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Research Paper

Nestling condition of a grassland bird is not associated with food availability in restored grasslands

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ABSTRACT. Grassland bird populations have experienced steep declines in recent decades, necessitating better understanding of factors affecting their reproductive success. Grasslands are highly variable environments, and such variation affects the diversity and abundance of arthropods, which constitute the diet for most nestling grassland songbirds. Changes in arthropod abundance might affect parental food provisioning to nestlings and, consequently, nestling condition and survival. During the summers of 2017–2019, we examined the condition of Dickcissel (*Spiza americana*) nestlings from 288 nests in relation to biomass of arthropod prey across 36 restored grassland sites in Kansas that varied in vegetative management. Orthopteran (principal food for nestling Dickcissels) and total arthropod biomass were not related to cattle grazing or plant diversity in the initial seeding mix. Neither Dickcissel clutch size nor maximum brood size (including Brown-headed Cowbirds, *Molothrus ater*) were correlated with arthropod biomass measures in all years, which indicated that forage availability varied independently of clutch size. Neither age-corrected mass, mass/tarsus residuals, variation of tarsus length within broods, or plasma triglyceride concentration showed clear relationships with field-level variation in either arthropod biomass measure. This might be due to parental compensation for variable prey abundance. Brood size (including cowbirds) explained some variation in nestling condition with nestlings in larger broods generally exhibiting poorer condition (lower weight vs. structural size) than those in smaller broods. Thus, parents may be more limited in their capacity to feed all nestlings in large broods rather than limited by the availability of food within habitat patches. Consistent with previous hypotheses discounting food limitation to birds nesting in grasslands, our results suggest that Dickcissel nestling condition, known to affect post-fledging survival, might not be affected by spatial variation in food availability, at least in some years.

La condition de nidification d'un oiseau de prairie n'est pas associée à la disponibilité de la nourriture dans les prairies restaurées

RÉSUMÉ. Les populations d'oiseaux des prairies ont connu un déclin important au cours des dernières décennies. Nous devons donc améliorer notre compréhension des facteurs qui affectent leur réussite reproductive. Les prairies présentent un environnement très variable. Or, ces variations affectent la diversité et l'abondance des arthropodes, qui constituent le régime alimentaire de la plupart des oiseaux chanteurs des prairies au stade d'oisillon. Les variations dans l'abondance des arthropodes pourraient affecter l'apport en nourriture des parents aux oisillons et, par conséquent, la condition et la survie de ces derniers. Au cours des étés 2017-2019, nous avons examiné la condition des oisillons du Dickcissel d'Amérique (*Spiza americana*) dans 288 nids en relation avec la biomasse des arthropodes proies sur 36 sites de prairies restaurées au Kansas présentant des variations en termes de gestion de la végétation. La biomasse d'orthoptères (nourriture principale des oisillons) et la biomasse totale d'arthropodes n'étaient pas liées au pâturage du bétail ni à la diversité des plantes dans le mélange initial de semis. Ni la taille des pontes du Dickcissel d'Amérique, ni la taille maximale des couvées (y compris celles du Vacher à tête brune, *Molothrus ater*) n'étaient corrélées avec les mesures de la biomasse d'arthropodes au cours de toutes les années. Ainsi, la disponibilité du fourrage a varié indépendamment de la taille des pontes. Ni la masse corrigée en fonction de l'âge, ni le rapport masse/tarses, ni la variation dans la longueur des tarses au sein des couvées, ni la concentration en triglycérides plasmatiques n'ont montré de relation claire avec les variations des différentes mesures de biomasse d'arthropodes au niveau des sites. Ceci pourrait traduire une compensation parentale de la variation d'abondance des proies. En revanche, la taille des couvées (y compris celles des vachers), était liée à certaines variations dans la condition des oisillons — les oisillons des couvées plus grandes étant généralement en moins bonne condition (poids inférieur par rapport à la taille de la structure) que ceux des couvées plus petites. Ainsi, les parents sont davantage limités par leur capacité à nourrir tous les oisillons des grandes couvées que par la disponibilité de la nourriture dans les parcelles d'habitat. En accord avec les hypothèses précédentes qui ne tiennent pas compte de la limitation de la nourriture pour les oiseaux nichant dans les prairies, nos résultats suggèrent que la condition des oisillons de Dickcissel d'Amérique, dont on sait qu'elle affecte la survie après l'envol, ne serait pas impactée par les variations spatiales de la disponibilité de la nourriture — tout du moins, certaines années.

Key Words: *body condition; CRP; Dickcissel; food limitation; grassland birds; nestling condition; sibling competition; Spiza americana*

INTRODUCTION

Food availability is generally thought to contribute substantially to reproductive success in birds (Martin 1987). Food can be limiting at any reproductive stage, including egg formation, incubation, nestling, and/or post-fledgling survival. However, in altricial species the greatest demand for food generally occurs during the nestling and fledgling stage when parents must meet the food requirements of their offspring as well as themselves (Ricklefs 1968, Martin 1987). Manipulative experiments, both food supplementation and reduction, have shown that food can limit both present and future reproductive output through consequences to nestling survival, post-fledgling survival, and second nesting attempts by parents (Simons and Martin 1990, Rodenhouse and Holmes 1992).

The ability of altricial birds to produce offspring of physiological condition sufficient for survival depends largely on the quantity and quality of food provided by parents (Balshine 2012), which can be limited by food availability to parents in breeding habitats. Higher arthropod abundance might benefit parental provisioning of nestling insectivorous birds and, consequently, result in improved physiological condition of nestlings and offspring survival. Young fed more often or with larger food items tend to weigh more (Simons and Martin 1990), have higher nestling growth rates (Blancher and Robertson 1987), and are in better condition at fledging (Wilkin et al. 2009). Young that fledge in better condition have higher post-fledgling survival (Vitz and Rodewald 2011, Jones et al. 2017), especially late in the nesting season (Naef-Daenzer et al. 2001), which suggests that condition during development carries over to later life stages. Nestlings have been found to be in better condition when reared in habitats with greater food availability (Bańbura et al. 2011, Herring et al. 2011, but see Adams et al. 1994, Zalik and Strong 2008). Consequently, food availability likely affects nesting habitat selection in birds (Cody 1981, Johnson and Sherry 2001).

Although food is limiting for breeding birds in many ecosystems (Pérez et al. 2016, Kaliński et al. 2017), some authors have argued that food is superabundant for birds in grasslands, at least during the breeding season (typically coincident with growing seasons) because of grassland bird communities and populations being largely limited by density independent factors of climate severity and instability (e.g., precipitation, temperature; Wiens 1974, 1977; Wiens and Rotenberry 1979; Zimmerman 1992; but see Hamer et al. 2006). Although competition-mediated food limitation might not explain low avifaunal diversity within grasslands, food may still limit reproductive output in some years (Zimmerman 1992). Grassland birds also have measurable effects on insect prey populations, which supports the idea of the potential for food limitation for this avian guild (Joern 1992). Grasslands are dynamic environments, in which periodic drought, fire, and herbivory by large mammals interact to shape and maintain these ecosystems (Anderson 2006). Precipitation can be highly variable within grasslands, especially during the summer months (Borchert 1950). Coincident with inter-annual fluctuations in precipitation is variation in aboveground net primary production in grasslands (Yang et al. 2008). As vegetation provides habitat and food for arthropods, overall primary production can influence the diversity and abundance of arthropods (Joern and Laws 2013), which represent the main food source of many grassland birds during the breeding season. Thus, variation in arthropod food availability may be an important factor impacting

survival and reproductive success in grassland birds, whose populations have experienced declines in recent decades (Rosenberg et al. 2019). Concurrent with grassland bird population declines have been population declines in grasshoppers and other herbivorous arthropods (Welti et al. 2020) that serve as important prey for grassland birds (Fowler et al. 1991, Bock et al. 1992), including their nestling offspring (Kaspari and Joern 1993). Effects of variation in food availability on grassland bird populations therefore warrants further research, in addition to the historic focus on changes in habitat availability (Samson and Knopf 1994, Vickery and Herkert 2001).

Few studies have tested the effects of variable food availability on nestling condition in grassland birds (Zalik and Strong 2008, van Vliet 2017) even though grassland habitat quality is spatially and temporally heterogeneous. For example, tallgrass prairie grazed by bison (*Bison bison*) has greater orthopteran densities than ungrazed prairie (Joern 2004), and orthopterans are common forage for grassland birds during the breeding season (Joern 1986). Additionally, the diversity and abundance of plants (Jog et al. 2006) and insects (McIntyre and Thompson 2003) are higher in native prairie compared to recently restored grasslands, which might affect forage for grassland birds. The Conservation Reserve Program (CRP), the most extensive federally subsidized habitat restoration program in the U.S., has provided important habitat for grassland birds. Populations of many species have increased in otherwise cropland-dominated landscapes following increasing enrollment of land in the program (Herkert 1997, 1998, 2007). Although studies have investigated avian reproductive success within CRP fields (Granfors et al. 1996, McCoy et al. 1999, Kraus et al. 2022), to our knowledge, none have investigated variability in nestling condition of insectivorous, altricial grassland birds in relation to availability of arthropods as food in these habitats. We expected that insect communities would vary markedly among CRP fields, depending on management activities, initial seeding mixes, or time since planting because of the resulting differences in vegetation composition and structure. We investigated nestling condition in the Dickcissel (*Spiza americana*), a common grassland bird species in mid-continental North America (Sousa et al. 2022). Our objective was to compare nestling condition to food availability, the latter possibly being mediated by grassland management, across 36 CRP grassland field sites over three breeding seasons. We compared arthropod biomass among CRP fields that were grazed by cattle versus ungrazed and under different planting practices (low vs. higher diversity of plant seed mix), as these management factors might affect food availability for Dickcissels and other grassland birds. We predicted that CRP fields with higher arthropod biomass would constitute better nesting habitat and result in higher nestling mass and structural growth metrics, reduced within-brood size variation, and higher plasma triglycerides levels, which serve as a dynamic index of energetic state and recent consumption (Guglielmo et al. 2002).

METHODS

Study area and site selection

We conducted our research in 36 CRP fields in central Kansas centered around the community of Great Bend (38.3645°N, 98.7648°W) and spanning 80.5 km of a longitudinal precipitation gradient and roughly 40.2 km of latitude (Watson et al. 2021). The precipitation gradient of the region increases from west to

east, spanning a 1981 to 2010 average from 56 cm to 76 cm (Arguez et al. 2010, Watson et al. 2021). Conservation Reserve Program fields are abundant across the study area (13.3% of landscape), with the majority (67.5%) of CRP acreage enrolled in either conservation practice 2 (CP2; establishment of permanent native grasses) or 25 (CP25; rare and declining habitat). Fields enrolled in CP2 are established with a low diversity seed mixture of predominantly grasses, while CP25 fields use a high diversity seed mixture, containing native grasses and a minimum 10 forb species (Watson et al. 2024). Our research sites were CP2 and CP25 fields ($n = 18$ each) and comprised a portion of the 108 sites included in a larger, statewide study on the impact of grazing on plants, insects, and birds (Kraus et al. 2022, Waite et al. 2022, Wilson et al. 2022, Watson et al. 2024). Little bluestem (*Schizachyrium scoparium*), big bluestem (*Andropogon gerardi*), and sideoats grama (*Bouteloua curtipendula*) were the dominant grasses in both types of restoration plantings. However, these planting types (CP2 vs. CP 25) were found not to differ in non-native species, realized plant community composition (despite differences in seeds planted), or floristic quality indices in the central region of Kansas where our study on nestlings occurred (Watson et al. 2024). Time since planting varied among CP2 and CP25 planting types with the latter having been planted on average approximately 11 years later (Kraus et al. 2022), likely because of later availability of this conservation practice. We initially expected more arthropods on CP25 sites because of the intent of increased plant diversity versus CP2 plantings (McIntyre and Thompson 2003), hence the retention of this factor in our analyses.

A grazing treatment (ungrazed or season-long grazing) by domestic cattle (*Bos taurus*) was crossed with the CPs in a factorial design resulting in four treatments. Nine of the 18 CP25 fields and eight of the 18 CP2 fields were grazed. Grazing was authorized through an experimental allowance from the U.S. Department of Agriculture (USDA) and the Natural Resources Conservation Service set stocking rates, which targeted a 50% reduction in standing plant biomass during the grazing season based on field size and dominant vegetation, cattle type (cow-calf, yearling steers, etc.), and duration of grazing (120 to 180 days between 1 April to 31 October). Prescribed stocking densities varied to meet biomass reduction according to 528 Prescribed Grazing (NRCS 2025). Grazing occurred the first two years of the study (2017–2018) to represent an experimental mid-CRP-contract management option, but no grazing occurred in 2019 to test for any residual effects of grazing one-year post-disturbance. We did not explicitly incorporate fire regime as a covariate because burning depended on landowner preference and few fields were burned with only 5, 1, and 2 fields burned in 2017, 2018, and 2019, respectively. Burning was not correlated with grazing treatment (Kraus et al. 2022).

To distribute the four treatments across the precipitation gradient, we divided the study area into three longitudinal zones (west, central, and east), within which approximately a third of the replicates per treatment ($n = 3$) were established via selection from a randomly ordered list of fields in a USDA database. We considered fields eligible for inclusion in the study if they had been seeded and established for at least five growing seasons prior to initial data collection in 2017, remained in CRP throughout the duration of the study, were at least 14.2 ha in size, and had a maximum perimeter:area ratio of 0.18 to ensure that minimum

area requirements for several grassland bird species were met (Helzer and Jelinski 1999, Winter and Faaborg 1999, Johnson and Igl 2001). Landowners of suitable fields were contacted to discuss the possibility of allowing access to their property and implementing cattle grazing; thus, inclusion in our study was based on landowner willingness to participate. After identifying grazing participants, we generated a list of potential ungrazed fields by selecting fields enrolled under the same CP within a 8.05-km radius of the grazed site but ≥ 1 -km distant so as to pair similar yet independent fields (and all fields, grazed and ungrazed, were ≥ 1 -km apart). If no field was located within that distance, we expanded the search area by 8.05-km increments until a landowner with a matching CP field agreed to allow access. On selected fields, we established 200-m x 300-m plots that were positioned within the largest interior portion of the field and oriented parallel to the longest axis of the field (Wilson et al. 2022). These plots were used as the focal areas for nest searches and prey abundance surveys.

Nest searching and monitoring

We searched for nests from mid-May to late-July for three breeding seasons (2017–2019). We systematically searched mostly within plots and opportunistically outside plots (e.g., en route to or from technician vehicles at roadsides) for approximately 2 hours per site visit. Sites were visited by three field crews, with each crew visiting the same 12 sites twice per week and four sites per day. We searched for and monitored nests on the first two sites visited each day, beginning at 06:00. Only nest monitoring occurred on the latter two sites visited. When logistically feasible, we rotated site visitation order throughout the season to vary timing of nest searching and monitoring, but similar nest search effort was attempted for all sites.

We located Dickcissel nests through rope dragging, which induces female birds to flush from nests. Two members of each crew towed a rope (2-cm diameter, 15-m length) perpendicular to the path of travel, gliding the rope atop the vegetative canopy, with two members following behind to monitor for flushing females. When a bird flushed at or near the rope, one crew member surveyed the flush location for a potential nest. The starting location for rope dragging was rotated each visit to maximize search area covered. Nests found incidentally via flushing incubating females while walking through sites, or found via parental behavior (carrying building material, food, alarm calls), were also included. We recorded the geographic coordinates of nests using a Global Positioning System unit (GPS; Garmin eTrex, Garmin Ltd., Olathe, KS) and tied two blue vinyl flags at a 90° arc from one another to vegetation 5-m from the nest. Wooden garden stakes were placed below flagging on all sites as a redundant marker in case flags were removed by cattle on grazed sites. We monitored Dickcissel nests every 3–4 days until young fledged or the nesting attempt failed.

Nestling condition indices

We determined nestling age from nest visitation history and nestling characteristics (e.g., feather tract development, eyes open/closed). Between 4 and 6 days post-hatch, we measured nestling mass (g) using a digital scale (0.01g) and tarsus (mm) and wing chord (mm) lengths using dial calipers. From these measurements, we calculated three morphological indicators of condition (Labocha and Hays 2012). One measure, nestling mass

corrected to a standardized age, allowed comparison between nestlings weighed at different ages. The standardized mass was calculated using a regression equation ($\text{mass} = 1.83 \times \text{age} + 2.34$, generated from our data) of nestling mass (dependent variable) against age for all nestlings weighed during the study. To adjust the observed values for individual nestlings to a standardized age, residuals between the observed and predicted mass on the day of measurement were added to, or subtracted from, the predicted mass at day 6 post-hatch (13.32 g). We justify this standardization based on observed linear growth curves of Dickcissel nestlings in Kansas (Winnicki 2019). Measurements were only regularly made on Dickcissel nestlings (and rarely on cowbird nestlings) for logistical reasons and to minimize disturbance at nest sites.

The second condition index we used was the residuals of nestling mass vs. tarsus length. Nestlings that were heavier relative to their size at any age were assumed to be in better condition. Data from nestlings at all ages were used to develop the regression equation ($\text{mass} = 0.74 \times \text{tarsus} - 1.81$). From the regression model we determined the difference between the observed and expected mass at a given tarsus length, with a residual > 0 representing a mass above that predicted for body size (i.e., good condition) and a residual < 0 signifying a mass below that expected for its body size (i.e., poor condition).

The final morphological condition index we used was variation in tarsus length within broods. Greater variation in this metric might be expected under greater nutritional stress (i.e., lower prey availability, parasitism; Merino and Potti 1995). To do this, we calculated the standard deviation of tarsus lengths among Dickcissel nestlings in each nest. Nests with only one nestling were excluded from this index.

Blood sampling and processing

Triglycerides (TRIG), a plasma metabolite, increase in concentration during food adsorption, transporting fat through the bloodstream to adipose tissue and working organs (Jenni-Eiermann and Jenni 1994). Thus, TRIG represents a measure of fat deposition and positive energy balance with high values reflecting high rates of parental feeding. We collected blood samples, when measuring morphometrics (day 4–6) from a maximum of two Dickcissel nestlings per nest. We collected 70 μl of whole blood from the brachial vein of nestlings, storing samples in heparinized capillary tubes on ice until they could be centrifuged later the same day. We transferred plasma to microcentrifuge tubes, stored them at -20°C for up to ~ 14 days, then stored samples at -80°C prior to analysis.

We assayed TRIG on an Eon microplate spectrophotometer (Biotek Instruments, Inc., Winooski, VT). TRIG concentration (total triglyceride - free triglyceride) was measured in duplicate in 400 μl flat-bottom, 96-well polystyrene microplates using colorimetric endpoint assays. We diluted plasma 1:1 with saline. In 2017, some samples had insufficient plasma to run in duplicate. When this occurred, we combined samples from nestlings on the same field to obtain the average TRIG; in three cases, we pooled samples from two nestlings, and in one case, we pooled samples from three nestlings. To each 5 μl sample we added 240 μl free glycerol reagent (Sigma F6428, MilliporeSigma, Darmstadt, Germany), which reacts with glycerol in plasma. We warmed samples for 10 min at 37°C , after which we measured absorbance at 540 and 750 nm. Then we added 60 μl triglyceride reagent

(Sigma T2449, MilliporeSigma, Darmstadt, Germany), which breaks apart ester bonds in triglycerides. We again incubated samples for 10 min at 37°C and measured absorbance at 540 and 750 nm. We calculated TRIG concentration based on standard curves obtained from a serial dilution of glycerol standard (Sigma G7793, MilliporeSigma, Darmstadt, Germany). We averaged estimated TRIG concentration from the same sample in duplicate wells and calculated the coefficient of variation (CV) of the mean concentration. Any estimates with $\text{CV} \geq 25\%$ were excluded from analysis ($n = 24$ of 407 nestlings). When TRIG values exceeded 5.64 mM (the limits of our standard curve), we further diluted samples and re-ran the assay (when enough sample remained) or removed samples from further analysis. To our knowledge, no study has investigated plasma metabolites of passerine nestlings in relation to prey abundance in grassland birds.

Prey abundance surveys

We measured the abundance of insect prey at each field using four 10-m sweep-net transects, with individual transects extending 50 m from the center of each 200×300 m plot in the direction of plot corners and separated by 90° (Fig. A6.1). Observers sampled each field twice during the field season, approximately 5 weeks between visits (22–57 days between samples on same sites), with the first sampling period occurring between 23 May and 22 June and second sampling period occurring between 24 June and 25 July each year (2017–2019). Between 07:30 and 16:00, observers used 38-cm diameter canvas sweep-nets to target the top 25% of vegetation in a 180° arc while walking each 10-meter transect. All sweep-netting was done into the wind, with maximum wind speed not exceeding 36 kmh. Generally, sweep-netting occurred on days with clear to mostly clear skies, but given logistical constraints some sweep-netting occurred on mostly cloudy (14.4%) and overcast (23.6%) days. Sweep-netting was synchronized with observer's walking pace, so that one sweep was taken with each step. Contents of sweep-nets were emptied into containers and stored in a freezer until they could be sorted, identified, dried, and weighed. We did not sort prey into size classes to represent potential prey with varying energy supplies (Mitchell et al. 2012) but assume that overall biomass per field is proportionate to prey available to Dickcissels.

In all samples, we separated arthropods from debris, identified to order, dried for 72 h at 60°C , and weighed biomass to the nearest mg. Because orthopterans are the preferred food provision for Dickcissel nestlings (Mitchell et al. 2012), we used orthopteran biomass summed across sweep-net transects from both sampling periods as a measure of food availability for each field. As an additional measure of prey availability, we used total arthropod biomass summed for all arthropod taxa across sweep-net transects from both sampling periods.

Statistical analyses

To determine whether management practice affected orthopteran or total arthropod biomass, we ran generalized-linear mixed models with CP, grazing, and year as fixed effects and field as a random effect due to sampling on the same fields in multiple years. Year was included as a factor in all analyses. In addition to single, additive effects and constant (intercept-only null) models, we considered the potential interaction between grazing and CP, grazing and year, and grazing and year with CP as an additive effect.

Given the potential for a positive effect of food availability on clutch size, and thus brood size and the resulting demand on parental effort (Ricklefs 1968), we tested for the effects of field-level orthopteran and total arthropod biomass on Dickcissel clutch size and maximum brood size, the latter including Brown-headed Cowbird (*Molothrus ater*, cowbird hereafter) young. If brood size were proportionate to food availability, we might not expect an effect of variable food availability on nestling condition. Because cowbirds sometimes (but not always) remove host eggs, brood parasitism obscures clutch size estimates. Thus, we examined arthropod biomass effects on clutch sizes in nests found during incubation and not containing cowbird eggs to understand how food availability to female Dickcissels might affect clutch size. Here we modeled mean clutch size per field in relation to field-level orthopteran biomass, total arthropod biomass, and the interaction between measures of arthropod biomass and year, with field included as a random effect. To more directly examine the demands imposed by offspring number in relation to food availability, we also included all nestlings (host and cowbird) in comparing mean maximum brood size to the arthropod biomass measures using the same candidate model structures as used for clutch size.

We assessed nestling condition in relation to field-level arthropod biomass measures in two ways: (1) using average nestling condition indices per field, and (2) using nestling-level metrics while accounting for nest-level covariates. For the first approach, we obtained average condition measures for a field by calculating the mean of each of the condition indices (age-corrected mass, mass/tarsus residuals, variation in tarsus length, and TRIG) per nest, then per field each year (means per nest were calculated to dampen the effect of potentially larger broods on field-level means). We evaluated field-level condition indices in relation to orthopteran biomass, total arthropod biomass, and the interaction between measures of arthropod biomass and year, with field included as a random effect to account for sampling on the same field in multiple years.

To assess condition indices at the nestling level, we ran generalized-linear mixed models, using all possible additive combinations of food availability and brood characteristics, including the latter because brood size can affect nestling condition though intra-brood competition (Ricklefs 1982). For brood characteristics, we included brood size at time of sampling, including nestling cowbirds, maximum brood size during the nestling period, categorical parasitism by cowbirds (containing cowbird nestlings or not), and maximum number of cowbird nestlings during the nestling period as covariates. Candidate models did not include maximum number of cowbird nestlings in tandem with either categorical parasitism or maximum brood size. Categorical parasitism was not strongly associated with brood size at time of sampling or maximum brood size (Program R, *glm* logistic regression $P = 0.481$ and 0.069 , respectively) and brood size at time of sampling and maximum number of cowbird nestlings were also not associated (package *glm* Poisson regression $P = 0.254$); thus, these predictor variables were allowed in tandem within the same models. We created additional model sets with interactions between measures of arthropod biomass and year. Because lower arthropod availability also might be especially taxing for nestlings in nests with large broods, we also tested for interactions between arthropod biomass measures and

brood size at the time of sampling and also interactions with maximum brood size. We included nest within field as a random effect for age-corrected mass, mass/tarsus residuals and TRIG. We also compared variation in tarsus length (SD) at the nest level to the predictors used in nestling condition analyses; here we included field as a random effect.

We performed all analyses in the R programming language (ver. 3.6.1; R Core Team 2019) using the package *lme4* (Bates et al. 2015) and functions *lme* and *lmer*. Using an information theoretic approach (Burnham and Anderson 2002), we evaluated and ranked models using AIC_c . Among competitive models ($\Delta AIC_c \leq 4$; Burnham et al. 2011), we accounted for model uncertainty in parameter effects by using conditional model averaging among competitive models ($\Delta AIC_c \leq 4$) and assessed parameter effects using 85% confidence intervals (Arnold 2010).

Because preliminary analysis revealed that response variables were not normally distributed (using Shapiro-Wilk test), we transformed variables using package *rcompanion* (Mangiafico 2019), which used Tukey's Ladder of Powers to determine the best transformation to obtain normality using Shapiro-Wilk normality tests (Tukey 1977; see Appendix 6, including Figs. A6.1–A6.6). Subsequently, we log-transformed total arthropod biomass and orthopteran biomass, as responses to management treatments. We transformed field-level means of age-corrected mass using $-1/x^{1.5}$ but no suitable transformation was available for mass-tarsus residuals (*rcompanion* returned as x^1 with values still not normally distributed; $W = 0.960$, $P = 0.016$). We transformed field-level means of SD tarsus and TRIG using $x^{0.5}$ and $\log(x)$, respectively. We transformed field-level means of clutch size as a response variable using $x^{1.5}$. We transformed field-level means of maximum brood size as a response variable using $x^{1.5}$. We transformed individual condition indices of age-corrected mass and TRIG using $x^{1.5}$ and $\log(x)$, respectively. No suitable transformation was available for individual mass-tarsus residuals (*rcompanion* returned as x^1 with values still not normally distributed; $W = 0.975$, $P < 0.001$). We transformed nest-level SD in tarsus length using $x^{0.5}$. We did not transform numerical variables when used as predictor variable for analyses of individual nestling condition.

RESULTS

We found 1030 Dickcissel nests, of which offspring in 288 nests survived to 4–6 days at which time we measured condition. From 6 June to 27 July 2017–2019, we collected morphometrics from 756 nestlings and collected blood samples from 407 nestlings. Among-field variation in orthopteran and total arthropod biomass in sweep-net samples ranged from 0.007 to 3.405 g (mean = 0.370 ± 0.042 SE) and 0.086 to 3.601 g (mean = 0.678 ± 0.048 SE), respectively. Orthopterans accounted for 38.5, 37.1, and 29.3% of total arthropod biomass in 2017, 2018, and 2019, respectively, and explained 57% of the variation in total arthropod biomass (simple linear regression $r^2 = 0.57$). An orthopteran biomass of 3.405 g was a clear outlier at one site in 2017 (over 6.8 standard deviations above the mean; Figs. A7.1–A7.4). We conducted analyses with and without this site- and year-specific outlier, including normality tests, although the results were largely similar with few changes to top ranked models (AIC_c) when censoring the outlier (exceptions discussed below).

Effect of management practice on arthropod biomass

Variation in invertebrate biomass was largely unrelated to the temporal or management variables we considered. The constant (intercept-only) model best explained variation in orthopteran biomass and total arthropod biomass (Appendix 1). Grazing and CP explained some variation in orthopteran and total arthropod biomass ($\Delta AIC_c \leq 4$; Appendix 1), but the model-averaged 85% CIs for these predictors included 0, indicating negligible effects (Table 1). Overall total arthropod biomass was slightly lower in grazed compared to ungrazed fields, but the grazing model had nearly 5 times less support (w_i) than the constant model (Appendix 1) and should be considered uninformative (Arnold 2010). Orthopteran biomass was lower in 2019 than 2017 (85% CIs did not overlap 0; Table 1), but the year predictor can also be considered uninformative relative to the constant model (Appendix 1; Arnold 2010). Because orthopteran biomass and total arthropod biomass did not differ among management practices, we did not include management practices in analyses of nestling condition indices.

Table 1. Parameter estimates (β) and 85% confidence intervals averaged across competitive models ($\Delta AIC_c \leq 4$) for measures of orthopteran and total arthropod biomass in relation to management practices (grazed vs. ungrazed by cattle; CP2 vs. CP25) in Conservation Reserve Program fields in central Kansas, USA, 2017–2019. Negative β for categorical predictors indicate that measures of arthropod biomass in, e.g., grazed (Y), were lower than ungrazed (N), lower in CP25 vs. CP2, etc. Parameter estimates for years (2018, 2019) are relative to responses in 2017.

Parameter	Orthopteran biomass		Total arthropod biomass	
Grazed (Y vs. N)	0.060	(-0.345, 0.466)	-0.149	(-0.397, 0.099)
CP (25 vs. 2)	0.215	(-0.186, 0.616)	0.106	(-0.143, 0.355)
Year 2018	-0.280	(-0.596, 0.036)		
Year 2019	-0.471	(-0.787, -0.155)		

Clutch and brood sizes

Clutch size did not vary significantly with arthropod biomass measures. The constant model for field-level clutch sizes (Appendix 2) was ranked highest among the candidate sets including arthropod biomass predictors (mean clutch size = 4.001 ± 0.069 SE in unparasitized nests). Models beyond the constant model of clutch size were uninformative (Arnold 2010) and all the 85% CIs of parameter estimates overlapped 0 (Table 2). The top-ranked models for field-level maximum brood size (Dickcissel + cowbirds, mean = 3.691 ± 0.059 SE) included orthopteran and total arthropod biomass interacting with year (Appendix 3), where there was a negative relationship between maximum brood size and orthopteran biomass in 2018 and between maximum brood size and total arthropod biomass in 2019 (model-averaged $\beta \pm 85\%$ CI: -4.663, -0.112 and -4.234, -0.313, respectively). Thus, there was no apparent increase in clutch or brood sizes with increased field-level invertebrate biomass.

Field-level nestling condition

Neither orthopteran biomass nor total arthropod biomass were important predictors of field-level means of age-corrected mass residuals (across nestling mean = 13.317 ± 0.081 SE), mass/tarsus

Table 2. Parameter estimates (β) and 85% confidence intervals averaged across competitive models ($\Delta AIC_c \leq 4$) of mean, field-level Dickcissel (*Spiza americana*) clutch size in relation to orthopteran and total arthropod biomass in Conservation Reserve Program fields in central Kansas, USA, 2017–2019. Parameter estimates for years (2018, 2019) are relative to responses in 2017.

Parameter	β (85% CI)	
Arthropod biomass	0.493	(-0.084, 1.070)
Orthopteran biomass	0.464	(-0.180, 1.108)
Orthopteran biomass * Year (2018)	-0.025	(-1.508, 1.458)
Orthopteran biomass * Year (2019)	0.206	(-1.467, 1.8793)
Arthropod biomass * Year (2018)	-0.158	(-1.371, 1.057)
Arthropod biomass * Year (2019)	0.199	(-0.955, 1.120)

residuals (across nestling mean = -0.003 ± 0.064 SE), standard deviation of tarsus (across nests = 1.695 ± 0.066 SE), or TRIG concentrations (across nestling mean = $2.132 \mu\text{l} \pm 0.062$ SE); for all condition-related indices, the constant model was the most competitive model (Appendix 4). Models including orthopteran biomass and total arthropod biomass were competitive ($\Delta AIC_c \leq 4$) in explaining average mass/tarsus residuals and mean standard deviation of tarsus at the field level of nestlings on fields. However, these were uninformative models (Arnold 2010), they had approximately half the support of the constant model (Appendix 4), and model-averaged 85% CI of parameter estimates for mass-tarsus residuals overlapped zero (model-averaged orthopteran biomass: $\beta \pm 85\%$ CI = -0.740, 0.158; total arthropod biomass: $\beta \pm 85\%$ CI = -0.653, 0.137). Additionally, little variation in mass/tarsus residuals was explained by orthopteran biomass ($r^2 = 0.011$) or total arthropod biomass ($r^2 = 0.012$). Orthopteran biomass and total arthropod biomass were similarly competitive ($\Delta AIC_c \leq 4$; $w_i = 0.240$ and 0.220 , respectively; Appendix 4) in explaining variation in mean standard deviation of tarsus at the field level and had slightly positive effects (orthopteran biomass: model-averaged $\beta \pm 85\%$ CI = 0.042, 0.303; total arthropod biomass: $\beta \pm 85\%$ CI = 0.040, 0.270), but these can be considered uninformative models (Arnold 2010) and explained little variation in the response ($r^2 = 0.051$ and 0.053 , respectively). Orthopteran biomass was in the competitive set ($\Delta AIC_c \leq 4$) of mean TRIG concentration at the field level (Appendix 4), but the constant model had roughly 5 times the support of this model and the orthopteran biomass model's model-averaged 85% CI of parameter estimates overlapped zero (-0.290, 0.026) with $r^2 = 0.020$.

Nestling-level condition

Models containing interactions between orthopteran biomass and year, with additive effects of maximum brood size and brood size, best explained variation in age-corrected mass residuals (Appendix 5), but the model-averaged 85% CI for all effects in these models overlapped zero (Table 3) and fixed effects in both models explained little variation ($r^2 = 0.024$ for both models). Models with total arthropod biomass and year, also with additive effects of maximum brood size and brood size, and arthropod biomass interactions with brood and maximum brood sizes were weaker models (Appendix 5) with similarly weak effects (Table 3). The best supported model for mass/tarsus residuals of nestlings contained brood size and had 4.9 times the support of

a less parsimonious model including brood size and cowbird parasitism (Appendix 5). Brood size alone explained little variation in mass/tarsus residuals (marginal $r^2 = 0.046$). Mass/tarsus residuals related negatively to brood size and positively to cowbird parasitism, though the effect of the latter was weak given the $\beta \pm 85\%$ CI overlapped zero (Table 3). Within-brood variation in tarsus length was best explained by orthopteran biomass and brood size (Appendix 5), this additive model receiving only 1.25 times the model weight than the next best model that included brood size only and the $\beta \pm 85\%$ CIs for all invertebrate biomass measures overlapped zero (Table 3). Brood size was an important predictor with nestlings within larger broods having greater variation in tarsus length than in smaller broods (Table 3). Brood size was also the only model predictor $\Delta AIC_c \leq 4$ when the site with the orthopteran biomass outlier in 2017 was removed from analysis of SD in tarsus. In general, however, brood size and all model predictors explained relatively little of the within-brood variation in tarsus length (marginal $r^2 = 0.092$ and conditional $r^2 = 0.177$, respectively). No models were any better than the constant model at explaining TRIG of Dickcissel nestlings, with all other models containing predictors having $\Delta AIC_c > 4$ (Appendix 5). When the site with the orthopteran biomass outlier in 2017 was removed, orthopteran and total arthropod biomass models were $\Delta AIC_c \leq 4$, but these models were weaker than the constant model and can be considered uninformative and their $\beta \pm 85\%$ CIs included zero (orthopteran biomass: model-averaged $\beta \pm 85\%$ CI = -0.061, 0.082; total arthropod biomass: model-averaged $\beta \pm 85\%$ CI = -0.064, 0.103).

DISCUSSION

Overall, Dickcissel nestling condition was unrelated to arthropod prey availability within CRP fields where they nested. Among-field variation in orthopteran and total arthropod biomass was unaffected by grassland management technique. Brood sizes were unrelated to arthropod biomass availability indicating potential for variation in demand vs. availability of food among nests. Field-level estimates of nestling condition were also unrelated to prey availability. At the scale of individual nests, brood size was the only significant predictor of nestling condition measures. Nestlings in larger broods tended to have lower mass relative to structural size (tarsus length) and greater variation in structural size (SD tarsus length). Thus, our results indicate that nestling condition is not as limited by the amount of food within current habitat patches but rather by variation in brood size.

We expected that grazing management of CRP fields might be a local driver of grasshopper populations (Joern 2004) and thus indirectly affect nestling condition of Dickcissels and perhaps other grassland birds. However, we found no such effect of cattle grazing on orthopteran or total arthropod biomass in CRP fields. Joern (2004) found a positive response of grasshoppers to over a decade of year-round bison grazing, which is considerably longer than the short-term grazing regime implemented in our study. It could be that densities of orthopterans or arthropods overall were unable to respond to short-term effects of grazing on plant community structure or composition in our study (Branson and Sword 2010). Similarly, there appeared to be no arthropod response to differing diversity of plant species seeded in CRP conservation practices. This is unsurprising given emergent plant communities in CP2 and CP25 plantings are similar in central Kansas (Watson et al. 2024).

Table 3. Parameter estimates (β) and 85% confidence intervals averaged across competitive models ($\Delta AIC_c \leq 4$) for individual Dickcissel (*Spiza americana*) nestling condition indices in relation to measures of orthopteran and total arthropod biomass and brood size in Conservation Reserve Program fields in central Kansas, USA, 2017–2019. Negative β for the categorical cowbird parasitism predictor indicate that nestling condition in parasitized nests (Y) was lower than un-parasitized nests (N). Parameter estimates for years (2018, 2019) are relative to responses in 2017.

Parameter	Model-averaged β (85% CI)	
Age-corrected mass:		
Arthropod biomass	-2.177	(-5.067, 3.203)
Year: 2018	1.498	(-1.588, 3.980)
Year: 2019	1.021	(-1.782, 3.412)
Brood size	-1.868	(-2.347, 0.636)
Arthropod biomass * Year: 2018	0.379	(-1.805, 2.019)
Arthropod biomass * Year: 2019	0.710	(-1.847, 2.248)
Max brood size	-2.025	(-2.751, 0.556)
Orthopteran biomass	-0.261	(-3.784, 3.485)
Orthopteran biomass * Year: 2018	0.253	(-3.311, 3.572)
Orthopteran biomass * Year: 2019	0.014	(-3.054, 3.068)
Arthropod biomass * brood size	1.676	(-0.577, 0.822)
Arthropod biomass * max brood size	2.035	(-0.712, 1.008)
Orthopteran biomass * max brood size	2.065	(-0.698, 0.929)
Mass/tarsus residuals:		
Brood size	-0.346	(-0.452, -0.239)
Parasitism (Y vs. N)	0.104	(-0.100, 0.127)
Orthopteran biomass	-0.101	(-0.124, 0.098)
Arthropod biomass	-0.068	(-0.092, 0.077)
SD in tarsus:		
Orthopteran biomass	0.175	(-0.065, 0.220)
Arthropod biomass	0.136	(-0.061, 0.115)
Brood size	0.132	(0.091, 0.173)

Contrary to prediction as well as many studies from other regions, food availability was unrelated to nestling condition as examined among nestlings at the field- (local population) or individual-level. Food supplementation experiments in a variety of other habitats have demonstrated positive changes to nestling body condition in the presence of additional food (Simons and Martin 1990, Bañura et al. 2011, Haley and Rosenberg 2013). Similarly, declines in nestling growth rates of Black-throated Blue Warblers (*Setophaga caerulea*) coincided with natural decline in caterpillar abundance (Rodenhouse and Holmes 1992). The influence of food availability on nutritional state of nestlings can also be seen in other un-manipulated breeding populations of wild birds. Nestling condition indices, such as TRIG and body mass, were higher in habitats with greater prey abundance (Kaliński et al 2017, Teglhøj 2017). However, the meta-analysis by Ruffino et al. (2014) found that positive effects of food supplementation on reproductive responses were less noticeable when natural availability of food resources was high. This was possibly the case for our study; our estimates arthropod abundance per sampling effort (1.08–45.01 mg/sweep; 80 sweep-net sweeps / field / year) exceeded the 0.13–6.05 mg/sweep range found by Hull et al. (1996) across CRP fields in northeastern Kansas. Parent birds might also compensate for reduced food availability through increased provisioning from larger home ranges. Reduction of grasshopper densities around nests of Vesper Sparrow (*Poocetes gramineus*) did not affect nestling

growth rate or mass before fledging but induced longer flights (ranged > 200 m) by provisioning parents during foraging bouts (Adams et al. 1994). This was also suspected for the Savannah Sparrow (*Passerculus sandwichensis*), a grassland songbird, where nestling mass for sparrows nesting in hay fields was unaffected by significant reductions in invertebrate biomass because of prior mowing (Zalik and Strong 2008). Such compensation via increased provisioning effort, however, might impose costs to parents via decreased adult survival (Dijkstra et al. 1990). Thus, although nestling condition might not respond to local food availability, demands and survival costs to parents might be indirectly affected. Similarly, if Dickcissel habitat selection and resulting spatial variation in density were proportionate to food availability (Romanowski and Zmihorski 2008), this might also explain why food was not limiting to nestlings across our field sites.

A complication in detecting effects of food availability on nestling condition is unexplained variation in nestling growth rates. Dickcissel nestling growth is highly variable, even within single study sites (Winnicki 2019), which could mask detection of environmental effects on nestling condition measures from single sampling events, as in our study. Such covariates of nestling growth could include genetic components (Merilä 1996) and variation in nest microclimate (Mueller et al. 2019).

Because sibling competition is thought to be a dominant force shaping growth rates in birds (Ricklefs 1982), competition among nest-mates for limited food resources delivered by parents might explain the observed negative pattern detected between condition indices of nestlings and brood size. For example, brood size manipulations have consistently demonstrated detrimental effects on nestling condition in enlarged broods, with nestlings in experimentally enlarged broods having lower body mass and mass/tarsus residuals (Neuenschwander et al. 2003, Bańbura et al. 2008, Losdat et al. 2010, Bourgeon et al. 2011). This occurs despite parental adjustment of food delivery to accommodate larger or smaller broods (Neuenschwander et al. 2003). Additionally, enlarged broods tended to have greater variation in body mass within nests (Losdat et al. 2010), perhaps similar to the greater variation in tarsus length we observed in larger broods. These results suggest that parents may be more limited in their capacity to adequately provide all nestlings in large broods with similar quantities of food rather than limited by the availability of food on the landscape.

Despite the large geographic extent of our sampling effort, lack of a detectable pattern between food availability and nestling condition could be due to arthropod samples that did not adequately represent local food availability for nestlings. Because of logistical limitations (e.g., the current study involved only 36 of 108 sites statewide where arthropods were sampled), we collected arthropod biomass on a limited area (40 m of sampling transects) within larger sample plots for nests within fields, and arthropods were sampled only twice during breeding seasons. Because prey abundance in grasslands fluctuates over time (Williams 2016), and potentially across space, the limited extent of our measures of prey availability might not have adequately represented food locally available to parent birds and their nestlings. Sampling closer to individual nests might have been more representative of food availability. However, prey

availability around Savannah Sparrow nests was not correlated with nestling condition (van Vliet 2017). Additionally, an ancillary study by one of our technicians (Whipple 2020) revealed that most (56%) of 100 foraging trips by parent Dickcissels observed around 22 nests on 4 of our CRP fields in 2019 were > 50 m from nests, suggesting that sampling near nests might not have been any more representative of food availability for Dickcissel nestlings than the approach we used in our study.

Overall, our research suggests site-level food availability is not limiting to Dickcissel nestlings in CRP fields during the breeding season and provides evidence consistent with the hypothesis that food is not limited during the breeding season for grassland birds (Wiens 1974). Such a hypothesis, however, doubtfully applies to the vast range of climate variability and disturbance regimes across the world's grasslands. Longer-term research might reveal that food is indeed limiting in some years under different environmental conditions (Pulliam and Dunning 1987). Drought years might negatively impact food availability for grassland bird populations (Wiens 1974, Zimmerman 1992), which should limit their reproduction as parents must increase search time and compete for scarce food resources. Species richness in grassland bird communities has been found to correlate positively with grasshopper species richness (Hamer et al. 2006), suggesting some food limitation to grassland bird community structure. However, recent evidence from Kansas tallgrass prairie suggests that many measures of grassland songbird reproduction are unaffected by drought and arthropods eaten by grassland songbirds during the breeding season are not reduced in drought years (Smith et al. 2024). Long-term declines in food availability or quality for grassland birds, or long-term changes in nestling condition cannot be inferred from our short-term study, but it is possible that these could coincide with observed declines in grasshopper populations (Welti et al. 2020) or nutrient dilution in those prey induced by increased atmospheric CO₂ (Kaspari and Welti 2024, Welti and Kaspari 2024). Given concerns regarding insect declines globally (Sánchez-Bayo and Wyckhuys 2019), and particularly in grasslands (Welti et al. 2020), further study is warranted on potential limitation of invertebrate food to consumers, such as grassland birds, and factors that might affect such limitations, such as grassland management, reduction in plant tissue nutrients, local and regional pesticide applications, and climate change.

Author Contributions:

W.E.J, H.M.K, G.R.H, M.L.J, and M.M.R. conceived the idea, design, experiment, and developed and designed the methods; H. M.K and W.E.J performed the experiments and analyzed the data; H.M.K, W.E.J, G.R.H, M.L.J, M.M.R, and W.A.B wrote the paper; W.A.B provided laboratory space and assisted with metabolite data interpretation.

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Data Availability:

The data that support the findings of this study are available here: <https://doi.org/10.5281/zenodo.14907921>.

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Appendix 1. Model selection results for measures of orthopteran and total arthropod biomass in relation to management practices (grazed vs. ungrazed by cattle; CP2 vs. CP25) in Conservation Reserve Program fields in central Kansas, USA, 2017-2019.

Model	Orthopteran biomass				Total arthropod biomass			
	Deviance	<i>K</i>	ΔAIC_c	w_i	Deviance	<i>K</i>	ΔAIC_c	w_i
Constant	324.57	3	0.00	0.501	210.50	3	0.000	0.669
CP	324.70	4	2.288	0.160	211.81	4	3.471	0.118
Year	322.59	5	2.380	0.152	211.90	5	5.760	0.038
Grazed	325.25	4	2.836	0.121	211.43	4	3.090	0.143
Grazed + CP	325.38	5	5.169	0.038	212.67	5	6.534	0.025
Grazed * CP	324.25	6	6.284	0.022	212.85	6	8.951	0.008
Grazed * Year	322.90	8	9.555	0.004	214.74	8	15.472	0.000
Grazed * Year + CP	323.03	9	12.070	0.001	215.99	9	19.098	0.000
Grazed * CP + Grazed * Year	321.90	10	13.372	0.001	216.16	10	21.703	0.000

Appendix 2. Model selection results for comparisons of mean (field-level) Dickcissel (*Spiza americana*) clutch size in relation to orthopteran and total arthropod biomass in Conservation Reserve Program fields in central Kansas, USA, 2017-2019.

Model	Deviance	K	ΔAIC_c	w_i
Constant	313.44	3	0.00	0.361
Arthropod biomass	311.94	4	0.72	0.252
Orthopteran biomass	312.20	4	0.98	0.221
Orthopteran biomass * Year	304.42	8	2.71	0.093
Arthropod biomass * Year	304.92	8	3.22	0.072

Appendix 3. Model selection results for comparisons of mean (field-level), maximum Dickcissel brood size (Dickcissel, *Spiza americana*, and Brown-headed Cowbird, *Molothrus ater*, nestlings) in relation to orthopteran and total arthropod biomass in Conservation Reserve Program fields in central Kansas, USA, 2017-2019.

Model	Deviance	K	ΔAIC_c	w_i
Orthopteran biomass * Year	342.34	8	0.00	0.376
Arthropod biomass * Year	342.34	8	0.00	0.375
Constant	356.67	3	2.57	0.104
Arthropod biomass	354.89	4	3.01	0.083
Orthopteran biomass	355.51	4	3.63	0.061

Appendix 4. Model selection results for mean, field-level condition indices of Dickcissel (*Spiza americana*) nestlings in relation to measures of orthopteran and total arthropod biomass in Conservation Reserve Program fields in central Kansas, USA, 2017-2019.

Model	<i>K</i>	Age-corrected			Mass/tarsus			SD tarsus			TRIG [†]		
		Dev.	ΔAIC_c	w_i	Dev.	ΔAIC_c	w_i	Dev.	ΔAIC_c	w_i	Dev.	ΔAIC_c	w_i
Constant	3	-660.84	0.00	0.998	258.67	0.00	0.555	60.52	0.00	0.535	86.83	0.00	0.772
Orthopteran biomass	4	-649.00	14.07	0.001	258.30	1.85	0.221	59.87	1.60	0.240	87.96	3.38	0.142
Arthropod biomass	4	-648.72	14.35	0.001	258.54	2.09	0.196	60.04	1.77	0.220	89.08	4.50	0.081
Orthopteran biomass * Year	8	-605.25	67.35	0.000	253.73	6.82	0.018	59.86	11.37	0.002	86.08	11.22	0.003
Arthropod biomass * Year	8	-603.20	69.40	0.000	254.91	8.01	0.010	59.51	11.02	0.002	86.62	11.76	0.002

[†]Plasma triglycerides.

Appendix 5. Competitive models ($\Delta AIC_c \leq 4$) for individual nestling condition indices of Dickcissels (*Spiza americana*) in relation to measures of orthopteran and total arthropod biomass, brood size (at time of measurement and maximum during nestling period), and nest parasitism by Brown-headed Cowbirds (*Molothrus ater*) in Conservation Reserve Program fields in central Kansas, USA, 2017-2019.

Model	Deviance	K	ΔAIC_c	w_i
Age-corrected mass				
Orthopteran biomass * year + max brood size	5787.58	10	0.00	0.231
Orthopteran biomass * year + brood size	5787.75	10	0.17	0.213
Arthropod biomass * year + max brood size	5788.82	10	1.24	0.124
Arthropod biomass * year + brood size	5788.92	10	1.34	0.118
Arthropod biomass * brood size	5796.33	7	2.61	0.063
Arthropod biomass * max brood size	5796.34	7	2.62	0.062
Orthopteran biomass * max brood size	5796.86	7	3.14	0.048
Mass/tarsus residuals				
Brood size	2826.26	5	0.00	0.555
Brood size + parasitism	2827.42	6	3.19	0.113
Orthopteran biomass + brood size	2827.45	6	3.22	0.111
Arthropod biomass + brood size	2827.84	6	3.61	0.091
SD tarsus				
Orthopteran biomass + brood size	233.78	5	0.00	0.410
Brood size	236.32	4	0.45	0.328
Arthropod biomass + brood size	235.35	5	1.57	0.187

TRIG[†]

Constant	730.22	4	0.00	0.692
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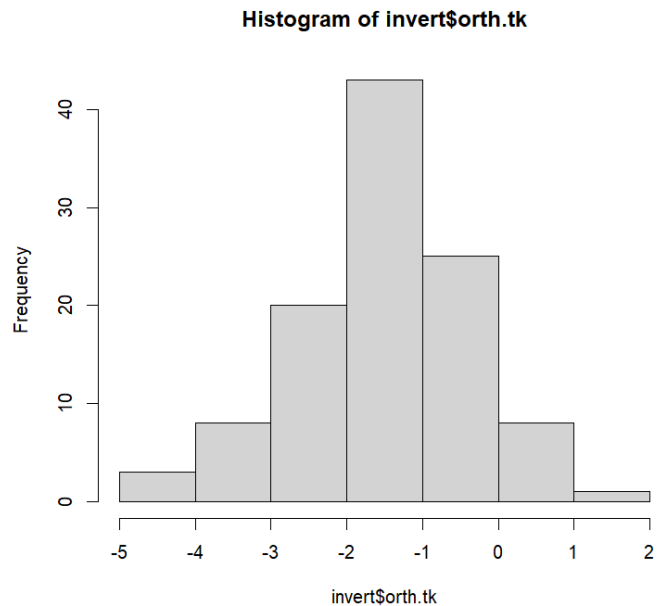
[†]Plasma triglycerides.

Appendix 6. Report on data transformations: Nestling condition of a grassland bird is not associated with food availability in restored grasslands.

Transformation mentioned herein were made according to Tukey's Ladder of Powers using Rcompanion (Mangiafico 2019). Specific transformations per response variable are described in manuscript.

Invertebrate and orthopteran biomass were not normal/right-skewed (Shapiro-Wilk $p < 0.01e-10$) and transformations selected normalized the responses (total invert biomass Shapiro-Wilk $W=0.986$, $p=0.307$) or nearly so in the case of orthopteran biomass (Shapiro-Wilk $W=0.973$, $p=0.028$, though the data appear normally distributed – see Figure S1).

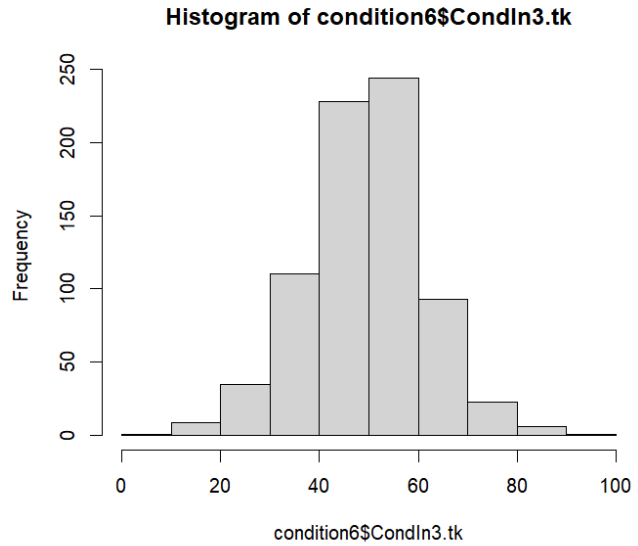
Figure A6.1. Frequency histogram of transformed (log) orthopteran biomass variable "orth.tk."



None of the per-nestling condition indices (mass-age residuals, mass-tarsus residuals, TRIG, SD tarsus at nest level) were normally distributed (all Shapiro-Wilk $p < 0.01e9$). Transformations of nestling-level condition indices approached normality:

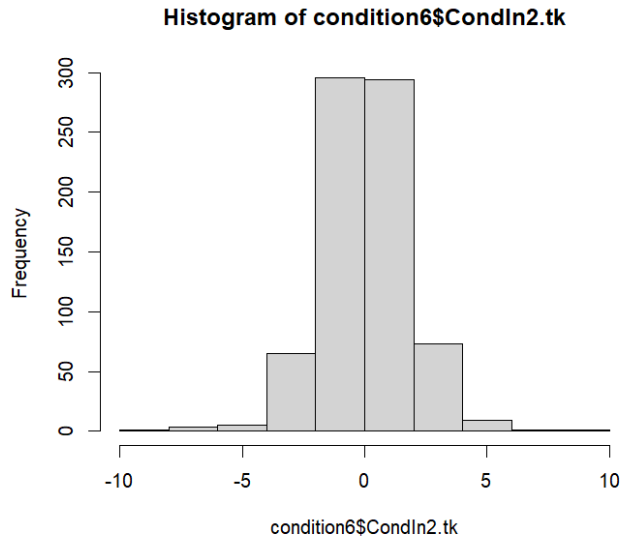
- Mass-age (day 6) residuals, Shapiro-Wilk $p=0.01$, but the histogram of this response appears sufficiently normally distributed (Figure S2).

Figure A6.2. Frequency histogram of transformed ($x^{1.5}$) mass-age (day 6) residuals (“Condln3.tk”) of Dickcissel nestlings.



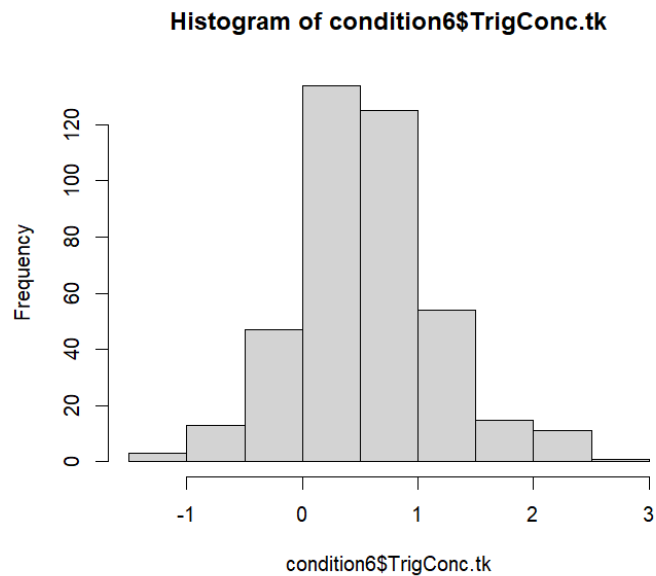
- Mass-tarsus residuals still not able to be improved toward normality by Rcompanion (x^1 selected as best transformation), but the histogram of this response has a convincing unimodal distribution (Figure S3).

Figure A6.3. Frequency histogram of untransformed (x^1) mass-tarsus residuals (“Condln2.tk”) of Dickcissel nestlings.



- Plasma triglycerides (TRIG) did not significantly approximate normality ($p=0.001$) after the selected $x^{1.5}$ transformation, but again, the data visually appear to approximate a normal distribution (Figure S4).

Figure A6.4. Frequency histogram of transformed ($x^{1.5}$) plasma triglyceride levels (“TrigConc.tk”) of Dickcissel nestlings.



Means of nestling condition indices across nestlings per nest, then per field, were not normally distributed, despite predictions of the central limit theorem (mean of means ~ normality).

Shapiro-Wilk tests:

- Mass-age (day 6) residuals, $p < 0.0001$ (right skewed)
- Mass-tarsus, $p = 0.02$ (right skewed)
- SD tarsus, $p < 0.01e-5$ (right skewed)
- TRIG, $p < 0.01e-5$ (right skewed)

Transformations of field-level means of nestling condition indices made improvements toward normality (Shapiro-Wilk test p-values shown):

- Mass-age (day 6) residuals, transformed as $-1/x^{1.5}$, $p = 0.082$ (normally distributed)
- Mass-tarsus residuals, $p = 0.02$ (only slightly right skewed) (but Rcompanion selected x^1 as best—i.e., no—transformation)
- SD of tarsus length, transformed as $x^{0.5}$, $p = 0.002$ (slightly right skewed)
- TRIG, transformed as $x^{0.5}$, $p = 0.352$ (normally distributed)

Field-level means of clutch size and maximum brood size were also compared to field-level arthropod biomass. Mean clutch size was not normally distributed (Shapiro-Wilk $p < 0.01e-5$) and neither was mean maximum brood size ($p = 0.02$). The best transformation for mean clutch sizes ($x^{1.5}$) still did not statistically approximate a normal distribution ($p = 0.0002$) but was unimodal (Figure S5). Transformed field level means ($x^{1.5}$) of maximum brood size were similarly not normally distributed ($p = 0.02$), but being only slightly right skewed (Figure S6).

Figure A6.5. Frequency histogram of transformed ($x^{1.5}$) mean clutch size (“clutch.tk”) per study field in Dickcissel nests.

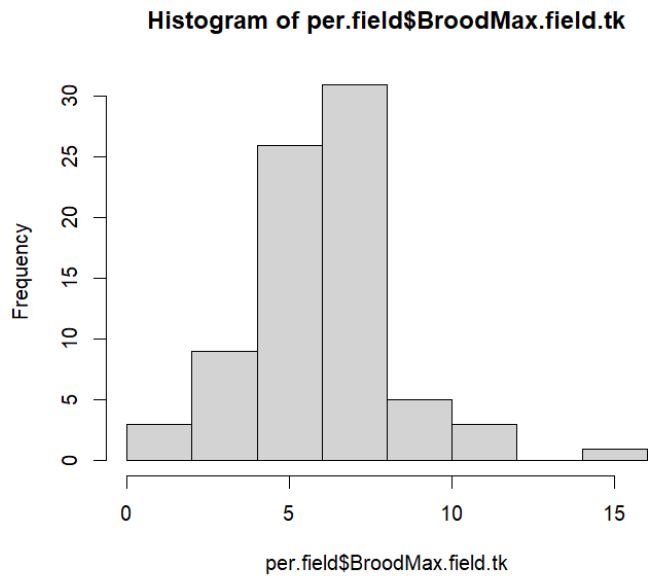
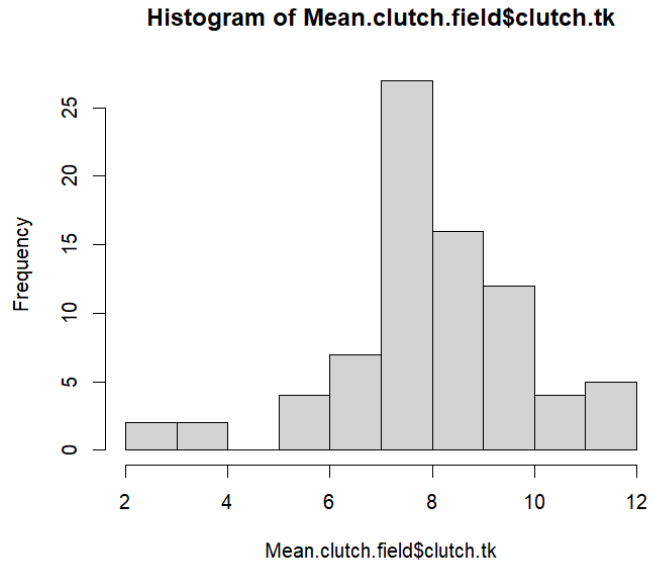


Figure A6.6. Frequency histogram of transformed ($x^{1.5}$) mean, maximum brood size (“clutch.tk”) per field in Dickcissel nests. Brood size here includes Dickcissels and Brown-headed Cowbirds.

LITERATURE CITED

Mangiafico, S. 2019. Rcompanion: functions to support extension education program evaluation. R package version 2.3.7. <https://CRAN.R-project.org/package=rcompanion>.

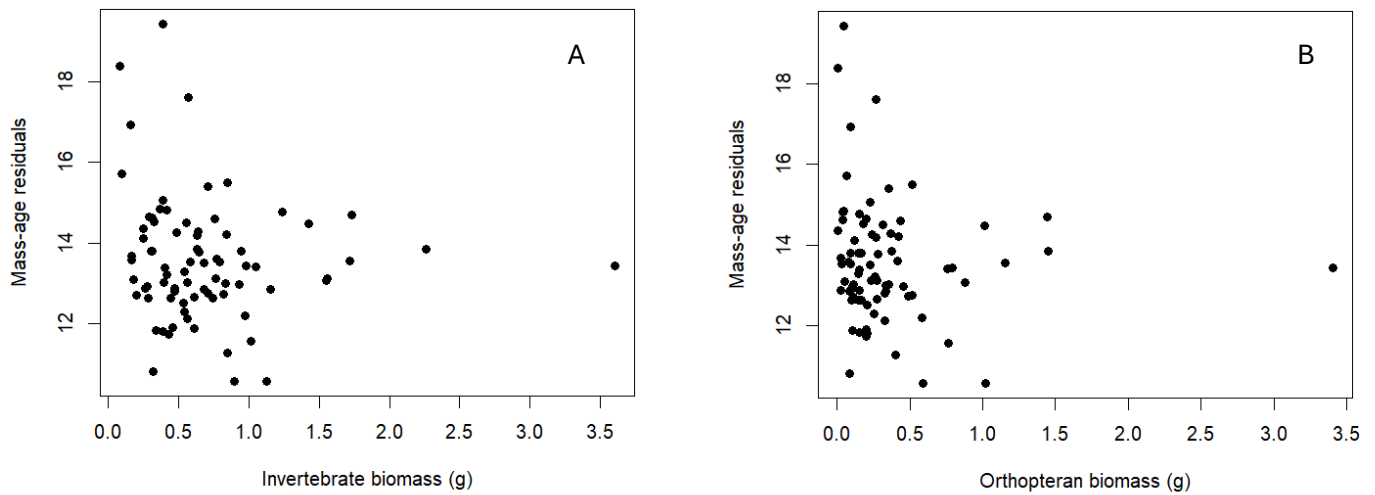


Figure A7.1. Regression residuals of mass vs. nestling age, adjusted to nestling day 6 (see Methods text), of Dickcissel nests per Conservation Reserve Program field in central Kansas (2017-2019) in relation to (A) total invertebrate biomass and (B) orthopteran biomass collected per field among two sweep-net rounds of four, 10-m transects. A clear outlier is apparent at the right of each graph pane (see manuscript for how this was addressed).

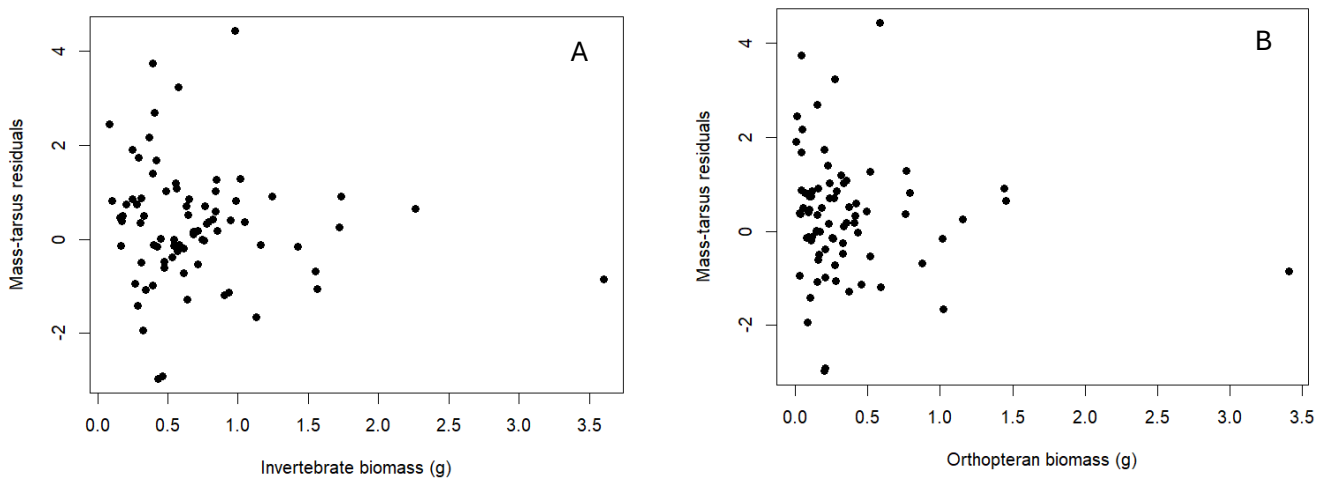


Figure A7.2. Regression residuals of mass vs. tarsus length of Dickcissel nestlings averaged per Conservation Reserve Program field in central Kansas (2017-2019) in relation to (A) total invertebrate biomass and (B) orthopteran biomass collected per field among two sweep-net rounds of four, 10-m transects. A clear outlier is apparent at the right of each graph pane (see manuscript for how this was addressed).

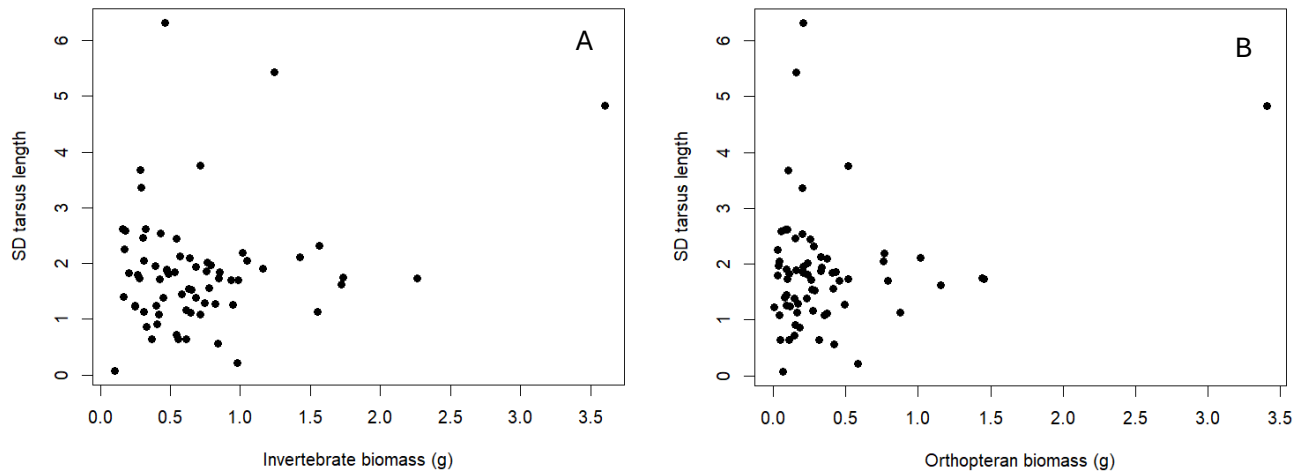


Figure A7.3. Standard deviation of tarsus length of Dickcissel nestlings averaged per Conservation Reserve Program field in central Kansas (2017-2019) in relation to (A) total invertebrate biomass and (B) orthopteran biomass collected per field among two sweep-net rounds of four, 10-m transects. A clear outlier is apparent at the right of each graph pane (see manuscript for how this was addressed).

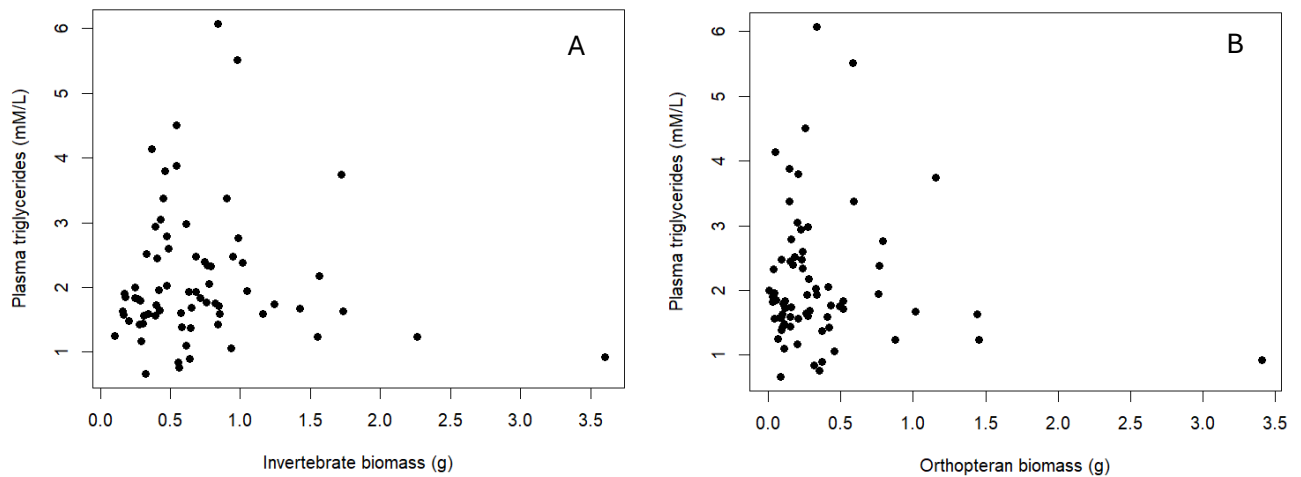


Figure A7.4. Standard deviation of plasma triglycerides in Dickcissel nestlings averaged per Conservation Reserve Program field in central Kansas (2017-2019) in relation to (A) total invertebrate biomass and (B) orthopteran biomass collected per field among two sweep-net rounds of four, 10-m transects.