Gardali et al. 2003, Cohen and Lindell 2004), information on grassland birds is limited (Kershner 2001, Yackel Adams 2001).

Existing research across bird groups indicates predation is the main cause of mortality during the post-fledging period (Sullivan 1989, Anders et al. 1997, Kershner 2001, Yackel Adams et al. 2001, Fink 2003). We have limited information about the identity of dominant predators of juvenile grassland birds, but research from nest studies indicates the dominant predator groups may shift from snakes or small mammals in grasslands (Pietz and Granfors 2000, Renfrew and Ribic 2003, Thompson and Burhans 2003) to mid-sized mammals in forests (Thompson and Burhans 2003). As a result, different conservation strategies to reduce predation may be required to increase the survival of juvenile grassland birds.

Understanding survival patterns and the most important factors influencing survival rates during the post-fledging period are needed for effective conservation and management. Current grassland conservation models, such as the Bird Conservation Area concept proposed by the Midwest Working Group of Partners in Flight (PIF) (Pashley and Fitzgerald 1996), strive to protect large blocks of grassland and minimize woody invasion to reduce edge effects and predation associated with edges. However, we suggest this model recommends landscape management strategies based on the assumption that the dominant predators are mammals and raptors, as documented for forest songbirds in Missouri (Thompson and Burhans 2003). Recent studies on nest predators of grassland birds have produced mixed results (Pietz and Granfors 2000, Renfrew and Ribic 2003, Thompson and Burhans 2003), but seem to indicate that raccoons (*Procyon lotor*) (Renfrew and Ribic 2003) and small mammals (Pietz and Granfors 2000) are the dominant nest predators in northern grasslands but that snakes predominate in southern grasslands and old fields (Thompson and Burhans 2003, L. Wolfenbarger pers. comm., K. Suedkamp Wells, pers. obs).

Our objective was to describe survival patterns for two species of grassland birds during the post-fledging period in southwestern Missouri. As a consequence, we were also interested in quantifying cause-specific mortalities. We focused on dickcissels (*Spiza americana*) and eastern meadowlarks (*Sturnella magna*) because they were common grassland species on our site, have a large enough body size (> 20 g) to facilitate wearing transmitters for a sufficient time period, and have shown declining population trends nationwide between 1966 and 2003 (Sauer et al. 2004). In addition, dickcissels are currently listed on the PIF Continental Watchlist as a species with declines or high threats and in need of management (Rich et al. 2004).

METHODS

Study Sites

We conducted this study at Taberville Conservation Area (38° N, 93° W) and Wah'Kon-Tah Prairie (37°N, 94° W) in Cedar and St. Clair Counties in southwestern Missouri. Taberville Conservation area is a 680-ha prairie owned and managed by the Missouri Department of Conservation (MDC) and embedded in an agricultural matrix of crops (wheat, soybeans, and corn) and private land. Wah'Kon-Tah Prairie is a 1930-ha prairie owned by the Missouri Chapter of The Nature Conservancy (TNC) and jointly managed by MDC and TNC located at the northern periphery of El Dorado Springs, Missouri (population ~ 4,000 people). Both sites are part of a network of focal areas targeting grassland bird conservation in Missouri. Dominant land management practices included livestock grazing, prescribed burning, seed harvesting, and haying. Woody removal is a minor management practice that occurs along woody draws, fence lines, and pasture borders. The study sites are divided into management units that receive some sort of management practice (primarily prescribed burning or haying) at least once every three years. Dominant vegetation was composed of bluestem grasses and included big bluestem (*Andropogon gerardi*), little bluestem (*Schizachyrium scoparium*), and indian grass (*Sorghastrum nutans*). Forb species included coneflowers (*Echinacea* spp.), white wild indigo (*Baptisia alba*), blazing star (*Liatrus* spp.), compass plant (*Silphium laciniatum*), milkweeds (*Aesclepias* spp), and sunflowers (*Helianthus* spp.). Dominant, native woody species include smooth sumac (*Rhus glabra*), persimmon (*Diospyros virginiana*), blackberry (*Rubus* spp.), and roses (*Rosa* spp.). Throughout this chapter unless specifically stated otherwise, we use the term woody species to describe the small, shrub species listed above that are native to this prairie and were not planted for human use.

Bird Capture and Handling

We located nests of both species using systematic searches and haphazard walks from 0600 to 1400 hours each day between the third week of April and the second week of August from 2002 to 2004. When possible, we used behavioral cues of the parents to indicate the presence of a nest nearby. After locating each nest, we recorded the GPS coordinates and marked the location by placing colored flagging tape at least 5 m away. At each nest we recorded the species, content, parental activity, and presence of any non-host eggs. If the nest contained nestlings, we attempted to age the nestlings using the presence of down, whether the eyes were open or not, the extent of pin feather development, or the presence of a full complement of feathers. Based on our observations, we were usually successful at aging nestlings within two days of their true age depending on growth rates and weather conditions. We monitored each nest every three to four days until just prior to fledging and then switched to daily nest checks. Two to three days prior to fledging, we attached a metal USFWS band to the left leg and a unique combination of plastic, UV-resistant Darvic bands (Avinet, Dryden, New York) to the right leg and weighed each individual.

We began processing each bird by weighing it to the nearest gram using a spring scale (Avinet, Dryden, New York). Following a modification of the Rappole and Tipton method (1991) previously evaluated (Suedkamp Wells et al. 2003), we attached 0.7-gram transmitters with a 10-cm whip antenna (Biotrack, Dorset, United Kingdom) to the back of each bird using a leg harness. Battery life for each transmitter was expected to range between 55 and 60 days. We constructed the leg harness from cotton, elastic beading cord to allow room for growth. Using super glue (Duro, Avon, Ohio), we secured the bottom of the transmitter to the back of the bird. After attaching transmitters to each bird, we placed the brood back in the nest. Handling and processing time usually was between 2 and 5 minutes per bird.

Radiotracking

Using telemetry, we began tracking birds using homing for visual confirmation (Mech 1983) the day after attaching transmitters. If the brood remained in the nest the day after attachment, we returned each subsequent morning and began tracking when at least one brood member fledged. We tracked each bird twice daily in non-consecutive time blocks for a minimum of 50 detections per individual for a total of 25 individuals of both species (Garton et al. 2001). The four tracking blocks were early morning (0600 to 0930 hours), mid-morning (0930 to 1230 hours), afternoon (1230 to 1700 hours), and evening (1700 to 2130 hours). We grouped the time blocks to reflect biological activity and environmental constraints such as hot temperatures when activity is reduced. The first day of tracking for an individual occurred in the early morning and afternoon periods. On the second day, tracking occurred in the mid-morning and evening hours. We continued to alternate days on this schedule to capture locations representative of all diurnal activities typical of tracking studies (Garton et al. 2001). We avoided tracking before 0600 hours and after 2130 hours to reduce the risk of stepping on juvenile birds when they could not be visually located. After reaching 50 detections, each individual was tracked once daily alternating between the first two and last two periods of the day until the bird died, the transmitter was recovered, or the study period ended.

We tracked each individual until we recovered the transmitter or dead bird through the end of August in each field season. Immediately after being unable to locate an individual, we performed extensive searches of the immediate area on foot with a team of assistants. If we were unable to locate the individual, we broadened the search to include all roads within 3.2 km of the last known location using an omni antennae mounted on the roof of a truck. We continued to search for missing individuals twice daily for one week after their disappearance. At the start of the second week, we reduced search time to one attempt per day. We also attempted to locate missing birds by flying at least 5 km strips over the study area in a helicopter twice monthly between 1 June and 30 August each year.

Survival Estimation

We used Cox proportional hazards models to estimate survival as a function of biological, spatial (site only), and temporal covariates. Biological covariates included the weight of each juvenile at fledging (g) and a dichotomous variable reflecting the number of juveniles fledging from each nest (dickcissels, < 2 coded 0, ≥ 3 coded 1; meadowlarks, < 3 coded 0, ≥ 4 coded 1). We selected these cut off points because they represent average and large brood sizes at fledging, respectively based on our observations (K. Suedkamp Wells, unpub. data). We coded brood size dichotomously because previous research has indicated heavier birds are more likely to survive (Krementz et al. 1989, Naef-Daenzer et al. 2001). In addition, we observed that a subset of the parents associated with large broods at fledging often demonstrated more vigilance and appeared to spend more time with their broods despite feeding constraints than parents of small to average broods (K. Suedkamp Wells, pers. obs.). We used study site as a spatial variable because prior work on resource selection (see Chapters 1 and 2) and nest success showed high variability between sites and within highly productive patches on a single site (K. Suedkamp Wells, unpubl. data). Finally, we used year and Julian date at fledging to capture potential differences in temporal patterns.

Prior to model fitting, we screened our data to check several assumptions. First, we checked for normality using probability plots in SYSTAT (SPSS 1999) and then applied appropriate transformations if necessary. Next, we removed individuals where mortality occurred within 24 hours of fledging to insure our data set only included those individuals who successfully fledged. If we obtained at least one detection to confirm a juvenile had successfully fledged, we included that individual in further analysis. Second, we also removed mortalities due to observers (see Table 3); immobile juveniles were accidentally stepped on while being located. Another assumption we made was that any juvenile that disappeared and could not be located within one week of fledging had been depredated. This assumption was based on our experience with the limited mobility of juvenile birds during the first week post-fledging (see Chapter 3), our intensive search patterns, and low transmitter failure rate (< 2%), we were comfortable with this assumption. We determined the duration (number of days) of risk for each bird by assuming each individual was at risk until we observed a fate or censored an individual. If there was a time gap between the last observation of a bird and the determination of fate, we assumed each bird was at risk for the entire day they were last located. If a bird was missing after 30 days of age and suspected to have dispersed, we used the last confirmed observation as the date for censorship.

Prior to fitting proportional hazards models, we used PROC REG in SAS (SAS Institute 2001) to assess multicollinearity using tolerance values (Allison 1995). Next, we fit a global model containing all of our covariates using Cox proportional hazards in PROC PHREG in SAS (SAS Institute 2001). We treated time as a continuous variable by using the exact method to handle ties with Breslow's approximation (Allison 1995). We used a likelihood ratio test to compare model fit between a model with covariates and the null model. We assessed the significance of each covariate in the model using chi square values from Wald tests (Allison 1995) and used an alpha level of 0.05 for significance. Finally, we graphed the hazard function for each species using the SMOOTH macro (Allison 1995) with PROC PHREG in SAS.

Causes of Mortality

We used visual observations combined with a series of assumptions to assign juvenile deaths to one of ten causes of mortality for juvenile birds (see Table 3). If a juvenile bird disappeared during the first week after fledging and we were unable to recover the transmitter, we classified those mortalities as unidentified depredations. Juvenile birds are relatively immobile during the first week post-fledge (see Chapter 3) and were not able to move a sufficient distant to prevent location with telemetry. If we observed the predator while tracking, we assigned that predation event to the mortality category associated with that species. Transmitters that were recovered in snake feces were particularly distinctive (K. Suedkamp Wells, pers. obs.) and assigned to the general snake predation category. Transmitters that were recovered with tooth marks, located near mammal scat, burrows, or dens, were assigned to the mammal predation category. Transmitters that were never recovered but tracked to a small burrow (< 6 cm in diameter) were assigned to the combined snake or small mammal burrow depredation category. Based on our observations, we suspect the majority of these burrows were snake burrows. In one case, we assigned a mortality to the raptor predation category because we recovered the transmitter on a carcass that been cleaned of flesh and was located near a known raptor nest. Juveniles that were recovered intact without evidence of injury in the absence of adverse weather (rain storms or heat waves) were classified as unknown natural deaths. Juveniles that were recovered wet or cold after a rain storm, or hot and decomposing after hot weather, were assigned to the weather mortality category. If juveniles were accidentally stepped on while being located, we classified those mortalities as human accidents. Juveniles that were killed during management activities were assigned to the farm and management equipment category. Finally, juveniles that were found dead with their bands or antennae tangled in vegetation were classified as research equipment mortalities.

We also generated cause-specific mortality rates for each day and the study period interval using MICROMORT (Heisey and Fuller 1985). We pooled data across years for each species and used one interval. To define the length of the study period interval, we used the longest duration at risk in our data set corresponding to a censored event or confirmed death (58 days for dickcissels and 72 days for meadowlarks). To calculate the number of transmitter days, we summed the duration of risk for individuals across years for each species. We estimated cause-specific mortality for the five dominant categories we observed (predation, snakes, mammals, natural death, and weather). Mortality rates for snakes most likely represent an underestimate because we did not include mortalities in the snake or small burrow category, even though we suspect the majority of those depredations were by snakes (K. Suedkamp Wells, pers. obs.).

RESULTS

We attached transmitters to 248 juvenile dickcissels and 164 eastern meadowlarks between 2002 and 2004. We assumed or confirmed mortality for 66% ($n = 164$) of dickcissels and 49% ($n = 81$) of meadowlarks. Our data showed no evidence of multicollinearity for either species (all tolerances > 0.90), so we proceeded with model fitting. For dickcissels, 56% of the data was censored. Results from the likelihood ratio test indicated models without covariates were better than models with covariates ($\chi^2 = 11$, $P = 0.40$) for dickcissels. In addition, there were no significant covariates (all $P > 0.07$). As a result, we estimated survival probabilities and the hazard function using the null model for dickcissels. The cumulative probability of survival for dickcissels declined rapidly during the first 4 days after fledging and remained at 0.547 from day 27 after fledging to the end of the study period at day 58 after fledging (Table 1). The hazard function declined rapidly during the first 10 days after fledging and then leveled off after 16 days (Fig. 1).

Results for meadowlarks were similar to those for dickcissels. For meadowlarks, 62% of the data were censored. Results from the likelihood ratio test indicated models without covariates were better than models with covariates ($\chi^2 = 4$, $P = 0.54$) for dickcissels. In addition, there were no significant covariates (all $P > 0.18$). As a result, we also estimated survival probabilities and the hazard function using the null model for meadowlarks. The cumulative probability of survival declined more slowly for meadowlarks (Table 2) compared to dickcissels. From day 37 after fledging to the end of the study period at day 72 after fledging, the cumulative probability of survival was 0.607. The hazard function started off lower and declined more quickly (Fig. 2) than dickcissels, but appeared to level off at 15 days post-fledge.

Snakes were the dominant predator of both species (Table 3). Cause-specific mortality rates for the entire study period showed that snakes, natural deaths, weather, and mammals were the leading specific causes (excluding general predation) of mortality for dickcissels (Table 4). Snakes, natural deaths, mammals, and weather were the leading specific causes of mortality for meadowlarks (Table 4). Mortality estimates during the study period were similar between species. In addition, mortality rates associated with snakes during the study period were higher for dickcissels (0.229) compared to meadowlarks (0.145), but the confidence intervals overlapped.

DISCUSSION

Our estimates of post-fledging survival were within the range of those previously reported (0.367 to 0.675) for several other species (Krementz et al. 1989, Sullivan 1989, Anders et al. 1997, Kershner 2001, Yackel Adams et al. 2001, Fink 2003, Cohen and Lindell 2004, Webb et al. 2004). Our estimate of post-fledging survival for dickcissels during the nine-week study period (0.547) is comparable to the meadowlark estimate (0.53) reported by Kershner (2001) for his 14-week study period in Illinois. Our post-fledging estimate for the study period is also comparable to annual estimates of juvenile survival (0.47) for common ravens (*Corvus corax*) in California (Webb et al. 2004). However, post-fledging survival rates for meadowlarks in this study also were lower than overall estimates for wood thrush in Missouri 13-weeks post-fledging (0.675) reported by

Fink (2003), but higher than estimates reported by Anders et al. (1997) for the same species through eight weeks post-fledging (0.423). Comparing our data with the only other published estimate of post-fledging survival for meadowlarks (Kershner 2001) versus other species illustrates the importance of further research during this critical period on other bird groups and species. Although our study sites and those of Kershner (2001) were located in agricultural matrices of the Midwest, grassland birds in other regions of the U.S. may show different survival patterns.

Comparing post-fledging survival estimates from grassland birds in shortgrass prairies to our estimates from tallgrass prairies suggests that differences in limiting factors may be important. For example, lark buntings (*Calamospiza melanocorys*) in the shortgrass prairie of Colorado showed lower survival rates after the third week post-fledge (0.367; Yackel Adams et al. 2001) than dickcissels in our study 16 days after fledging (0.566). Although dickcissels (mean weight at time of transmitter attachment 15.3 ± 3.0 g, Chapter 3) are similar to lark buntings in terms of body size (mean weight at fledging 21.4 ± 0.5; Yackel Adams et al. 2001), we suggest that different limiting factors in each system shaped survival. For lark buntings, Yackel Adams et al. (2001) suggested that brood division was an evolutionary strategy to increase food delivery or reduce the risk of predation. The presence of brood division and the large daily movements made by juvenile lark buntings suggest that obtaining food resources may have been more important than avoiding risk of predation. In our study, we found limited support for starvation as a dominant hypothesis affecting resource selection at a small scale (see Chapter 1). However, the availability of crops at a landscape-scale was important (see Chapter 2), which we attributed to increased food availability based on our observations.

Finally, daily movements by juvenile birds of both species in our study (see Chapter 3) were also smaller than estimates reported for lark buntings (Yackel Adams et al. 2001), which further suggests that limiting mechanisms affecting survival may differ between the two prairie systems.

Survival patterns during the first month after fledging in our study differed from those reported for other species (Sullivan 1989, Anders et al. 1997, Fink 2003). Sullivan reported that juncos (*Junco phaenotus*) showed peaks in age-specific survivorship when juveniles were flying (day 17 post-fledge) and had become experienced, independent juveniles (day 28 post-fledge). Anders et al. (1997) reported that wood thrush in Missouri forests also showed peaks in survivorship at two weeks, and four to eight weeks post-fledge. Both sets of authors (Sullivan 1989, Anders et al. 1997) associated these peaks in survivorship with developmental milestones, such as learning to fly and forage, that posed less risk for juvenile birds. However, our data show steady and consistent declines in survival probability without any peaks. In contrast to the development process described by other authors (Sullivan 1989, Anders et al. 1997, Fink 2003), we observed gradual increases in ability to fly and forage without punctuated periods. Near the end of the dependence period, especially for meadowlarks, we often observed parents sporadically feeding juveniles, which may have prompted faster learning. In addition, parents of both species also were observed sporadically feeding or halting feeding temporarily during storm events. If parental feeding rates are highly variable, then grassland bird species may be forced to develop their skills more rapidly as an insurance policy against bad weather or other adverse situations limiting feeding rates.

Causes of Mortality

Overall, our estimate of cause-specific mortality for predation during the study period (0.490 to 0.517) was comparable to the estimate Anders et al. (1997) reported for juvenile wood thrush (0.506) during her eight-week study period. In addition, the dominant predator groups we identified are consistent with a video camera study of nest predators in old fields of Missouri (Thompson and Burhans 2003), but are different than other studies of grassland predators (Pietz and Granfors 2000, Renfrew and Ribic 2003). Snakes were the dominant source of mortality and had highest mortality rates for both species. However, we believe our estimates of mortality during the study period associated with snakes underestimate their impact because we had to exclude predation events in the snake or small mammal burrow category. Although we were unable to positively identify the predator group associated with this class of mortality, we suspect the majority were associated with snakes. In our experience, we never observed small mammals using vertical burrows that extended several feet under the ground or burrows without signs of excavation near the entrance (K. Suedkamp Wells, unpubl. data). By inserting cables attached to our receivers into suspected snake burrows, we could often determine that the vertical burrow continued for nearly 1 m. The higher number of mortalities associated with snake or small mammal burrows for meadowlarks compared to dickcissels is most likely because snakes had to exert greater force to swallow the larger juveniles. On several occasions, we observed that snakes depredating juvenile meadowlarks had difficulty disengaging their jaws and swallowing the larger species. As a result, transmitters attached to meadowlarks were more likely to be damaged than those attached to dickcissels because of their larger body size (K Suedkamp Wells, pers. obs.).

Consequently, damaged transmitters or those excreted underground were less likely to be recovered.

Similar to our study, snakes were the dominant predator on songbirds nests in old fields of Missouri where they were documented at 33 of 46 events captured on camera (Thompson and Burhans 2003). In contrast, studies of grassland nest predators in the northern U.S. (Pietz and Granfors 2000, Renfrew and Ribic 2003) and post-fledging studies of Swainson's thrushes (Catharus ustulatus) in coastal California (White 2005) have shown mammals were the dominant predators. In northern grasslands, raccoons and ground squirrels (*Spermophilus* spp.) were the dominant predators (Pietz and Granfors 2000, Renfrew and Ribic 2003). In coastal scrub communities of California, White (2005) reported that casue-specific mortalities during the study period were greatest for small mammals and raptors, respectively. Contrasting studies across plant community types (prairies vs forests vs. coastal scrub) and within grasslands along a latitudinal gradient shows that predator communities vary greatly. Specifically in grasslands, the dominant predator group shifts from mid and small-sized mammals in northern prairies to snakes in southern prairies.

CONSERVATION IMPLICATIONS

Our results have demonstrated a shift in the predator suite between plant community types, and between southern and northern grasslands. As a result, conservation strategies designed to increase population trends by reducing predation levels should target the landscape resources used by the dominant predator groups in that particular system. However, documenting the dominant predators in different systems is often expensive and requires specialized equipment such as video cameras or radio telemetry. As a result, we recommend that researchers consider documenting predators in avian studies using survey methods designed for the predator groups most likely to be dominant. In southern grasslands or other systems where snakes are likely to be among the dominant predators, we suggest that researchers consider the natural history of the dominant snake species and potential biases associated with sampling methods as recommended by Weatherhead and Blouin-Demers (2004).

In addition, we have described post-fledging survival rates that were within the range previously reported by others authors, and high in comparison to one estimate for grassland birds (lark buntings; Yackel Adams et al. 2001). Although juvenile survival is one component of population demographics, we also suggest that researchers evaluate their results in context with other important information such as fecundity and nest success rates. In this study (see Table 1, Chapter 1), the number of young produced per successful nest, daily nest survival rate, Mayfield nest success were similar to other studies from Missouri on these species (McCoy et al. 1999, McCoy et al. 2001, Winter and Faaborg 1999, Winter 1999). Although McCoy et al. (1999, 2001) estimated the source-sink status for several grassland species in northern Missouri, we are uncomfortable extrapolating adult survival rates from other species (dickcissels in this case) and making assumptions about the number of nesting attempts and broods contributing to seasonal fecundity estimates (both species) with the current lack of data. However, the combination of demographic measures we have presented do not suggest that populations of either species are doing comparatively worse than the populations studied elsewhere in Missouri (McCoy et al. 1999, McCoy et al. 2001, Winter and Faaborg 1999, Winter 1999). Given the relatively large size of our study sites for prairie reserves in Missouri, populations on smaller prairie fragments may not perform as well, which should be considered when evaluating the conservation potential of prairie fragments.

LITERATURE CITED

Allison, P. D. 1995. Survival analysis using SAS. A practical guide. SAS Institute, Cary, North Carolina, USA.

Anders, A. D., D. C. Dearborn, J. Faaborg, and F. R. Thompson III. 1997. Juvenile survival in a population of Neotropical migrant birds. Conservation Biology 11:698-707.

Baker, R. R. 1993. The function of post-fledging exploration: a pilot study of three species of passerines ringed in Britain. Ornis Scandinavica 24:71-79.

Cohen, E. B., and C. A. Lindell. 2004. Survival, habitat use, and movements of fledgling white-throated robins (*Turdus assimilis*) in a Costa Rican agricultural landscape. Auk 121:404-414.

Ellsworth, E. A., and J. R. Belthoff. 1999. Effects of social status on the dispersal behavior of juvenile western screech-owls. Animal Behavior 57:883-892.

Fink, M. L. 2003. Post-fledging ecology of juvenile Wood thrush in fragmented and contiguous landscapes. Ph.D. Dissertation, University of Missouri, Columbia, Missouri, USA.

Gardali, T., D. C. Barton, J. D. White, and G. R. Geupel. 2003. Juvenile and adult survival of Swainson's thrush (*Catharus ustulatus*) in coastal California: annual estimates using capture-recapture analyses. Auk 120:1188-1194.

Garton, E. O., M. J. Wisdom, F. A. Leban, and B. K. Johnson. 2001. Experimental design for radio telemetry studies. Pages 16–44 in J. J. Millspaugh and J. M. Marzluff, editors. Radio tracking and animal populations. Academic Press, Inc. San Diego, California, USA.

Heisey, D. M., and T. K. Fuller. 1985. Estimation of survival and cause-specific motality rates using telemetry data. Journal of Wildlife Management 49:668-674.

Kershner, E. L. 2001. Conservation of grassland birds in an agricultural landscape: the importance of habitat availability and demography. Ph.D. Dissertation, University of Illinois at Urbana-Champaign., Urbana, Illinois, USA.

King, R. A. and J. R. Belthoff. 2001. Post-fledging dispersal of burrowing owls in southwestern Idaho: characterization of movements and use of satellite burrows. Condor 103:118-126.

Krementz, D. G., J. D. Nichols, J. E. Hines. 1989. Postfledging survival of European starlings. Ecology 70:646-655.

McCoy, T. D., M. R. Ryan, E. W. Kurzejeski, and L. W. Burger, Jr. 1999. Conservation Reserve Program: source or sink habitat for grassland birds in Missouri? Journal of Wildlife Management 63:530-538.

McCoy, T. D., M. R. Ryan, and L. W. Burger, Jr. 2001. Grassland bird conservation: CP1 vs. CP2 plantings in Conservation Reserve Program Fields in Missouri. American Midland Naturalist 145:1-17.

Mech, L. D. 1983. Handbook of animal radio-tracking. University of Minnesota Press, Minneapolis, Minnesota, USA.

Moore, F. R., S. A. Gauthreaux, Jr., P. Kerlinger, and T. R. Simmons. 1993. Stopover habitat: management implications and guidelines. Pages 58-69 *in* Status and management of Neotropical Migratory birds (D. M. Finch and P. W. Stangel, Eds.). US. Forest Service General Technical Report RM-229. Fort Collins, Colorado, USA.

Morton, M. L., M. W. Wakamatsu, M.E. Pereyra, and G. A Morton. 1991. Postfledging dispersal, habitat imprinting, and philopatry in a montane, migratory sparrow. Ornis Scandinavica 22:98-106.

Naef-Daenzer, B., F. Widmer, and M. Nuber. 2001. Differential post-fledging survival of great and coal tits in relation to their condition and fledging date. Journal of Animal Ecology 70:730-738.

Part, T. 1990. Natal dispersal in the collared flycatcher: possible causes and reproductive consequences. Ornis Scandinavica 21:83-88.

Pashley, D. and J. Fitzgerald. 1996. Results of Prairie Pothole Joint Venture/Partners in Flight meeting, July 11 – 12. Unpublished memorandum.

Pietz, P. J., and D. A. Granfors. 2000. Identifying predators and fates of grassland passerine nests using miniature video cameras. Journal of Wildlife Management 64:71-87.

Rappole, J. H., and A. R. Tipton. 1991. New harness design for attachment of radio transmitters to small passerines. Journal of Field Ornithology 62:335-337.

Renfrew, R. S., and C. A. Ribic. 2003. Grassland passerine nest predators near pasture edges identified on videotape. Auk 120:371-383.

Rich, T. D., C. J. Beardmore, H. Berlanga, P. J. Blancher, M. S. W. Bradstreet, G. S. Butcher, D. W. Demarest, E. H. Dunn, W. C. Hunter, E. E. Inigo-Elias, J. A. Kennedy, A. M. Martell, A. O. Panjabi, D. N. Pashley, K. V. Rosenberg, C. W. Rustay, J. S. Wendt, and T. C. Will. 2004. Partners in Flight North American Landbird Conservation Plan. Cornell Lab or Ornithology, Ithaca, New York, USA.

Samson, F., and F. Knopf. 1994. Prairie conservation. BioScience 44:418-421.

SAS Institute. 2001. SAS, version 8.0. SAS Institute, Cary, North Carolina, USA.

Sauer, J. R., J. E. Hines, and J. Fallon. 2004. The North American Breeding Bird Survey, Results and Analysis 1966 - 2003. Version 2004.1. USGS Patuxent Wildlife Research Center, Laurel, Maryland, USA.

SPSS. 1999. SYSTAT, version 9.0. SPSS, Chicago, Illinois, USA.

Suedkamp Wells, K. M., B. E. Washburn, J. J. Millspaugh, M. R. Ryan, and M. W. Hubbard. 2003. Effects of radio-transmitters on fecal glucocorticoid levels in captive dickcissels. Condor 105:805-810.

Sullivan, K. A. 1989. Predation and starvation: age-specific mortality in juvenile juncos (*Junco phasenotus*). Journal of Animal Ecology 58:275-286.

Thompson, F. R. III, and D. E. Burhans. 2003. Predation of songbird nests differs by predator and between field and forest habitats. Journal of Wildlife Management 67:408-416.

Vega Rivera, J. H., J.H. Rappole, W. J. McShea, and C. A. Haas. 1998. Wood thrush postfledging movements and habitat use in northern Virginia. Condor 100:69-78.

Weatherhead, P. J., and G. Blouin-Demers. 2004. Understanding avian nest predation: why ornithologists should study snakes. Journal of Avian Biology 35:185-190.

Webb, W. C., W. I. Boarman, and J. T. Rotenberry. 2004. Common raven juvenile survival in a human-augmented landscape. Condor 106:517-528.

White, J. D. 2005. Post-fledging survival, resource selection, and dispersal of juvenile Swainson's thrushes in central coastal California. Unpublished PhD Dissertation, University of Missouri, Columbia, Missouri, USA.

Winter, M., and J. Faaborg. 1999. Patterns of area sensitivity in grassland-nesting birds. Conservation Biology 13:1424-1436.

Winter, M. 1999. Nesting biology of dickcissels and Henlsow's sparrows in southwestern Missouri prairie fragments. Wilson Bulletin 111:515-527.

Woolfenden, G. E. 1978. Growth and survival of young Florida scrub jays. Wilson Bulletin 90:1-18.

Yackel Adams, A. S. K. Skagen, and R. D. Adams. 2001. Movements and survival of lark bunting fledglings. Condor 103:643-647.

Zann, R., and D. Runciman. 1994. Survivorship, dispersal, and sex ratios of zebra finches *Taeoniopygia guttata* in southeast Australia. Ibis 136:136-146.

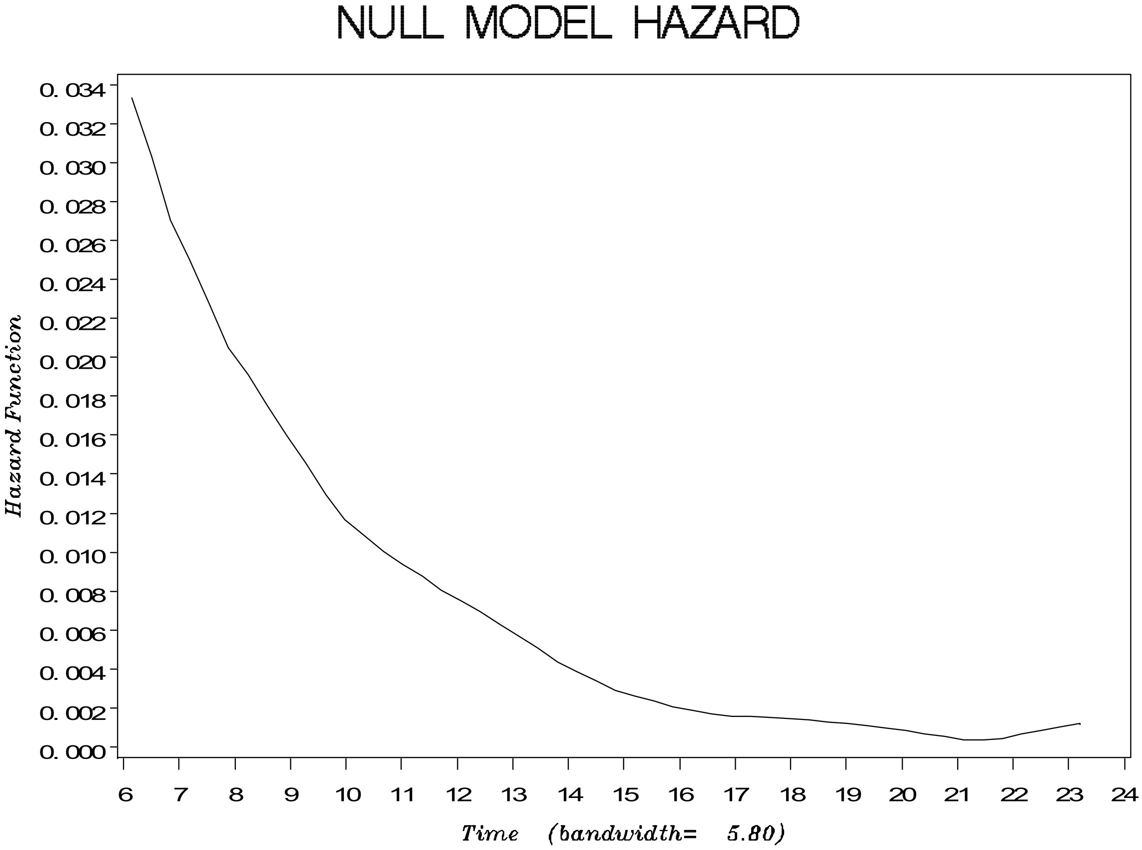

Fig. 1. Hazard function from survival estimates for juvenile dickcissels (*n* = 155) in southwestern Missouri, 2002 to 2004.

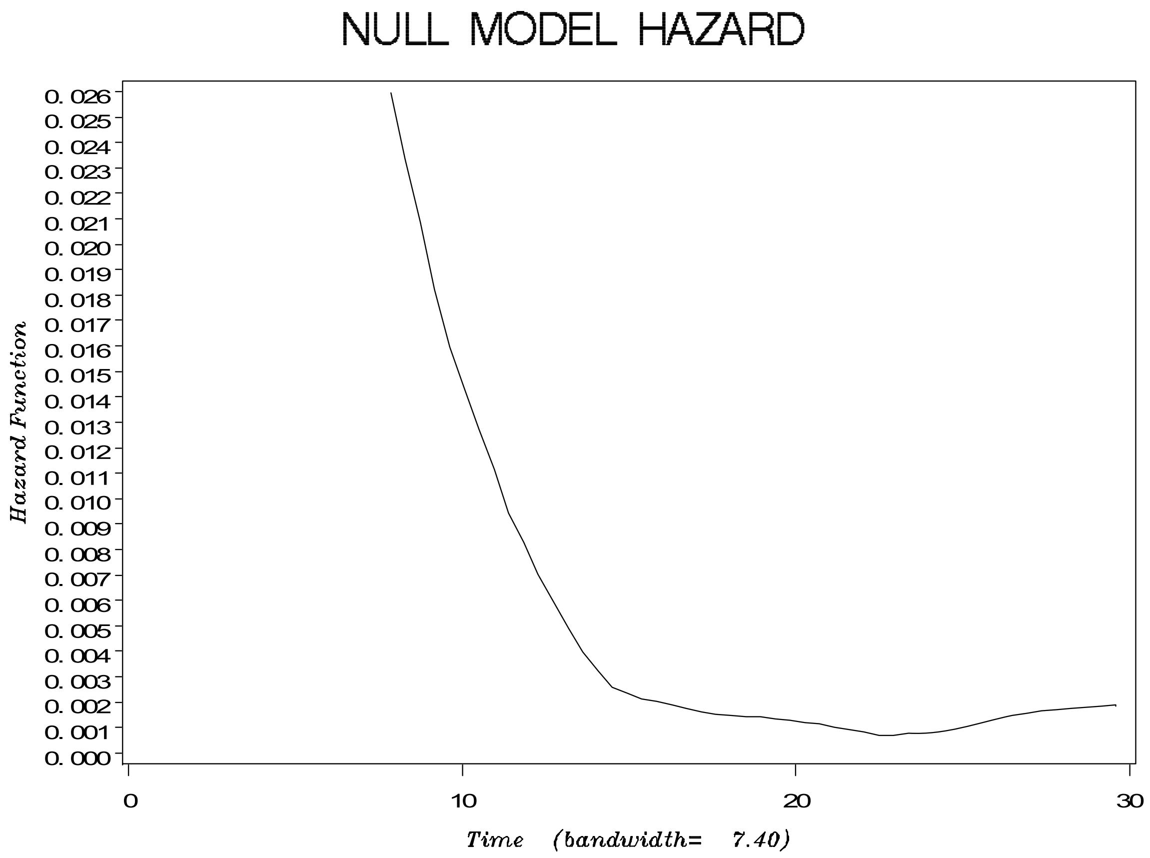

Fig. 2. Hazard function from survival estimates for juvenile eastern meadowlarks (*n* = 107) in southwestern Missouri, 2002 to 2004.

Table 1. Survival probability (S) by time interval (days) shown with 95% confidence limits for juvenile dickcissels (*n* = 155) in southwestern Missouri, 2002 to 2004. Displayed estimates represent those time intervals were mortalities occurred beginning with day zero as the first day out of the nest (Allison 1995).

Time interval (days)	S	LCL	UCL
0	0.903	0.860	0.948
1	0.801	0.744	0.862
2	0.724	0.660	0.793
3	0.698	0.633	0.770
4	0.653	0.585	0.728
5	0.634	0.565	0.710
6	0.621	0.552	0.698
7	0.614	0.545	0.692
8	0.601	0.531	0.680
9	0.587	0.517	0.667
10	0.580	0.510	0.661
11	0.574	0.503	0.654
16	0.566	0.496	0.647
27	0.557	0.486	0.639
29	0.547	0.475	0.631

Table 2. Survival probability (S) by time interval (days) shown with 95% confidence limits for juvenile eastern meadowlarks (*n* = 107) in southwestern Missouri, 2002 to 2004. Displayed estimates represent those time intervals were mortalities occurred beginning with day zero as the first day out of the nest (Allison 1995).

Time interval (days)	S	LCL	UCL
0	0.971	0.949	1.000
1	0.934	0.889	0.981
2	0.831	0.766	0.902
3	0.803	0.734	0.879
4	0.757	0.682	0.839
5	0.728	0.651	0.815
6	0.700	0.621	0.790
7	0.663	0.582	0.756
8	0.654	0.572	0.747
10	0.644	0.562	0.739
17	0.635	0.552	0.730
30	0.623	0.539	0.720
37	0.609	0.524	0.708

Table 3. Sources of mortality for juvenile dickcissels (*n* = 248) and eastern meadowlarks (*n* = 164) in southwestern Missouri, 2002 to 2004.

	Species	
Mortality Source	dickcissel	eastern meadowlark
total snake predation	51	16
unidentified snake predation[a]	31	10
bullsnake (*Pituophis catenifer*)	8	4
speckled king snake (*Lampropeltis getula*)	6	0
prairie king snake (*Lampropeltis callagaster)*	3	0
northern water snake (*Nerodia sipedon*)	1	2
yellow-bellied racer (*Coluber constrictor*)	2	0
three-toed box turtle (*Terrapene carolina*)	1	0
raptor[b]	1	0
mammal[c]	15	11
snake or small mammal burrow[d]	28	4
unknown natural death[e]	23	12
unidentified depredation[f]	19	23
weather[g]	16	9
human accident[h]	2	6
farm or management equipment	5	0
research equipment[i]	3	0
Total	164	81

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[a] Mortalities where the snake species could not be identified or where the transmitter was recovered from snake feces

[b] Assumed to be raptor mortality because the skeleton was picked clean of flesh and located near a known raptor nest.

[c] Identified as mammal mortalities because of teeth marks on the transmitter or carcass.

[d] Mortalities tracked to a snake or small burrow generally < 6 cm in diameter where a transmitter recovery was never made. We suspect the majority of these were most likely snake depredations.

[e] Mortalities without apparent injuries where the bird appeared to have died from natural causes not related to predation, weather, or other specific causes listed here.

[f] Mortalities that occurred before juvenile birds were able to move the distance associated with the point of transmitter recovery where a predator identification could not be made.

[g] Mortalities that were discovered immediately after a rain storm or period of hot weather where the juvenile bird had no apparent injuries, but was either cold and wet or showing signs of decomposition from heat.

[h] Mortalities were immobile juveniles were accidentally stepped on.

[i] Mortalities associated with events where bands or the transmitter antennae became entangled in vegetation and the juvenile was unable to move.

Table 4. Cause-specific mortality for five factors during the post-fledging period for juvenile dickcissels (DICK) and eastern meadowlarks (EAME) in southwestern Missouri, 2002 to 2004. Daily and interval mortality rates for each cause are presented with 95% confidence intervals. See Table 3 for descriptions of each mortality category.

Species	Cause	Daily	LCL, UCL	Interval	LCL, UCL
DICK	predation	0.039	0.032, 0.046	0.517	0.451, 0.582
	snakes	0.017	0/012, 0.022	0.229	0.174, 0.284
	mammals	0.005	0.002, 0.007	0.067	0.034, 0.100
	natural death	0.007	0.004, 0.011	0.103	0.063, 0.143
	weather	0.005	0.002, 0.008	0.071	0.037, 0.105
EAME	predation	0.018	0.013, 0.023	0.490	0.398,0.582
	snakes	0.005	0.002, 0.008	0.145	0.079, 0.211
	mammals	0.003	0.001, 0.006	0.099	0.044, 0.155
	natural death	0.004	0.001, 0.006	0.109	0.050, 0.167
	weather	0.003	0.001, 0.005	0.081	0.030, 0.132

VITA

Kimberly Marie Suedkamp Wells was born on 4 September 1975 to Nancy and Robert Suedkamp in Springerville, Arizona. She graduated from Taos High School in Taos, New Mexico in 1993. She received a Bachelor of Science in Renewable Natural Resources with an option in Wildlife Ecology from the University of Arizona in 1998. She proceeded to complete her Master of Science in Fisheries and Wildlife Ecology at Oklahoma State University in December of 2000. The following year, she entered the doctoral program in Fisheries and Wildlife Sciences at the University of Missouri. In addition to her doctoral program, Kim pursued the Certificate of Conservation Biology and the Graduate Minor in College Teaching.

Kim met Jason Blayne Wells while in graduate school at Oklahoma State University and they were married on 25 March 2001. Together, they have three dogs, Sasabe, Kayla, and Logan that keep them sane.