EFFECTS OF LIVESTOCK GRAZING MANAGEMENT ON THE ECOLOGY OF SHARP-TAILED GROUSE, GRASSLAND BIRDS, AND THEIR PREDATORS IN NORTHERN MIXED GRASS PRAIRIE HABITATS

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EXECUTIVE SUMMARY

This report summarizes the results of the second year (2017) of a three-year (2016–18) research project conducted by researchers at Montana State University to assess the effects of livestock grazing management and rangeland conditions on the population and spatial ecology of grassland birds and their predators. The primary objectives of this study are to 1) investigate rest-rotation grazing as a rangeland management technique to improve habitat conditions for sharp-tailed grouse and 2) develop a mechanistic understanding of the effects of grazing management on the occurrence and abundance of grassland passerines and mesopredators.

For objective 1, we examined the effects of grazing management on the ecology of sharp-tailed grouse by comparing demographic rates and habitat selection among properties managed with rest-rotation grazing to those managed primarily with either season-long or summer rotation grazing. Field efforts in 2017 focused on capturing grouse, tracking radio-marked females, monitoring nests, conducting habitat surveys at nest and brood locations as well as random points in the study area. Sharp-tailed grouse were trapped at 7 leks using walk-in funnel traps during 23 March – 5 May 2017. Overall, 156 sharp-tailed grouse (87 males, 69 females) were captured, including 121 new captures (58 males, 63 females) and 35 recaptures from 2016 (29 males, 6 females). A total of 71 radio-marked females were monitored ≥ 3 times per week throughout the nesting and brood-rearing period (April – July).

Females initiated 76 nests (60 first nests, 16 renests). All females available for monitoring initiated nests (nesting frequency = 1.0), while the probability of renesting (± SE) after first nest failure was 0.64 ± 0.04. Thirty-seven nests successfully hatched and 39 failed (31 depredated, 2 abandoned, 6 hen mortalities). Daily nest survival varied with day of nesting season and ranged from 0.992 ± 0.004 early in the season to 0.887 ± 0.06 at the end of the nesting season. Overall nest survival for the 37-d nesting period including both egg-laying and incubation was 0.22 ± 0.06 in 2017. The best predictors of nest survival were day of the nesting season and visual obstruction at the nest site, which were, respectively, negatively and positively related to survival. We did not find evidence of a grazing system effect on nest survival. Fecundity, the number of female fledglings produced per female, was estimated to be 0.86 ± 0.002. Nest site selection was best predicted by VOR at the nest bowl, density of edge habitat and percent new grass, residual grass, and shrubs, which were all negatively related to survival, with the exception of VOR which was positively related to selection. Nest site selection models with grazing management effects were not supported by the data.

We monitored 37 broods to estimate survival and document habitat selection. Six broods spent the majority of the time (>70% of brood locations) within the easement, 25 spent the majority of time in the reference area, and 6 split time between the two areas. Brood success, calculated as the proportion of broods fledging ≥1 chick to 14-d of age, was 0.17 ± 0.30, 0.68 ± 0.09, and 0.67 ± 0.19 for broods located on the easement, reference, and both areas, respectively. Within successful broods, the proportion of chicks that survived was 0.33 ± 0.14, 0.52 ± 0.09, and 0.52 ± 0.10 for broods located on the easement, reference, and both areas, respectively. We captured 30 fledglings from 13 broods and attached radio-transmitters to 13 fledglings.

Of the 71 radio-marked females, 37 were killed by predators (16 mammalian, 14 avian, 7 unknown predation). An additional 4 females were harvested by hunters. Six females were right censored from the study when their transmitters were found with no sign of death and 2 females left the study area within 2 weeks of capture and were removed from the sample after they could

not be relocated for more than 2 months. Breeding season survival was 0.554 ± 0.09 for birds that primarily used summer rotation pastures, 0.686 ± 0.07 in rest-rotation pastures, and 0.693 ± 0.08 in season-long pastures. However, we found no support for grazing system effects on daily nest survival. Mortalities peaked during September – October, but no carcasses were recovered for testing, so cause of death could not be determined.

We collected a total of 2,944 locations from 71 females and 13 fledglings during April – August, 2017. Mean breeding season home range size for all females was 503 ± 56 ha, but varied from 64 to 3,716 ha. Females showed strong preference for mixed-grass prairie habitats over other non-grassland habitats (e.g., forest, cultivation) during the breeding season. Distance to grassland edge was the best predictor of home range size and was negatively related to the size of female home ranges during the breeding season ($\beta = -1.46 \pm 0.92$). There was no evidence for an effect of grazing system on home range size.

The second objective of the study evaluates the effects of livestock grazing management on the ecology of grassland birds and their predators by comparing grassland bird abundance and mesopredator occupancy within the rest-rotation grazing systems to season-long or summer rotation grazing systems. Field efforts this year focused on conducting grassland bird point count surveys paired with habitat sampling and mesopredator camera trap surveys.

We observed 6,875 birds of 62 species in 915 point count surveys conducted during 19 May – 12 June, 2017; 3,362 birds of 56 species were detected in pastures with rest-rotation grazing systems and 3,513 birds of 52 species were detected on reference properties adjacent to the easement managed under season-long and intensive summer-rotation grazing systems. We selected three grassland birds as focal species for initial investigation; grasshopper sparrows, vesper sparrows, and western meadowlarks each have specific habitat requirements of native grasslands for breeding, recruitment, and survival throughout the breeding season.

Grasshopper sparrow abundance showed a quadratic response to standard deviation VOR and bare ground coverage, where abundance decreased with increasing standard deviation VOR and decreased with increasing bare ground coverage beyond 10%. Grasshopper sparrow abundance also showed a quadratic response to residual grass height and litter depth, where abundance was maximized at ~15cm and ~3cm, respectively. Finally, grasshopper sparrow abundance showed a negative pseudo-threshold response to proportion wooded coulee within the point count area ($\beta = -0.337 \pm 0.038$). Grasshopper sparrow abundance showed a quadratic response to increasing proportion coulee (1000m) and increasing fence length (500m).

Vesper sparrow abundance showed a positive response to increasing bare ground coverage ($\beta = 0.235 \pm 0.071$), a quadratic response to litter depth where abundance was maximized at ~3 cm, and a positive pseudo-threshold response to increasing shrub density ($\beta = 0.131 \pm 0.051$). Vesper sparrow abundance showed a positive pseudo-threshold response to increasing proportion coulee at the 1000 m scale ($\beta = 0.157 \pm 0.047$).

Western meadowlark abundance showed support for a quadratic effect of visual obstruction and residual grass coverage, but the effect was essentially a negative response to increasing visual obstruction and residual grass coverage. Western meadowlark abundance showed a positive pseudo-threshold response to increasing litter depth ($\beta = 0.275 \pm 0.045$) and forb coverage ($\beta = 0.100 \pm 0.030$).

Our candidate model set for management-level effects on bird abundance included effects of grazing system, stocking rate, and average rangeland productivity. The top management-level model for grasshopper sparrow included an interaction between grazing system and rangeland productivity, with the additive effect of stocking rate. Grasshopper sparrow abundance tended to increase with increasing rangeland productivity ($\beta = 0.38 \pm 0.03$); abundance was similar across grazing systems when productivity was moderate (1000 – 2000 $\text{lb} \cdot \text{ac}^{-1}$), but tended to be highest in pastures managed with a season-long grazing system when rangeland productivity was low (<1000 $\text{lb} \cdot \text{ac}^{-1}$) and high ($>2000$ $\text{lb} \cdot \text{ac}^{-1}$). Grasshopper sparrow abundance showed a negative response to increasing stocking rate, and the effect was similar among grazing systems.

The top management-level model for effects on vesper sparrow abundance included an interaction between grazing system and stocking rate, with a main effect of average rangeland productivity. Vesper sparrow abundance decreased with increasing average rangeland productivity beyond 1000 $\text{lb} \cdot \text{ac}^{-1}$, and the effect was similar among grazing systems. Vesper sparrow abundance showed a weak quadratic response to stocking rate, increasing with increasing stocking rate up to approximately 1 $\text{AUM} \cdot \text{ha}^{-1}$, and decreasing beyond this rate.

The top management-level model for effects on western meadowlark abundance included main effects of stocking rate and rangeland productivity. Western meadowlark abundance showed a positive pseudo-threshold response to rangeland productivity, where abundance increased with increasing productivity ($\beta = 0.101 \pm 0.034$) up to 2000 $\text{lb} \cdot \text{ac}^{-1}$, and the effect was similar among grazing systems. Western meadowlark abundance showed a weak quadratic response to stocking rate, where abundance decreased with increasing stocking rates from 0 to 1 $\text{AUM} \cdot \text{ha}^{-1}$, with increases in stocking rate beyond 1 $\text{AUM} \cdot \text{ha}^{-1}$ not strongly affecting western meadowlark abundance.

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2017 Annual Report

OBJECTIVES

Objective 1: Investigate rest rotation grazing as a rangeland management technique to improve sharp-tailed grouse fecundity and survival.

Accomplishments

Initial efforts during 2017 focused on securing access to the necessary private lands in the study area, obtaining research materials and equipment, and hiring field technicians. Subsequent efforts focused on capturing and radio-marking female sharp-tailed grouse and intensive monitoring of radio-marked females to locate nests and broods and monitor survival and space use.

Sharp-tailed grouse were trapped using walk-in funnel traps on both the easement and reference areas of the study site. We recorded standard morphometrics including body mass, wing chord, tarsus length, and culmen length, and fitted all birds with a uniquely numbered metal leg band. Birds were sexed and aged by plumage characteristics. Males were fitted with a unique combination of color bands to allow for resighting at leks next year. We fitted captured females with 18-g necklace-style radio-transmitters with a 6-8 hour mortality switch and an expected battery life of 12 months (model A4050; Advanced Telemetry Systems, Insanti, MN). Previous work found no impact of necklace-style radio-transmitters on prairie-grouse demography (Hagen et al. 2006).

Radio-marked females were located by triangulation or homing ≥3 times/week using portable radio receivers and handheld Yagi antennas during the nesting and brood-rearing period (April—August). When females localized in an area and their estimated location did not change for 2 successive visits, we assumed that the female was sitting on a nest. For half of the females, we used portable radio receivers and handheld Yagi antennas to locate and flush the female so eggs could be counted and the nest location recorded with a handheld GPS unit. We marked nest locations with natural landmarks at a distance ≥ 25 m to aid in relocation. Nests were visited a second time, during which eggs were removed and carried >200 m from the nest and floated in a small container of lukewarm water to assess stage of incubation, estimate hatch date, and estimate the date of clutch initiation by backdating. Nest sites were not visited again until it was determined that the female had departed (i.e., was located away from the nest for ≥ 2 days during incubation and ≥ 1 day after expected hatch date) due to successful hatching of the clutch or failure due to either predation or abandonment. Nesting females were otherwise monitored by triangulation from a distance > 25 m. Thus, nest sites for half of the females were only disturbed by the presence of an observer a maximum of 2 times during the laying and incubation period. The remaining half of the females were never flushed and nest attempts were monitored from a distance of >25 m to evaluate whether the protocol of flushing females has a negative effect on nest survival. A female was assumed to be incubating if she was located in the same location for 2 consecutive visits and nest sites were only visited after the female was located away from the nest for ≥ 2 days during incubation or ≥ 1 day after expected hatch date.

Once the female departed the nest, we classified nest fate as successful (≥1 chick produced), failed, depredated, or abandoned. Nests were considered abandoned if eggs were cold and unattended for >5 days. Nests were considered failed if the eggs were destroyed by flooding, trampling by livestock, or construction equipment. Nests were considered depredated if the entire clutch disappeared before the expected date of hatching, or if eggshell and nest remains indicated that the eggs were destroyed by a predator. When a predation event occurred, the egg remains were evaluated and the area was searched for predator sign. For successful nests, hatchability was calculated as the proportion of the total clutch that hatched and produced chicks. Eggs that failed to hatch were opened to determine stage of development and possible timing of embryo failure.

Successful broods were relocated ≥3 times/week until failure. Pre-fledging brood survival was estimated by conducting flush counts between 14 and 16 days post hatch. Fledging was considered to occur at 14 days post hatch because at that point chicks are able to thermoregulate and are capable of weak flights (Pitman et al. 2006). Flush counts were conducted at dawn when chicks were close to radio-marked females to determine the number of surviving chicks in the brood. After females were flushed, the area was systematically searched and the behavior of the female observed to assess whether chicks were present but undetected. For counts of 0 chicks, the brood female was flushed again the following day to be certain no chicks remained in the brood. Broods were considered successful if ≥ 1 chick survived until fledging at 14-d post-hatch (Pitman et al. 2006). Flush counts were repeated at 14, 30, 45, and 60 days post-hatch or until we were confident that no chicks remained with the female. We used spotlights and a large net to capture ≥35 day old chicks by relocating radio-marked females at night. We recorded morphometrics and equipped 1 fledgling/brood with radio-transmitters attached with glue and sutures. Radio-marked fledglings were monitored ≥3 times per week until death or transmitter failure or loss.

We monitored radio-marked females ≥3 times per week to estimate survival. Transmitters were equipped with a mortality switch that activated after 6–8 hours of inactivity. Once the mortality switch activated, transmitters were located and the area searched to determine probable cause of death. Mortality events were classified as either predation, hunter, other, or unknown. Predation mortalities were further identified as either mammal, avian, or unknown predator. A mortality event was classified as mammalian predation if bite marks, chewed feathers, or mammalian tracks were present. Mortality was determined to be avian predation if the carcass had been decapitated and/or cleaned of the breast muscle with no bite marks, or if the feathers had been plucked. If none of these signs were present or if there were conflicting signs of mortality, the event was classified as unknown predation. Females were censored from the study if their collars were found with no sign of death or if they could not be located for ≥2 months.

We evaluated habitat conditions at each nest and brood flush site within 3 days of hatching or expected hatch date in the case of failure (Figure 1). We recorded visual obstruction readings (VOR) at the nest bowl and at four points 6 m from the nest in each cardinal direction. At each point, VOR was measured in each cardinal direction from a distance of 2 m and a height of 0.5 m using a Robel pole (Robel et al. 1970). We estimated non-overlapping vegetation cover (percent new grass, residual grass, forbs, shrubs, bare ground, and litter) at 12 subsampling locations within 6 m of the nest using a 20 × 50 cm sampling frame (Daubenmire 1959). At each subsampling plot, we measured the heights of new grass, residual grass, forbs, and shrubs. We also estimated shrub cover using the line-intercept method, recording the species, height, and length of each shrub intersecting the transect. For nests, we conducted parallel sampling at randomly selected points within a study area defined by a minimum convex polygon placed around the leks of capture and buffered to 2 km. For broods, we conducted parallel sampling at paired points in a randomly determined direction and distance (maximum of 250 m) from each flush location to represent available habitat within the average daily distance traveled by broods (Goddard et al. 2009). Random points that fell within unsuitable habitat (i.e., water, cultivation, etc.) or were located on properties to which we did not have access were replaced.

We also measured habitat at the home range scale (503 ha, based on estimated home range sizes of sharp-tailed grouse during the breeding season, see below) under the assumption that the home range contained the resources utilized by a female during the nesting season. The home range area was defined as a circular plot with a 1,300-m radius centered on each nest, brood, and random location. We calculated habitat variables at the home range scale using remotely sensed data and ArcMap 10.4. We included road datasets for both Montana and North Dakota and calculated the density of paved and gravel roads within each home range (Montana State Library, North Dakota GIS Hub Data Portal). Paved roads, including state highways, had higher traffic volumes and were assumed to represent a different level of disturbance than gravel roads. We also included the locations of oil pads which represented another form of disturbance in the study area and calculated the distance to the nearest oil pad from the center of each home range. Landcover analyses utilized the 30 m resolution LANDFIRE data depicting vegetation type (LANDFIRE 2013). We measured the distance from the center of each home range to the nearest patch of non-grassland habitat. In addition, we used the Patch Analyst Extension in ArcMap to calculate the proportion of grassland, the density of edge habitat, and grassland shape complexity.

We collected stocking information for every pasture in which radio-marked sharp-tailed grouse were located. For each pasture, we recorded the type of animal (cow/calf pairs, heifers, bulls, or horses), number of head, and the dates when animals were in the pasture. This allowed us to calculate the following grazing management variables: grazing system (rest-rotation, summer rotation, season-long), stocking rate (AUM ha^{-1}), stocking intensity (AU ha^{-1}), stocking duration (in months), and season of stocking (growing season [May—July], post-growing season [August—November], winter [December—April], or rest [no grazing for entire year]).

We used field data to estimate eight demographic parameters related to fecundity for sharp-tailed grouse (Table 1). Some of these parameters, including clutch sizes and chicks per egg laid (CPE), can be estimated directly from field data. However, other parameters are observed imperfectly. Nests are not observed from the initiation date and nests that fail before discovery must be considered to make population-level inferences. To account for imperfect observation, we used the nest survival model in Program MARK to calculate maximum likelihood estimates of daily nest (NSURV) following the methods described below.

Nesting rate (NEST) was calculated as the percentage of females that attempted a nest. The probability of renesting (RENEST) was calculated as the number of observed renesting attempts divided by the number of unsuccessful first nests minus the number of females that had first nests but were unavailable to renest. A hen was considered unavailable if she was killed during the first nest attempt or was not relocated after the failure of a first nest attempt. Initial brood size was determined by the number of chicks that were known to hatch based on nest observations. Brood success (BSURV) was calculated as the proportion of broods that successfully fledged ≥1 chick. Fledging success (FPC) was calculated as the proportion of chicks that survived until fledging among successful broods. Broods were included in the easement category if >70% of brood locations were within the easement boundaries, in the reference category if >70% of locations were in the reference area, and in the category "both" if they split their time between the two areas.

Fecundity (F), or the number of female fledglings produced per female, is expressed as a function of these parameters using the following equation:

$$F = \left[(NEST * CS_1 * NSURV_1) + [(1 - NSURV_1) * RENEST * CS_2 * NSURV_2]\right] * CPE * BSURV * FPC * 0.5$$

Bootstrapping procedures were used to calculate 85% confidence intervals for fecundity estimates by randomly drawing from the underlying distributions of input parameters (McNew et al. 2012).

Nest success is the proportion of nests that produce ≥1 chick, whereas nest survival accounts for potential losses of nests before discovery. We constructed nest survival models using the RMARK package in Program R (R Core Team 2014, Vienna, Austria) to calculate maximum likelihood estimates of daily nest survival and evaluate the effects of habitat conditions on daily nest survival during a 75-d nesting period during 28 April – 10 July (White and Burnham 1999, Dinsmore et al. 2002). Before fitting models, we examined correlations for each pair of variables and if a pair was highly correlated ($r \geq 0.5$, $p < 0.05$), we used single-factor models to determine which of the two variables accounted for the largest proportion of variation in daily nest survival. We considered the variable with the lowest model deviance to be the primary variable to consider in subsequent analyses.

We developed a candidate set of models representing hypotheses regarding the effects of grazing management and local and home-range scale habitat conditions. Models were compared using Akaike's Information Criterion adjusted for small sample sizes (AIC_c) and models with large model weights (w_i) and AIC_c values ≤ 2 from the best-fit model were considered equally parsimonious (Burnham et al. 2011). We first assessed underlying effects models including variables of nest age, nest attempt, female age, female condition, flushing effect compared to a null model of constant daily nest survival (Goddard and Dawson 2009). Female condition was calculating by regressing body mass against the length of the wing chord using the reduced major axis method (Green 2001). We also included and evaluated the effect of Year (2016, 2017) to account for any annual variation in nest survival not addressed by our habitat measures. We then selected the most parsimonious models at each of the different spatial scales (nest- and home range level) and assessed them in the final candidate model set with grazing management variables, including grazing system and stocking rate. The stocking rate during the previous year's grazing season rather than the current year was used, because the majority of first nests were initiated prior to pastures being stocked. Variables considered at the nest level included VOR at the nest bowl and averaged within the 6 m radius plot, distance to grassland edge, and the percentage of shrubs, new grass, residual grass, forbs, and bare ground. Different functional relationships with VOR were examined, including linear, exponential and natural log models (i.e., pseudo-threshold effects; Dugger et al. 2005, McNew et al. 2014). Variables considered at the home range level included the proportion of grassland habitat, density of edge habitat, grassland shape complexity, and distance to oil pad. The density of either gravel or paved roads was ≥ 0 for only a small portion of nests and so neither variable was included in analyses.

We developed a separate candidate set of models to evaluate the effects of flushing on nest survival. In addition to a flushing variable, covariates considered included hen age and day of the nesting season to examine what factors might be mediating effects of flushing on nest survival. Models were compared using the criteria described above.

Overall nest survival for precocial species is the probability that a nest will survive the entire nesting period, defined as the mean laying plus incubation interval for grouse at our study sites (37-d). We calculated the overall nest survival probability with parametric bootstrapping, using the beta estimates and variance-covariance matrix from the top model in the nest survival analysis. Variance of overall nest survival was estimated with the delta approximation (Powell 2007). The average duration of incubation period (27-d) was determined from observations of our sample of successful nests and from previous work (Connelly et al. 1998).

We also examined habitat variables influencing nest site selection at multiple spatial scales using resource selection functions. Nest were considered used sites and, as we did not conduct searches for nests of unmarked grouse, random points were considered available following sampling protocol B of Manly et al. (2002). Therefore, we used resource selection functions to compare habitat variables at used sites (1) and available sites (0). Habitat variables at each spatial scale are described above. We built sets of candidate models at each spatial scale and across all scales using the 'ResourceSelection' package in Program R with female ID as a random effect to account for potential autocorrelation among nests. Models were compared using AIC and model selection was based on minimization of AIC and AIC weights (w_i). Variables considered at the nest level included VOR at the nest bowl and averaged within the 6 m radius plot, distance to grassland edge, the percentage of shrubs, new grass, residual grass, forbs, and bare ground, and shrub and forb height. Different functional relationships with VOR were examined, including linear, exponential and natural log models (i.e., pseudo-threshold effects; Dugger et al. 2005, McNew et al. 2014). Variables considered at the home range level included the proportion of grassland habitat, density of edge habitat, grassland shape complexity, and distance to either oil pad or lek. The density of either gravel or paved roads was ≥ 0 for only a small portion of nests and so neither variable was included in analyses.

We calculated survival rates of radio-marked females using staggered entry Kaplan-Meier models with the survival package in Program R. Individuals that survived <1-week post-capture were not included in analyses to account for any potential negative effects of capture on survival. Encounter histories were created on a weekly time-step for the breeding season from 15 March to 15 August (i.e., 15–21 March = Week 1). Data were left-censored to allow for the staggered entry of birds into the marked population and right-censored to account for radio failure and missing birds. We used Cox proportional hazards models to evaluate differences in breeding season survival among years and between each grazing treatment (Sandercock et al. 2011, Winder et al. 2014). Females were assigned to the grazing system in which their home range was centered or the system which contained the majority of relocations if there were insufficient locations to calculate a home range.

Results.– Seven sharp-tailed grouse leks were monitored during 15 March – 5 May 2017. Sharp-tailed grouse were trapped at 4 easement and 3 reference leks during 23 March – 5 May 2017. Mean overall lek attendance was 14.4 birds (average of 12.6 males and 2.1 females) during this period (Table 2). We captured a total of 156 sharp-tailed grouse (87 males, 69 females), including 121 new captures (58 males, 63 females) and 35 recaptures from 2016 (29 males, 6 females) and 58 females were radio-marked (Table 3). An additional 13 females that were radio-marked during the 2016 field season were still present in the study area and therefore monitored for a second year. Overall, 71 radio-marked females were monitored ≥ 3 times per week throughout the nesting and brood-rearing period.

Females initiated 76 nests (60 first nests, 16 renests; Figure 2, Table 4). Nine females died or were censored from the study before initiating a nest (see below for full mortality results). Median nest initiation date for all nests was 2 May (28 April for first nests, 26 May for renests; range: 17 April 17 – 10 June). Nesting frequency was 1 (all females available for monitoring initiated nests), while the probability of renesting (± SE) after first nest failure was 0.64 ± 0.04. Thirty-seven nests successfully hatched and 39 failed (31 depredated, 2 abandoned, 6 hen mortalities; Table 4). Hatch rate of eggs (± SE) for first nests and renests was 91.4 ± 15.3% and 88.0 ± 16.5%, respectively. Mean clutch size for all nest attempts was 11.5 ± 2.3 eggs. Mean clutch size for first nest and renests was 11.9 ± 2.3 and 10.1 ± 1.7 eggs, respectively.

Eight demographic parameters were estimated using field data (Table 5). Fecundity was calculated as a function of these parameters using the equation described above and was estimated to be 0.86 ± 0.002 female fledglings produced per female.

Daily nest survival varied with day of nesting season and ranged from 0.992 ± 0.004 early in the season to 0.887 ± 0.06 at the end of the nesting season. Due to this variation, overall nest survival for the 37-d nesting period including both egg-laying and incubation was calculated using parametric bootstrapping and was 0.22 ± 0.06. The underlying-effects model that best predicted nest survival included day of the nesting season ($\Delta AIC_c = 0$, $w_i = 0.27$, Table 6). Models including flushing effects, nest attempt, female age and year also improved model fit over the null model and so were examined in later analyses. Preliminary analyses suggested that visual obstruction averaged across the 6 m radius plot best predicted nest survival and that a pseudo-threshold model best represented the relationship between visual obstruction and nest survival, so only models with the natural log transformation of average VOR were included in analyses. At the nest-level, VOR was in the top four models, accounting for 77% of model weight (Table 7). Percentage forb, residual grass, and new grass each received some support in combination with VOR ($\Delta AIC_c = 1.60 – 1.94$, $w_i = 0.13 – 0.15$), but confidence intervals overlapped zero, so the more parsimonious model including only VOR was included in the final model set. At the home-range level, no variables improved model fit over the null model (Table 8) and so only variables at the nest level and underlying effects were included with grazing management variables in the final model set. In the final candidate model set, the model that included both VOR and day of the nesting season received the most support ($\Delta AIC_c = 0$, $w_i = 0.49$, Table 9). Nest survival declined with day of the nesting season, and increased with VOR up to a threshold, as represented by the pseudo-threshold model (Figure 3). Confidence intervals for both VOR and day of the nesting season did not overlap zero (Figure 4), suggesting significant effects. Models with grazing system effects were not supported (Table 9), and nest survival did not differ among grazing systems (Figure 5).

We found no evidence for a system-level effect of grazing management, so we examined the effects of finer-scale grazing management variables from both the current and previous year on nest survival. We examined the effects of all grazing management variables defined above (grazing system, stocking rate, stocking intensity, duration of stocking, and season of stocking) in combination with our top model from the full analysis described above, which included both VOR and day of the nesting season. There was significant model uncertainty, but the top model included the habitat variables in addition to grazing duration in the current year ($\Delta AIC_c = 0$, $w_i =$

0.25, Table 10), which represented a slight improvement over the null model including only VOR and day of the nesting season ($\Delta AIC_c = 0.73$, $w_i = 0.18$). Nest survival increased slightly with an increase in the duration of grazing ($\beta = 0.08 \pm 0.05$), but confidence intervals overlapped zero suggesting a non-significant effect. Taken together with the full analysis described above, a biologically-relevant effect of grazing management on nest survival does not appear to be supported.

Flushing females from their nests had a significant negative effect on nest survival (Table 11), with reduced survival due almost entirely to predation rather than nest abandonment. Overall nest survival during the 37-d nesting period was 0.20 ± 0.05 for females that were flushed at least once off the nest, compared to 0.44 ± 0.10 for females that were never flushed from their nests (Figure 6). To evaluate how biases in estimates of nest survival associated with flushing would bias estimates of population growth rates, we built deterministic matrix models using estimated vital rates from our study population with the only difference between the two groups being the estimates of nest survival. Flushing decreased population growth rates by 63% (Figure 7). Projected population growth (λ) was 1.30 ± 0.002 for females that were flushed at least once off the nest, compared to 2.06 ± 0.003 for females that were never flushed from their nests (Figure 7).

Preliminary analyses suggested that visual obstruction at the nest bowl best predicted nest site selection and that a quadratic model best represented the relationship between visual obstruction and nest site selection, so only models with the quadratic relationship of nest VOR (Nest VOR + Nest VOR^2) were included in analyses. At the nest-level, nest site selection was best predicted by VOR at the nest bowl and the percentage of new grass, residual grass, and shrubs, with all other models >40 AIC away from the top model (Table 13). At the home-range level, only density of edge habitat improved model fit over the null model (Table 14) and so was included with nest level and grazing management variables in the final model set. In the final candidate model set, the model that included all habitat variables at both the nest and home-range level received the most support, with all other models >56 AIC below the top model (Table 15). Nest site selection increased with VOR at the nest bowl up to a threshold before declining and decreased with edge density and percent new grass, residual grass and shrubs (Figure 8). Confidence intervals for VOR, edge density, and percent new grass, residual grass, and shrubs did not overlap zero (Figure 8), suggesting significant effects. Models with grazing management effects were not supported (Table 15) and the probability of nest site selection did not differ among grazing systems (Figure 8).

We monitored 37 broods to estimate survival and document habitat use (Table 12). Six broods spent the majority of the time (>70% of brood locations) on the easement, 25 spent the majority of time in the reference area, and 6 split time between the two areas. Brood success, calculated as the proportion of broods fledging ≥1 chick to 14-d of age, was 0.17 ± 0.30, 0.68 ± 0.09, and 0.67 ± 0.19 for broods located on the easement, reference, and both areas, respectively. Of broods that survived to fledging, the proportion of chicks that survived was 0.33 ± 0.14, 0.52 ± 0.09, and 0.52 ± 0.10 for broods located on the easement, reference, and both areas, respectively. We captured 30 fledglings from 13 broods and attached radio-transmitters to 13 fledglings. Three of the radio-marked fledglings survived until the 60-d flush count. However, 54% of radio-marked fledglings were censored, and all the censored fledglings survived at least until 45-d post-hatch, so it is possible that transmitters were not reliably staying attached past a certain age.

We determined that 37 (52%) females were killed by predators: 16 (23%) and 14 (20%) by mammalian and avian predators, respectively, and 7 (10%) by an unknown predator. An additional 4 (6%) females were killed by hunters during the fall hunting season. Six (8%) females were right censored from the study when their transmitters were found with no sign of death. An additional 2 (3%) females left the study area within 2 weeks of capture and were right censored after they could not be relocated for more than 2 months. Mortalities peaked during September – October, but no carcasses were recovered for testing, so cause of death could not be determined. Breeding season survival (April—August) did not differ between years or between any of the three grazing treatments with the null model receiving the majority of model weight ($\Delta AIC_c = 0$, $w_i = 0.62$) compared to models with either year ($\Delta AIC_c = 1.89$, $w_i = 0.24$) or grazing system effects ($\Delta AIC_c = 2.88$, $w_i = 0.15$). There was no evidence that survival differed between grazing treatments, as a model including survival as a function of grazing treatment did not improve AIC_c over the null model (Figure 9). Breeding season survival ranged from 0.554 ± 0.09 in summer rotation pastures, 0.686 ± 0.07 in rest-rotation pastures, and 0.693 ± 0.08 in season-long pastures (Table 16).

Goals For Next Quarter:

We will continue to monitor radio-marked females ≥ 1 time/month through the non-breeding season (Sept – March) until death or transmitter failure or loss. In January and February, we will prepare for the 2017 field season by securing additional access to private lands if necessary, procuring research materials and field equipment, and hiring technicians. Three technicians will be hired to start in March and we will also begin trapping and radio-marking new females in mid-March.

Objective 2: Investigate impacts of rest-rotation grazing on sharp-tailed grouse home ranges, movements and habitat selection.

Accomplishments

Radio-marked females were located via triangulation or homing ≥ 3 times/week using portable radio receivers and handheld Yagi antennas during the nesting and brood-rearing period (April–August) and ≥ 1 times/month during the rest of the year (September–March).

Coordinates for triangulated locations were calculated using Location of a Signal software (LOAS; Ecological Software Solutions LLC, Hegymagas, Hungary) and examined for spatial error. All locations with low estimation precision (>200 m error ellipse) were discarded for initial analysis, but the level of acceptable error will be examined on a case-by-case basis in the future. Previous studies have found that small sample sizes can bias home range estimates (Seaman et al. 1999), so analyses were restricted to birds with ≥ 30 unique locations after excluding multiple relocations of a female at the same nest. We used the fixed kernel method (Worton 1989) with the default smoothing parameter to calculate 95% home ranges for the breeding season (April–August) using the adehabitatHR package in Program R. We also calculated centroids for each home range by estimating the 1% volume contour of each home range and used the geographic center of that contour as the centroid. We then measured distance from each centroid to the nearest lek, grassland patch edge, road and oil pad. We examined second-order habitat selection, or the selection of habitat for an individual's home range within the larger study area, using the adehabitat package in Program R and conducted composition analysis of used versus available habitat (Johnson 1980, Aebischer et al. 1993). Available habitat was defined as the area within 2 km of capture leks and each female's 95% home range represented the used space at an individual level. Habitat classifications utilized the 30-m resolution LANDFIRE data depicting vegetation type (LANDFIRE 2013). Linear models were used to evaluate the effects of year, nest outcome, distance to nearest lek, grassland patch edge, road, and oil pad, and grazing conditions at the home range centroid on home range size.

Results.– During the 2017 breeding season, we collected a total of 2,944 locations from 71 females and 13 juveniles. During the 2017 breeding season (April–August), 53 females had ≥30 unique locations. Mean breeding season home range size for all females was 503 ± 56 ha, but varied from 64 ha to 3,716 ha (Figure 10). Breeding season habitat use was ranked as follows: mixed grass prairie > close grown crop > shrubland > introduced upland perennial grasslands >> wheat >> developed ruderal grasslands (Table 17). Mixed grass prairie is defined by soil texture and includes areas where soils are primarily fine and medium-textured and do not include sands, sandy soils, or sandy loams. Dominant graminoid species include *Pascopyrum smithii*, *Nassella viridula*, *Hesperostipa comata*, and *Festuca* spp. Shrubland is distinguished from mixed grass prairie in that it contains greater than 10% shrub cover in combination with topographic relief. Close grown crops and wheat are defined as areas where crop vegetation accounts for greater than 20% of total vegetation; close grown crops include land being actively tilled. Introduced upland perennial grasslands include areas that are dominated by introduced perennial forb or grassland species including such as *Agropyron cristatum*, *Bromus inermis*, and *Poa pratensis*. Developed ruderal grasslands are defined as herbaceous vegetation resulting from succession following a significant anthropogenic disturbance and are usually characterized by unnatural combinations of species. Females strongly selected for mixed-grass prairie habitats, even though roughly 81% of the entire study area was composed of prairie. Distance to grassland edge was the best predictor of home range size (Table 18) and was negatively related to the size of breeding season home ranges for females ($\beta = -1.46 \pm 0.92$). There was also evidence ($\Delta AIC_c < 2$) for an effect of distance to lek and nest outcome on home range size, with home range size increasing with distance to lek ($\beta = 0.09 \pm 0.07$) and decreasing for females who had a successful nest attempt in a given year ($\beta = -107.64 \pm 125.42$). There was no evidence for an effect of the grazing system at the centroid of a female's home range on home range size (Figure 11, Table 18).

Goals For Next Quarter:

We will focus efforts in the next quarter on evaluating third-order habitat selection, or habitat selection within the home range, using resource utilization functions (Johnson 1980, Marzluff et al. 2004). RUFs use utilization distributions to quantify the probability of space use and then relate that space use to habitat variables by quantifying use within a defined area (e.g., the home range) as continuous rather than discrete (used vs. unused) and evaluating the entire distribution of an animal's movements rather than the individual sampling points (Marzluff et al. 2004). In addition, RUFs are recommended for use with VHF telemetry data which is subject to measurement error (Hooten et al. 2013). While home ranges provide information on where an individual was located, RUFs can evaluate why particular areas are used more frequently by linking differential space use to habitat variables. To implement RUFs, we will create utilization distributions within 99% fixed kernel utilization distributions using the adehabitatHR package in Program R. We will then evaluate habitat conditions at each grid cell within the polygons using the methods described above. Variables considered will include grazing system, stocking rate, vegetation type, distance to nearest non-grassland habitat, distance to road, and distance to oil pad. The height of the utilization distribution represents the relative amount of use of a given cell, which can then be related to habitat conditions using multiple regression (Marzluff et al. 2004). Using multiple regression, we will estimate coefficients of relative resource use by relating the cell values from the utilization distribution to the habitat variables using the ruf package in Program R. Coefficients represent the degree to which an individual or population utilize resources within the defined home range (Marzluff et al. 2004). Mean standardized coefficients will be used to make population-level inferences about each habitat variable and standardized coefficients with 85% confidence intervals that do not overlap zero will be considered significant. While unstandardized coefficients are necessary to predict expected use, standardized coefficients allow for inferences about the relative influence of habitat variables on animal use and can be used to rank the relative importance of each variable (Marzluff et al. 2004). Results of the habitat selection analyses will be presented at the Society for Range Management's 71st Annual Meeting in February.

Objective 3: Develop a mechanistic understanding of the ecological effects of various grazing treatments with a focus on rest rotation grazing by examining abundance and space use of the grassland bird and mesopredator communities

Accomplishments:

Methods.– Efforts this past year focused on collecting data to test primary hypotheses regarding effects of livestock grazing management on abundance, diversity, and space use of grassland birds and mesopredators in a northern mixed-grass prairie ecosystem. During 23 May through 27 July, 2017, we conducted grassland bird point count surveys paired with habitat sampling, and mesopredator camera trap surveys. Prior to our 2016 field season, we randomly generated 305 points across gradients of habitat conditions within a Montana Department of Fish, Wildlife, and Parks conservation easement managed under rest-rotation grazing, and on adjacent private and federal lands managed under season-long or intensive summer rotational grazing methods (Figure 12). We used the same survey points during the 2017 field season, as grassland birds show low site fidelity between breeding seasons (Jones et al. 2007). We surveyed 150 points on the conservation easement, with 50 points in each of the three rotational pasture types. We surveyed 155 points on reference pastures adjacent to the easement, with 60 points located within season-long grazing systems and 95 points within summer rotational grazing systems, where cattle are turned out at the end of May and are moved between pastures after 6–8 weeks. To avoid double counting of individuals and assure statistical independence, points were spaced ≥ 300m apart. Points were located ≥ 200 m from pasture boundaries to avoid counting birds using multiple treatments, ≥ 400 m from oil pads, and ≥ 250 m from gravel roads to control for bird avoidance of these areas (Thompson et al. 2015).

Avian point count surveys began 19 May 2017, after all breeding species had arrived. Surveys were completed within a 5-week period to ensure population closure; our surveys concluded 12 June, 2017. At each randomly generated location, grassland birds were surveyed with three replicated 5-minute point count surveys. A single trained observer identified and tallied all birds detected visually or aurally within 100 m of the point, noting the time of first detection and the distance from observer to the bird when it was first detected (0-25 m, 26-50 m, 51-75 m, 76-100 m; Ralph et al. 1993). Other data recorded included sex (dichromatic species only), group size, vocalization, and behavior of each species identified. At each survey location, the observer recorded the point and pasture to be surveyed, date and time, percent overcast, precipitation, temperature, and wind speed. Point count surveys were conducted from one-half hour before sunrise through no later than 0900h MST. Surveys were not conducted if average wind speed exceeded 10 mph or during rainfall.

Habitat conditions were measured within bird survey areas the same day point count surveys were conducted. Three 20-m transects were established within 100 m of each survey point, with one transect located at the survey point and oriented in a random direction, and two transects located and oriented randomly within 100 m of the survey point. Subplots were spaced 5 m apart along each transect. At each subplot, visual obstruction was measured from the north at a distance of 2 m and a height of 0.5 m (VOR; Robel et al. 1970), and vegetation coverages were measured using methods of Daubenmire (1959). Percent coverage of new growth grass, residual grass, litter, forb, and bare ground were measured in percentage classes (0-5, 5-25, 25-50, 50-75, 75-95, and 95-100%). Heights (cm) of the nearest plant were measured for each new growth grass, residual grass, litter, and forb. We estimated shrub cover using line intercept surveys, where the species of each shrub intersecting the transect was recorded, as well as the height and length of the shrub as it crossed the transect (Canfield 1941).

To estimate occupancy and evaluate space use of the mesopredator community, passive infrared remote field cameras (Browning BTC 5HD, Browning, Morgan, Utah) were used to survey random sites within the study area. Remote cameras have been cited as the best survey method for detecting medium and large sized carnivores in most habitats (Silveira et al. 2003). Automated cameras also record the time and date for every photographic event captured, making them useful for temporal associations, such as daily and seasonal activity patterns. Ninety-three predator survey points were randomly selected within the study site, with 45 points in rest-rotation pasture treatments (easement) and 48 in season-long and summer rotation grazing pasture treatments (reference areas). Cameras were set in the most optimal location within 200 m of the point, where detection of predators was maximized, and spaced ≥ 600m apart to ensure independence. Cameras were often set at heavy use areas along a habitat edge, where land cover changes on the landscape at the intersection of water, grassland, agriculture, and/or trees and shrubs (Burr 2014). Habitat edges and game trails were used with a goal of increasing detection probabilities, as mammalian predators are thought to prefer such edges while traveling and foraging (Andrén 1995).

Cameras were programmed to be active 24 hours per day with a 1-minute delay between photographic events and a two photo burst for each event. For each photographic event, the date and time of the event were recorded, along with the temperature (°C), barometric pressure, moon phase, and camera ID. Cameras were secured to tree trunks or, if not available, mounted on metal stakes, and positioned approximately 0.5m above the ground and 2m in front of a scent or bait lure. When present, cameras were faced toward game trails to maximize detections.

For each of the three sampling periods, first (23 May – 13 June, 2017), second (14 June – 5 July, 2017), and third (6 July – 27 July, 2017), camera sites were revisited weekly to replenish scent attractants, download and clear memory cards of digitally recorded images, change camera batteries, and remove any obstructive vegetation. For each three-week sampling period, camera sites were revisited weekly to re-bait stations, download and clear memory cards of digitally recorded images, change camera batteries, and remove any obstructive vegetation. Each camera trap was baited with a long-distance trapping lure (Gusto; Minnesota Trapline Products, Inc.) in an attempt to increase detection frequencies of predators. Following the 3-week survey period, cameras were moved to new random points for another 3 weeks. Thus, 31 camera traps were used to survey 93 sites during the 2017 field season.

Following the 2017 field season, we conducted analyses of 2016 and 2017 grassland bird survey data, evaluating the effects of vegetation conditions at the local and landscape levels, and rangeland productivity, stocking rate, and grazing system at the management level on grassland bird abundance. To evaluate these effects on abundance of focal species, we used N-mixture models, which simultaneously estimate detection probability and abundance, using program R package 'unmarked' (Fiske and Chandler 2011). The binomial N-mixture model has proven an effective means to simultaneously estimate detection probability and abundance of unmarked individuals identified during spatially replicated count data, operating under the assumption of population closure, where births/deaths and emigration/immigration are equal to zero for the duration of the surveying period (Royle 2004). Grassland birds have low site fidelity from one breeding season to the next, allowing for the pooling of survey data from multiple seasons and treatment of bird survey locations as independent (Jones et al. 2007, Ahlering and Merkord 2016). Additionally, habitat conditions at each bird survey location were measured during both seasons and applied to the respective survey season when evaluating avian abundance and detection probability, eliminating a potential year-effect among local-level habitat conditions at the survey location and the associated avian abundances.

We used stepwise model selection techniques to identify the factors influencing detection probability and abundance for each grassland bird species, where we started with a highly parameterized model and eliminated uninformative parameters based on their lack of influence on detection probability or abundance. Submodels related to species-specific detection probabilities were fit prior to fitting models for local abundance. Once stepwise model selection was complete, we used $\mathrm{AIC_c}$ to compare models and identify important variables to include in the final detection or abundance model for each grassland bird species. Supported models with large model weights (w_i) and $\mathrm{AIC_c}$ values ≤ 2 from the best-fit model were considered parsimonious (Burnham et al. 2011). When a supported model differed from the top model by a single parameter, we considered the additional parameter to be uninformative and excluded this parameter from inclusion in the global model (Arnold 2010).

Each of the habitat conditions we measured in the field have been shown to affect avian detection probability or abundance, so stepwise model selection was most appropriate in the absence of any *a priori* hypotheses. Variables included in the detection probability submodel were observer, year, day of season, time of day, wind speed, slope, shrub density, and shrub height. Variables included in the local-level habitat submodel hypothesized to affect avian abundance were visual obstruction reading, standard deviation of visual obstruction readings, residual grass coverage, residual grass height, litter depth, bare ground coverage, shrub density, shrub height, wooded coulee coverage. For each covariate included in detection and abundance submodels, we evaluated whether a linear, quadratic, or pseudo-threshold effect was best supported for its influence on avian detection probability or abundance.

We built a candidate model set for the management-level effects on avian abundance based on *a priori* hypotheses. Factors hypothesized to affect avian abundance at the management-level were grazing system, stocking rate, and rangeland productivity. We interviewed landowners to acquire stocking information for pastures within the study area, confirming the accuracy of these rates based on field observation. Average rangeland productivity within each point count survey area was quantified based on ecological site data from SURGO Web Soil Survey, using the Representative Value (RV) for each ecological site type within the study area.

Results.– We detected 6,875 birds of 62 species during 915 point count surveys; 3,362 birds of 56 species were detected in pastures with rest-rotation grazing systems and 3,513 birds of 52 species were detected on reference properties adjacent to the easement managed under season-long and intensive summer-rotation grazing systems (Table 19, 20). We identified 23 species of grassland obligate birds, Baird's sparrow (*Ammodramus bairdii*), bobolink (*Dolichonyx oryzivorus*), Brewer's sparrow (*Spizella breweri*), clay-colored sparrow (*Spizella pallida*), common yellowthroat (*Geothlypis trichas*), eastern bluebird (*Sialia sialis*), eastern kingbird (*Tyrannus tyrannus*), field sparrow (*Spizella pusilla*), grasshopper sparrow (*Ammodramus savannarum*), horned lark (*Eremophila alpestris*), killdeer (*Charadrius vociferus*), lark bunting (*Calamospiza melanocorys*), lark sparrow (*Chondestes grammacus*), loggerhead shrike (*Lanius ludovicianus*), marbled godwit (*Limosa fedoa*), mountain bluebird (*Sialia currucoides*), red-winged blackbird (*Agelaius phoeniceus*), Say's phoebe (*Sayornis saya*), Sprague's pipit (*Anthus spragueii*), sharp-tailed grouse (*Tympanuchus phasianellus*), upland sandpiper (*Bartramia longicauda*), vesper sparrow (*Pooecetes gramineus*), western kingbird (*Tyrannus verticalis*), and western meadowlark (*Sturnella neglecta*; Table 21).

We identified three focal species as representative of three broader guilds of grassland birds. Within our study site in the northern mixed-grass prairie ecosystem, the grasshopper sparrow was representative of the dense grass guild of grassland birds (Dechant et al. 2002, Fritcher et al. 2004), the vesper sparrow was representative of the sparse grass guild (Browder et al. 2002), and the western meadowlark was representative of the generalist guild (Davis and Lanyon 2008). These three species have specific habitat requirements of native grasslands for breeding, recruitment, and survival throughout the season. These species also have the largest sample sizes within our study.

We found grasshopper sparrow detection probability was most influenced by Julian day, time of day, slope, and shrub height (Figure 13). After accounting for detection probability, the top local-scale habitat model describing local abundance of grasshopper sparrows included strong support (non-overlapping 95% CI's) for the standard deviation of visual obstruction reading (VOR), residual grass height, litter depth, bare ground coverage, and wooded coulee coverage (Figure 14). The standard deviation of VOR is often used as an index of heterogeneity in herbaceous cover, with larger standard deviation indicating higher heterogeneity and smaller standard deviation indicating lower heterogeneity. Grasshopper sparrow abundance showed a quadratic response to standard deviation VOR and bare ground coverage, where abundance decreased with increasing standard deviation VOR and decreased with increasing bare ground coverage beyond 10% (Figure 15). Grasshopper sparrow abundance also showed a quadratic response to residual grass height and litter depth, where abundance was maximized at ~15cm and ~3cm, respectively (Figure 15). Finally, grasshopper sparrow abundance showed a negative pseudo-threshold response to proportion wooded coulee within the point count area ($\beta = -0.337 \pm 0.038$; Figure 14, 15). The top landscape-scale habitat model describing grasshopper sparrow abundance included strong support (non-overlapping 95% CI's) for the proportion wooded coulee within a 1000m radial area from the survey point and the amount of fence within a 500m radial area from the point (Figure 14). Grasshopper sparrow abundance showed a quadratic response to increasing proportion coulee (1000m) and increasing fence length (500m; Figure 15).

We found vesper sparrow detection probabilities were most influenced by year and Julian day (Figure 16). After accounting for detection probability, the top local-level habitat model describing vesper sparrow abundance included strong support (non-overlapping 95% CI's) for litter depth, bare ground, shrub density, and wooded coulee coverage (Figure 17). Vesper sparrow abundance showed a positive response to increasing bare ground coverage (β = 0.235 ± 0.071), a quadratic response to litter depth where abundance was maximized at ~3 cm, and a positive pseudo-threshold (i.e., pseudo-asymptotic) response to increasing shrub density (β = 0.131 ± 0.051; Figure 17, 18). The top landscape-scale habitat model describing vesper sparrow abundance included strong support for the proportion wooded coulee within a 1000 m radial area from the survey point. Vesper sparrow abundance showed a positive pseudo-threshold response to increasing proportion coulee at the 1000 m scale (β = 0.157 ± 0.047; Figure 17, 18).

We found western meadowlark detection probabilities were most influenced by year, Julian day, and time of day (Figure 19). After accounting for detection probability, the top local-level habitat model describing western meadowlark abundance included strong support (non-overlapping 95% CI's) for visual obstruction, residual grass coverage, litter depth, and forb coverage (Figure 20). Western meadowlark abundance showed support for a quadratic effect of visual obstruction and residual grass coverage, but the effect was essentially a negative response to increasing visual obstruction and residual grass coverage (Figure 21). Western meadowlark abundance showed a positive pseudo-threshold response to increasing litter depth (β = 0.275 ± 0.045) and forb coverage (β = 0.100 ± 0.030; Figure 20, 21). The top landscape-scale habitat model describing western meadowlark abundance had support over the null model and included proportion coulee (1000 m scale), but did not show strong support for the effect (overlapping 95% confidence intervals; Figure 20).

Our candidate model set for management-level effects on bird abundance included effects of grazing system, stocking rate, and average rangeland productivity. The top model for grasshopper sparrow, which had the majority of model support (w_i = 0.86), included an interaction between grazing system and the quadratic effect of rangeland productivity, with the main quadratic effect of stocking rate (Table 22). Grasshopper sparrow abundance tended to increase with rangeland productivity (β = 0.38 ± 0.03); abundance was similar across grazing systems when productivity was moderate (1000 – 2000 lb · ac^{-1}), but tended to be highest in pastures managed with a season-long grazing system when rangeland productivity was low (<1000 lb · ac^{-1}) and high (>2000 lb · ac^{-1}; Figure 22). Grasshopper sparrow abundance showed a negative response to increasing stocking rate, and the effect was similar among grazing systems (Figure 23). Two management-level models shared support ($\Delta AIC_c < 2.0$) for effects on vesper sparrow abundance; the top model (w_i = 0.49), included an interaction between grazing system and the quadratic effect of stocking rate, with a main quadratic effect of average rangeland productivity, and the next top model included main effect of grazing system and main quadratic effect of stocking rate (Table 22). Vesper sparrow abundance decreased with increasing average rangeland productivity beyond 1000 lb · ac^{-1}, and the effect was similar among grazing systems (Figure 24). Vesper sparrow abundance showed a weak quadratic response to stocking rate, increasing with increasing stocking rate up to approximately 1 AUM · ha^{-1}, and decreasing beyond this rate (Figure 25). An interaction between grazing system and stocking rate was supported; however, season-long grazing systems were only stocked at rates between 1 and 2 AUM · ha^{-1}, so the resulting quadratic effect of stocking rate on vesper sparrow abundance within season-long grazing systems may be uninformative (Figure 25). Two management-level models shared support ($\Delta AIC_c < 2.0$) for effects on western meadowlark abundance; the top model (w_i = 0.55) included main effect of rangeland productivity and the main quadratic effect of stocking rate, and the next top model (w_i = 0.26) included an interaction between grazing system and rangeland productivity, with a main quadratic effect of stocking rate (Table 22). Western meadowlark abundance showed a positive pseudo-threshold response to rangeland productivity, where abundance increased with increasing productivity (β = 0.101 ± 0.034), and the effect was similar among grazing systems (Figure 26). Western meadowlark abundance showed a weak quadratic response to stocking rate, where abundance decreased with increasing stocking rates from 0 to 1 AUM · ha^{-1}, with increases in stocking rate beyond 1 AUM · ha^{-1} not strongly affecting western meadowlark abundance (Figure 27).

We evaluated the influence of grazing system on the local habitat conditions which grassland birds select. Using season-long grazing as the reference, we found a number of habitat conditions whose means were significantly lower or higher in rest-rotation or intensive summer-rotation systems based on non-overlapping 95% confidence intervals ($\bar{x}$ ± 95% CI; Table 23). Overall residual grass coverage was significantly lower in rest-rotation systems (37.687 ± 1.827) than season-long systems (47.644 ± 2.887). Bare ground coverage was significantly higher in rest-rotation (22.361 ± 1.588) and summer-rotation (19.834 ± 1.878) grazing systems than season-long systems (15.507 ± 2.189). Shrub density and shrub height were significantly higher in rest-rotation (0.064 ± 0.012 and 25.191 ± 1.862, respectively) and summer-rotation (0.062 ± 0.012 and 26.471 ± 2.162, respectively) grazing systems than season-long systems (0.035 ± 0.012 and 18.528 ± 2.354, respectively). Proportion wooded coulee on the landscape at the 100 m, 500 m, and 1000 m scales are significantly higher in rest-rotation and summer-rotation grazing systems than season-long systems. Fence length (km/ha) within 500 m and 1000 m of bird survey points was significantly higher in rest-rotation grazing systems than season-long or summer-rotation systems. Stocking rates were significantly lower in rest-rotation (0.919 ± 0.090) and summer-rotation (0.974 ± 0.033) systems than season-long grazing systems (1.281 ± 0.041). Average rangeland productivity was significantly lower in rest-rotation (1,224 ± 46) and summer-rotation (1,354 ± 46) grazing systems than season-long grazing systems (1,596 ± 52).

Goals for Next Quarter:

We complete all analyses of grassland bird survey data and will offer management recommendations for the effects of livestock grazing systems on grassland bird abundance based on results of our analysis. We will analyze all photos from the remote camera traps and identify predators based on body shape and coloration. Analyses of mesopredator camera trap data will be finalized, relating mesopredator occupancy to landscape-level, and management-level conditions using N-mixture modeling within R program unmarked. Results of the grassland bird analyses will be presented at the Society for Range Management's 71st Annual Meeting in Reno, Nevada at the beginning of February, 2018. A Master's thesis will be produced and circulated to project collaborators.

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Table 1. Demographic rates estimated for sharp-tailed grouse.

Demographic Rate	Description
Nesting rate (NEST)	The probability of a female initiating a nest.
Clutch size (CS)	The final clutch size per nest. Estimates generated for both first (CS_1) and renesting (CS_2) attempts.
Nest survival (NSURV)	The probability of a nest producing ≥1 chick.
Renesting rate (RENEST)	The probability of a female initiating a replacement nest after failure of the first attempt
Chicks per egg laid (CPE)	The proportion of eggs laid that produced chicks, or the viability of the eggs; calculated only for successful nests (≥1 egg hatched).
Brood survival (BSURV)	The probability that ≥1 chicks survived to fledging at 14-d post-hatch
Fledglings per chick hatched (FPC)	The proportion of hatched chicks that survived to fledging conditional upon brood survival

Table 2. Average attendance at 7 leks during 15 March – 5 May 2017. The four leks located within the easement are listed first.

Lek	Average Total Attendance	Minimum Total Attendance	Maximum Total Attendance	Average Male Attendance	Average Female Attendance
EasState1	14.0	7.0	26.0	12.9	1.6
Prewitt1	15.4	8.0	25.0	13.0	2.7
Laumeyer2	18.6	12.0	27.0	15.6	3.4
OilpadLek	8.0	3.0	18.0	6.1	1.9
Pennington01	16.6	9.0	35.0	13.9	2.7
Iversen1	14.9	3.0	24.0	12.9	1.6
Ullman01	18.2	11.0	29.0	16.2	1.9
Total	14.4	3.0	35.0	12.6	2.1

Table 3. Total number of grouse captured and radio-marked on and off the easement in 2017. The total radio-marked females includes females radio-marked in 2016 but monitored again in 2017

	Males	Females	New Radio-marked Females	Total Radio-marked Females
Easement	44	40	29	39
Reference	43	29	29	32
Total	87	69	58	71

Table 4. Overview of sharp-tailed grouse nests in pastures managed with different grazing systems in 2017. Egg hatch rate (± SE) is the percentage of eggs that hatched from the initial clutch size.

	Median Initiation Date	**Clutch Size**	**First Nests**	**Renests**	**Nests Hatched**	**Median Hatch Date**	**Egg Hatch Rate**
Rest-rotation	6 May	10.4 ± 4.3	22	4	13	7 June	0.90 ± 0.14
Winter	29 April	11.1 ± 3.1	3	1	2	3 June	0.71 ± 0.39
Summer rotation	29 April	10.5 ± 3.7	23	4	12	7 June	0.92 ± 0.12
Season-long	8 May	10.6 ± 4.1	12	7	10	11 June	0.93 ± 0.08
Total	2 May	10.5 ± 3.9	60	16	37	7 June	0.90 ± 0.15

Table 5. Estimated demographic rates (± SE) for female sharp-tailed grouse during the 2016–17 breeding seasons.

Demographic Rate	Estimate ± SE
Nesting rate (NEST)	1
Clutch size - first nest (CS1)	11.38 ± 0.37
Clutch size - renests (CS2)	9.53 ± 0.43
Nest survival (NSURV)	0.22 ± 0.06
Renesting rate (RENEST)	0.64 ± 0.04
Chicks per egg laid (CPE)	0.91 ± 0.02
Brood survival (BSURV)	0.56 ± 0.06
Fledglings per chick hatched (FPC)	0.47 ± 0.05

Table 6. Support for candidate models predicting sharp-tailed grouse nest survival in 2016–17 based on underlying variables. The number of parameters (K), AIC_c values, ΔAIC_c values, model weights (w_i) and deviance are reported.

Model	**K**	**AIC_c**	**ΔAIC_c**	**AIC_c w_i**	**Deviance**
Day of Nesting Season	2	550.59	0.00	0.32	546.59
Flush Nest	2	552.11	1.51	0.15	548.10
Nest Attempt	2	552.21	1.61	0.14	548.20
Female Age + Nest Attempt	3	552.69	2.09	0.11	546.67
Year	2	553.04	2.45	0.09	549.03
Null	1	553.40	2.81	0.08	551.40
Female Age	2	554.21	3.62	0.05	550.21
Female Condition	2	555.18	4.59	0.03	551.18

Female Condition + Age	3	555.72	5.12	0.02	549.71

Table 7. Support for candidate models predicting sharp-tailed grouse nest survival in 2016–17 at the nest site scale. The number of parameters (K), AIC_c values, ΔAIC_c values, model weights (w_i) and deviance are reported.

Model	**K**	**AIC_c**	**ΔAIC_c**	**$\text{AIC}_c\ w_i$**	**Deviance**
ln(VOR)	2	549.82	0.00	0.34	545.81
ln(VOR) + %Forb	3	551.42	1.60	0.15	545.41
ln(VOR) + %Residual	3	551.52	1.70	0.15	545.51
ln(VOR) + %Grass	3	551.75	1.94	0.13	545.74
Null	1	553.40	3.58	0.06	551.40
Dist. to Grassland Edge	2	554.91	5.09	0.03	550.90
%Forb	2	554.96	5.15	0.03	550.96
%Residual	2	554.98	5.16	0.03	550.97
%Grass	2	555.06	5.25	0.02	551.06
ln(VOR) + %Grass + %Residual + %Forb	5	555.08	5.27	0.02	545.06
%Bare	2	555.22	5.40	0.02	551.21
%Grass + %Residual + %Shrub	4	556.29	6.48	0.01	548.28
%Grass + %Residual + %Shrub + %Forb	5	558.10	8.28	0.01	548.07

Table 8. Support for candidate models predicting sharp-tailed grouse nest survival in 2016–17 at the home range scale. The number of parameters (K), AIC_c values, ΔAIC_c values, model weights (w_i) and deviance are reported.

Model	**K**	**AIC_c**	**ΔAIC_c**	**$\text{AIC}_c\ w_i$**	**Deviance**
Null	1	553.40	0.00	0.38	551.40
Edge Density	2	554.97	1.57	0.17	550.96
Prop. Grassland	2	555.03	1.63	0.17	551.02
Mean Shape Complexity	2	555.34	1.94	0.14	551.33
Dist. to Oil Pad	2	555.36	1.96	0.14	551.35

Table 9. Support for final candidate models predicting sharp-tailed grouse nest survival in 2016–17. Important variables representing underlying influences, habitat at the nest and home-range level, and grazing are included. The number of parameters (K), AIC_c values, ΔAIC_c values, model weights (w_i) and deviance are reported.

Model	**K**	**AIC_c**	**ΔAIC_c**	**AIC_c w_i**	**Deviance**
Day of Nesting Season + ln(VOR)	3	545.79	0.00	0.49	539.78
Nest Attempt + ln(VOR)	3	548.76	2.97	0.11	542.75
Flush Nest + ln(VOR)	3	549.03	3.24	0.10	543.02
Female Age + Nest Attempt + ln(VOR)	4	549.75	3.96	0.07	541.74
ln(VOR)	2	549.82	4.02	0.07	545.81
Day of Nesting Season	2	550.59	4.80	0.04	546.59
Year + ln(VOR)	3	550.64	4.84	0.04	544.63
Flush Nest	2	552.11	6.31	0.02	548.10
Nest Attempt	2	552.21	6.41	0.02	548.20
Female Age + Nest Attempt	3	552.69	6.89	0.02	546.67
Year	2	553.04	7.25	0.01	549.03
Null	1	553.40	7.61	0.01	551.40
Stocking Rate	2	554.69	8.90	0.01	550.69
Grazing System	3	556.65	10.86	0.00	550.64

Table 10. Support for candidate models evaluating effects of grazing management on sharp-tailed grouse nest survival in 2016–17. Grazing management variables from both the current and the previous year are included. The number of parameters (K), AICc values, ΔAICc values, model weights (wi) and deviance are reported.

Model	**K**	**AIC_c**	**ΔAIC_c**	**AIC_c w_i**	**Deviance**
Day of Nesting Season + ln(VOR) + Grazing Duration (current)	4	545.06	0.00	0.26	537.04
Day of Nesting Season + ln(VOR)	3	545.79	0.73	0.18	539.78
Day of Nesting Season + ln(VOR) + Stocking Rate (current)	4	546.87	1.81	0.11	538.86
Day of Nesting Season + ln(VOR) + Grazing Duration (previous)	4	546.94	1.88	0.10	538.92
Day of Nesting Season + ln(VOR) + Stocking Rate (previous)	4	546.99	1.93	0.10	538.97
Day of Nesting Season + ln(VOR) + Stocking Intensity (current)	4	547.68	2.62	0.07	539.66
Day of Nesting Season + ln(VOR) + Stocking Intensity (previous)	4	547.79	2.73	0.07	539.78
Day of Nesting Season + ln(VOR) + Grazing Season (current)	6	548.05	2.99	0.06	536.02
Day of Nesting Season + ln(VOR) + Grazing Season (previous)	6	548.77	3.71	0.04	536.74

Table 11. Support for candidate models evaluating effects of flushing on sharp-tailed grouse nest survival in 2016–17. The number of parameters (K), AIC_c values, ΔAIC_c values, model weights (w_i) and deviance are reported.

Model	**K**	**AIC_c**	**ΔAIC_c**	**w_i**	**Deviance**
Flush Bird	2	454.81	0	0.32	450.8
Flush Bird + Day of Nesting Season	3	454.94	0.14	0.3	448.93
Flush Bird + Female Age	3	456.31	1.5	0.15	450.29
Null	1	457.86	3.05	0.07	455.86
Flush Bird x Female Age	4	458.12	3.31	0.06	450.1
Day of Nesting Season	2	458.5	3.69	0.05	454.49
Female Age	2	459.7	4.89	0.03	455.69
Day of Nesting Season + Female Age	3	460.31	5.5	0.02	454.29

Table 12. Sharp-tailed grouse brood success (± SE) to fledging at 14-d post hatch in the easement and reference sections of the study area in 2017. Brood success is the proportion of broods that successfully fledged ≥1 chick. Fledging rate is the proportion of chicks within broods that survived to fledging at 14 days.

	Number of Broods	**Brood Success**	**Fledging Rate**
Easement	6	0.17 ± 0.30	033 ± 0.14
Reference	25	0.68 ± 0.09	0.52 ± 0.09
Both	6	0.67 ± 0.19	0.52 ± 0.10
Total	37	0.59 ± 0.08	0.51 ± 0.07

Table 13. Support for candidate models predicting sharp-tailed grouse nest site selection at the nest-site level in 2016–17. The number of parameters (K), AIC values, ΔAIC values, and log-likelihoods are reported.

Model	**K**	**AIC**	**ΔAIC**	**LogLik**
Nest VOR + Nest VOR^2 + % New Grass + % Residual Grass + % Shrub	6	699.44	0.00	-343.72
Nest VOR + Nest VOR^2 + % Grass	4	740.43	40.99	-366.22
Nest VOR + Nest VOR^2 + % Shrub	4	758.00	58.56	-375.00
Nest VOR + Nest VOR^2 + % Residual Grass	4	773.52	74.08	-382.76
Nest VOR + Nest VOR^2	3	791.46	92.02	-392.73
Nest VOR + Nest VOR^2 + % Forb	4	792.26	92.82	-392.13
Shrub Height	2	1159.85	460.40	-577.92
% Bare Ground	2	1183.06	483.62	-589.53
% New Grass + % Residual Grass + % Shrub	4	1192.20	492.76	-592.10

% New Grass + % Forb + % Residual Grass + % Shrub	5	1193.53	494.09	-591.77
% Shrub	2	1209.84	510.40	-602.92
Forb Height	2	1231.42	531.98	-613.71
% Residual Grass	2	1235.06	535.62	-615.53
% New Grass	2	1247.65	548.21	-621.82
Null	1	1248.09	548.65	-623.04
% Forb	2	1248.97	549.53	-622.48
Distance to Grassland Edge	2	1249.84	550.39	-622.92

Table 14. Support for candidate models predicting sharp-tailed grouse nest site selection at the home range level in 2016–17. The number of parameters (K), AIC values, ΔAIC values, and log-likelihoods are reported.

Model	**K**	**AIC**	**ΔAIC**	**LogLik**
Edge Density	2	1244.38	0.00	-620.19
Null	1	1248.09	3.71	-623.04
Distance to Oilpad	2	1248.93	4.55	-622.46
Grassland Shape Complexity	2	1249.95	5.57	-622.97
Distance to Lek	2	1250.04	5.66	-623.02
Prop. Grassland	2	1250.08	5.70	-623.04

Table 15. Support for final candidate models predicting sharp-tailed grouse nest site selection in 2016–17. Important variables representing habitat at the nest and home-range level and grazing management are included. The number of parameters (K), AIC values, ΔAIC values, and log-likelihoods are reported.

Model	**K**	**AIC**	**ΔAIC**	**LogLik**
Nest VOR + Nest VOR^2 + % New Grass + % Residual Grass + % Shrub + Edge Density	7	643.00	0.00	-314.50
Nest VOR + Nest VOR^2 + % New Grass + % Residual Grass + % Shrub	6	699.44	56.44	-343.72
Edge Density	2	1244.38	601.38	-620.19
Null	1	1248.09	605.09	-623.04
Stocking Rate	2	1249.64	606.64	-622.82
Grazing System	3	1250.54	607.54	-622.27

Table 16. Estimated breeding season survival for females in each of the three grazing treatments. Females were assigned to the grazing system in which their home range was centered or the system which contained the majority of locations if there were insufficient locations to calculate a home range. Estimates, standard error, and lower and upper 85% confidence intervals (LCI and UCI, respectively) are reported.

Model	**Estimate**	**SE**	**LCI**	**UCI**
Season-long	0.693	0.08	0.58	0.81
Rotation	0.554	0.09	0.42	0.68
Rest-rotation	0.686	0.07	0.59	0.79

Table 17. Simplified ranking matrix of female sharp-tailed grouse breeding season habitat selection in 2016 and 2017. Matrix is based on comparing proportional habitat use within home ranges with proportion of available habitat types. Mixed grass prairie is defined by soil texture and includes areas where soils are primarily fine and medium-textured and do not include sands, sandy soils, or sandy loams. Dominant graminoid species include *Pascopyrum smithii*, *Nassella viridula*, *Hesperostipa comata,* and *Festuca* spp. Shrubland is distinguished from mixed grass prairie in that it contains greater than 10% shrub cover in combination with topographic relief. Close grown crops and wheat are defined as areas where crop vegetation accounts for greater than 20% of total vegetation; close grown crops include land being actively tilled. Introduced upland perennial grasslands include areas that are dominated by introduced perennial forb or grassland species including such as *Agropyron cristatum, Bromus inermis*, and *Poa pratensis*. Developed ruderal grasslands are defined as herbaceous vegetation resulting from succession following a significant anthropogenic disturbance and are usually characterized by unnatural combinations of species.

	Mixed grass prairie	Close grown crop	Shrubland	Introduced upland perennial grasslands	Wheat	Developed ruderal grasslands	RANK
Mixed grass prairie	0	+	+++	+++	+++	+++	1
Close grown crop	-	0	+	+	+++	+++	2
Shrubland	---	-	0	+	+++	+++	3
Introduced upland perennial grasslands	---	-	-	0	+++	+++	4

Wheat	---	---	---	---	0	+++	5
Developed ruderal grasslands	---	---	---	---	---	0	6

Table 18. Support for candidate models predicting home range size of female sharp-tailed grouse during the breeding seasons of 2016 and 2017.

Model	**K**	**AIC_c**	**ΔAIC_c**	**AIC_c w_i**	**LL**
Dist. to Grassland Edge	3	1436.62	0.00	0.26	-715.18
Null	2	1437.02	0.40	0.21	-716.44
Dist. to Lek	3	1437.66	1.04	0.15	-715.70
Nest Outcome	4	1438.28	1.66	0.11	-714.91
Year	3	1438.66	2.04	0.09	-716.20
Dist. to Road	3	1438.97	2.35	0.08	-716.35
Dist. to Oil Pad	3	1439.12	2.50	0.07	-716.43
Grazing System	5	1443.16	6.54	0.01	-716.23

Table 19. Bird detections and species diversity from 915 point count surveys at 305 survey sites on the Buxbaum conservation easement and adjacent reference properties in eastern Richland County, Montana in 2017.

	Buxbaum Conservation Easement[a]				Reference Pastures[b]					Total
	[1] Pastures	[2] Pastures	[3] Pastures	**Subtotal**	Season-long	Rotation1	Rotation2	Rotation3	**Subtotal**	
Number points	50	50	50	**150**	60	59	21	15	**155**	305
Total birds	1109	1189	1064	**3362**	1175	1404	595	339	**3513**	6875
Mean birds / Point	22.2	23.8	21.3	**22.4**	19.6	23.8	28.3	22.6	**22.7**	22.5
Number species	43	42	44	**56**	39	39	38	27	**52**	62

[a] Easement pasture designations: [1] A1, B3, C1; [2] A2, B1, C2; [3] A3, B2, C3

[b] Reference Pastures include 2 pastures that are grazed annually during the growing season (season-long), and three pastures managed with intensive summer rotational grazing.

Table 20. Total birds of each species detected during 915 point count surveys at 305 survey sites on the Buxbaum conservation easement and adjacent reference properties during the 2017 field season.

	Buxbaum Conservation Easement			**Total**	Reference Pastures				**Total**	Grand Total
	[1] Pastures	[2] Pastures	[3] Pastures		Season-long	Rotation1	Rotation2	Rotation3		
AMCR	0	0	0	**0**	0	2	0	0	**2**	2
AMGO	27	23	23	**73**	6	15	31	5	**57**	130
AMKE	2	1	1	**4**	2	1	0	0	**3**	7
AMRO	8	4	11	**23**	1	3	12	4	**20**	43
BAIS	0	0	0	**0**	21	29	0	0	**50**	50
BANS	2	0	2	**4**	0	1	0	0	**1**	5
BAOR	0	0	0	**0**	3	0	0	0	**3**	3
BARS	8	4	1	**13**	1	1	2	0	**4**	17
BBMA	1	0	0	**1**	1	4	0	0	**5**	6

BHCO	136	92	55	**283**	48	44	55	14	**161**	444
BHGR	4	1	0	**5**	0	0	0	0	**0**	5
BOBO	0	1	5	**6**	1	0	0	14	**15**	21
BRBL	9	12	36	**57**	30	0	6	2	**38**	95
BRSP	3	0	0	**3**	0	0	0	0	**0**	3
BRTH	24	8	10	**42**	6	3	9	3	**21**	63
BUOR	11	5	3	**19**	0	1	2	1	**4**	23
CCSP	8	51	2	**61**	7	50	3	2	**62**	123
CEDW	0	1	4	**5**	1	0	5	3	**9**	14
CHSP	3	0	7	**10**	0	0	3	0	**3**	13
CLSW	2	0	1	**3**	0	0	0	0	**0**	3
COGR	0	6	0	**6**	3	0	1	2	**6**	12
CONI	1	3	2	**6**	0	6	2	0	**8**	14
COYE	0	4	0	**4**	0	3	0	0	**3**	7
EABL	0	1	0	**1**	0	0	0	0	**0**	1
EAKI	33	52	31	**116**	25	24	31	13	**93**	209
EUST	5	1	2	**8**	1	2	2	0	**5**	13
FISP	50	23	13	**86**	5	51	35	10	**101**	187
GRCA	0	2	0	**2**	0	0	2	1	**3**	5
GRSP	91	331	323	**745**	435	472	16	115	**1038**	1783
HAWO	1	0	0	**1**	0	0	0	0	**0**	1
HOLA	1	1	19	**21**	16	2	3	2	**23**	44
HOWR	39	24	13	**76**	5	19	33	9	**66**	142
KILL	0	0	3	**3**	0	3	3	0	**6**	9
LARB	31	1	1	**33**	0	0	0	0	**0**	33
LASP	15	7	11	**33**	5	2	11	0	**18**	51
LEFL	0	3	1	**4**	0	5	5	1	**11**	15
LOSH	3	7	3	**13**	1	3	0	1	**5**	18
MAGO	0	0	1	**1**	0	0	0	0	**0**	1
MOBL	7	1	2	**10**	2	7	6	5	**20**	30
MODO	30	21	29	**80**	31	20	38	9	**98**	178

NOFL	12	6	6	**24**	5	9	10	4	**32**	56
NRWS	5	8	1	**14**	1	7	3	0	**11**	25
OROR	0	3	3	**6**	1	3	2	0	**6**	18
PRFA	0	0	0	**0**	0	0	1	0	**1**	1
RNEP	1	3	1	**5**	0	0	1	0	**1**	6
ROPI	2	0	0	**2**	0	0	0	0	**0**	2
ROWR	9	2	0	**11**	2	1	0	0	**3**	14
RTHA	0	0	0	**0**	0	0	0	1	**1**	1
RWBL	0	3	5	**8**	17	4	0	1	**22**	30
SAPH	8	0	0	**8**	0	1	1	0	**2**	10
SPPI	1	3	13	**17**	9	1	0	0	**10**	27
SPTO	64	17	16	**97**	9	36	31	7	**83**	180
STGR	0	1	2	**3**	4	2	1	0	**7**	10
TRES	0	0	2	**2**	1	0	0	0	**1**	3
UPSA	6	4	10	**20**	15	10	0	7	**32**	52
VESP	89	79	71	**239**	60	142	48	18	**268**	507
WEKI	25	8	13	**47**	2	4	16	0	**22**	69
WEME	270	320	284	**874**	384	381	116	73	**954**	1828
WEWP	2	0	0	**2**	0	0	0	0	**0**	2
WIFL	0	0	0	**0**	0	0	1	0	**1**	1
YBCH	11	3	1	**15**	0	10	14	0	**24**	39
YWAR	49	40	21	**110**	5	22	33	14	**74**	184

Table 21. Bird species observed during 2017 avian point count surveys at 305 survey sites located on the Buxbaum conservation easement and adjacent reference properties.

4-letter Code	Common Name	Scientific Name
AMCR	American Crow	*Corvus brachyrhynchos*
AMGO	American Goldfinch	*Carduelis tristis*
AMKE	American Kestrel	*Falco sparverius*

AMRO	American Robin	*Turdus migratorius*
BAIS*	Baird's Sparrow	*Ammodramus bairdii*
BANS	Bank Swallow	*Riparia riparia*
BAOR	Baltimore Oriole	*Icterus galbula*
BARS	Barn Swallow	*Hirundo rustica*
BBMA	Black-billed Magpie	*Pica hudsonia*
BHCO	Brown-headed Cowbird	*Molothrus ater*
BHGR	Black-headed Grosbeak	*Pheucticus melanocephalus*
BOBO*	Bobolink	*Dolichonyx oryzivorus*
BRBL	Brewer's Blackbird	*Euphagus cyanocephalus*
BRSP*	Brewer's Sparrow	*Spizella breweri*
BRTH	Brown Thrasher	*Toxostoma rufum*
BUOR	Bullock's Oriole	*Icterus bullockii*
CCSP*	Clay-colored Sparrow	*Spizella pallida*
CEDW	Cedar Waxwing	*Bombycilla cedrorum*
CHSP	Chipping Sparrow	*Spizella passerina*
CLSW	Cliff Swallow	*Petrochelidon pyrrhonota*
COGR	Common Grackle	*Quiscalus quiscula*
CONI	Common Nighthawk	*Chordeiles minor*
COYE*	Common Yellowthroat	*Geothlypis trichas*
EABL*	Eastern Bluebird	*Sialia sialis*
EAKI*	Eastern Kingbird	*Tyrannus tyrannus*
EUST	European Starling	*Sturnus vulgaris*
FISP*	Field Sparrow	*Spizella pusilla*
GRCA	Gray Catbird	*Dumetella carolinensis*
GRSP*	Grasshopper Sparrow	*Ammodramus savannarum*
HAWO	Hairy Woodpecker	*Leuconotopicus villosus*
HOLA*	Horned Lark	*Eremophila alpestris*
HOWR	House Wren	*Troglodytes aedon*
KILL*	Killdeer	*Charadrius vociferus*
LARB*	Lark Bunting	*Calamospiza melanocorys*

LASP*	Lark Sparrow	*Chondestes grammacus*
LEFL	Least Flycatcher	*Empidonax minimus*
LOSH*	Loggerhead Shrike	*Lanius ludovicianus*
MAGO*	Marbled Godwit	*Limosa fedoa*
MOBL*	Mountain Bluebird	*Sialia currucoides*
MODO	Mourning Dove	*Zenaida macroura*
NOFL	Northern Flicker	*Colaptes auratus*
NRWS	Northern Rough-Winged Swallow	*Stelgidopteryx serripennis*
OROR	Orchard Oriole	*Icterus spurius*
PRFA*	Prairie Falcon	*Falco maxicanus*
RNEP	Ring-necked Pheasant	*Phasianus colchicus*
ROPI	Rock Pigeon	*Columba livia*
ROWR	Rock Wren	*Salpinctes obsoletus*
RTHA	Red-tailed Hawk	*Buteo jamaicensis*
RWBL*	Red-winged Blackbird	*Agelaius phoeniceus*
SAPH	Say’s Phoebe	*Sayornis saya*
SPPI*	Sprague’s Pipit	*Anthus spragueii*
SPTO	Spotted Towhee	*Pipilo maculatus*
STGR*	Sharp-tailed Grouse	*Tympanuchus phasianellus*
TRES	Tree Swallow	*Hirundo nigricans*
UPSA*	Upland Sandpiper	*Bartramia longicauda*
VESP*	Vesper Sparrow	*Pooecetes gramineus*
WEKI*	Western Kingbird	*Tyrannus verticalis*
WEME*	Western Meadowlark	*Sturnella neglecta*
WEWP	Western Wood-Pewee	*Contopus sordidulus*
WIFL	Willow Flycatcher	*Empidonax traillii*
YBCH	Yellow-breasted Chat	*Icteria virens*
YWAR	Yellow Warbler	*Dendroica petechia*

*Designates grassland obligate species.

Table 22. Support for candidate models predicting management-scale effects on grassland bird abundance. Included are the effects of grazing system, stocking rate, and average rangeland productivity. The number of parameters (K), AIC_c values, ΔAIC_c values, model weights (w_i), and cumulative model weights (Cum w_i) are reported.

Model	**K**	**AIC_c**	**ΔAIC_c**	**w_i**	**Cum w_i**
Grasshopper Sparrow					
Grazing System × Rangeland Productivity^2 + Stocking Rate^2	18	4170.88	0.00	0.86	0.86
Rangeland Productivity^2 + Stocking Rate^2	14	4175.45	4.57	0.09	0.95
Grazing System + Rangeland Productivity^2 + Stocking Rate^2	16	4177.81	6.93	0.03	0.98
Grazing System × Stocking Rate^2 + Rangeland Productivity^2	18	4179.15	8.27	0.01	0.99
Null Model	10	4380.73	209.85	0.00	1.00
Vesper Sparrow					
Grazing System × Stocking Rate^2 + Rangeland Productivity^2	16	3039.26	0.00	0.49	0.49
Grazing System + Rangeland Productivity^2	12	3040.79	1.53	0.23	0.71
Grazing System + Rangeland Productivity^2 + Stocking Rate^2	14	3041.61	2.35	0.15	0.87
Rangeland Productivity^2 + Stocking Rate^2	12	3043.76	4.50	0.05	0.92
Grazing System × Stocking Rate^2	14	3044.70	5.45	0.03	0.95
Rangeland Productivity^2	10	3045.85	6.59	0.02	0.97
Grazing System × Rangeland Productivity^2 + Stocking Rate^2	16	3046.18	6.92	0.02	0.98
Null Model	8	3057.49	8.49	0.00	1.00
Western Meadowlark					
Stocking Rate^2 + ln(Rangeland Productivity)	17	4887.40	0.00	0.55	0.55
Grazing System × ln(Rangeland Productivity) + Stocking Rate^2	19	4888.86	1.46	0.26	0.81
Grazing System + Stocking Rate^2 + ln(Rangeland Productivity)	19	4891.01	3.61	0.09	0.90
Grazing System × Stocking Rate^2 + ln(Rangeland Productivity)	21	4892.29	4.89	0.05	0.95
Stocking Rate^2	16	4894.72	7.33	0.01	0.96
ln(Rangeland Productivity)	15	4894.98	7.58	0.01	0.97
Null Model	14	4903.68	16.28	0.00	1.00

^{2} Variable indicates support for a quadratic effect. ln(variable) indicates support for a pseudo-threshold effect.

Table 23. Local habitat conditions among season-long, rest-rotation, and intensive summer rotation grazing systems.

	Season-long (n = 120)				Rest-Rotation (n = 300)				Summer-Rotation (n = 190)			
	x̄	SE	Min.	Max	x̄	SE	Min.	Max	x̄	SE	Min.	Max
Visual Obstruction (VOR)	0.916	0.047	0.000	2.533	0.905	0.036	0.067	5.000	0.951	0.045	0.067	5.067
Standard Deviation of VOR	0.614	0.043	0.000	3.380	0.752	0.041	0.176	4.627	0.722	0.043	0.176	4.626
% Residual Grass Coverage	47.644	1.473	8.333	83.500	37.687*	0.932	3.000	81.167	42.052	1.193	3.333	81.333
Residual Grass Height (cm)	14.361	0.468	3.533	32.467	13.345	0.262	3.467	30.933	13.685	0.311	4.667	26.133
% Forb Coverage	10.349	0.375	1.333	19.333	11.040	0.316	1.833	53.167	9.819	0.318	0.667	26.333
Litter Depth (cm)	1.588	0.069	0.400	4.333	1.342	0.039	0.000	5.133	1.429	0.045	0.233	3.433
% Bare Ground Coverage	15.507	1.117	0.000	71.167	22.361*	0.810	0.333	85.333	19.834*	0.958	0.167	68.833
Shrub Density (proportion)	0.035	0.006	0.000	0.485	0.064*	0.006	0.000	0.690	0.062*	0.006	0.000	0.448
Shrub Height (cm)	18.528	1.201	0.000	59.327	25.191*	0.950	0.000	85.000	26.471*	1.103	0.000	76.739
% Wooded Coulee (100m)	0.005	0.002	0.000	0.169	0.018*	0.002	0.000	0.263	0.019*	0.003	0.000	0.245
% Wooded Coulee (500m)	0.008	0.001	0.000	0.045	0.020*	0.001	0.000	0.069	0.024*	0.001	0.000	0.110
% Wooded Coulee (1000m)	0.014	0.001	0.000	0.037	0.020*	0.001	0.005	0.048	0.026*	0.001	0.009	0.075
Fence Length (m/ha; 500m)	6.3043	0.595	0.000	22.115	10.543*	0.405	0.000	30.264	8.2928	0.532	0.000	28.407
Fence Length (m/ha; 1000m)	8.659	0.356	0.000	20.010	12.934*	0.244	4.786	26.376	9.396	0.392	0.000	17.883
Stocking Rate (AUM·ha^{-1})	1.281	0.021	0.768	1.458	0.919*	0.046	0.000	2.543	0.974*	0.017	0.227	1.394
Rangeland Productivity (lb·ac^{-1})	1596.2	26.3	916.0	2327.0	1224.4*	23.5	176.0	2342.0	1353.7*	23.4	550.0	2131.0

*Indicates habitat conditions with significantly lower or higher mean values in reference to season-long grazing system, based on non-overlapping 95% confidence intervals.

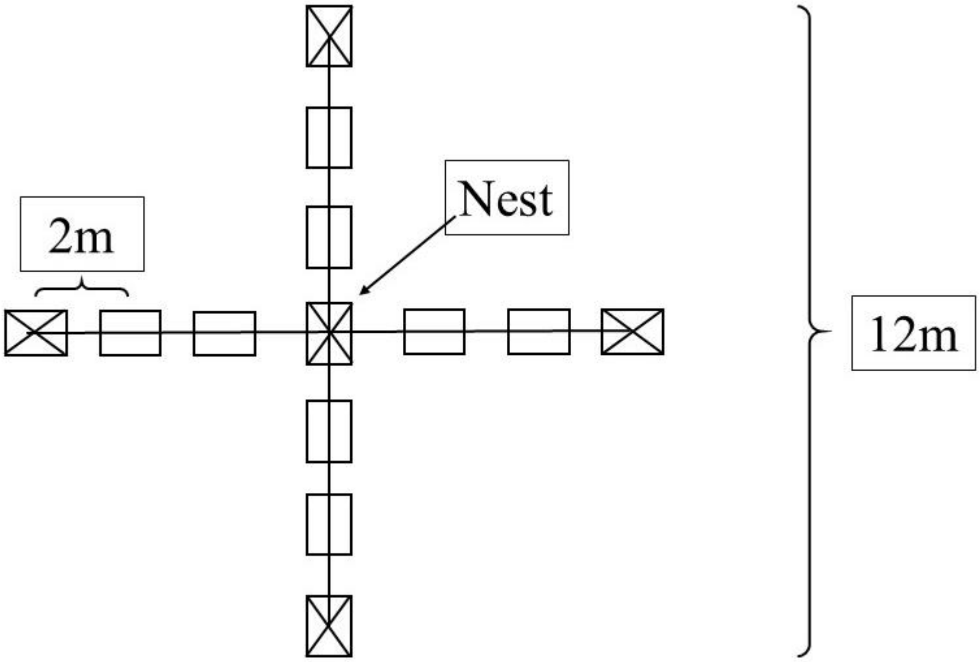

Figure 1. Setup of a vegetation plot. Vegetation cover and height were measured using a Daubenmire frame at each rectangle and visual obstruction with a Robel pole at each X. The lines represent the 12 m transects that were used to estimate shrub cover with the line-intercept method.

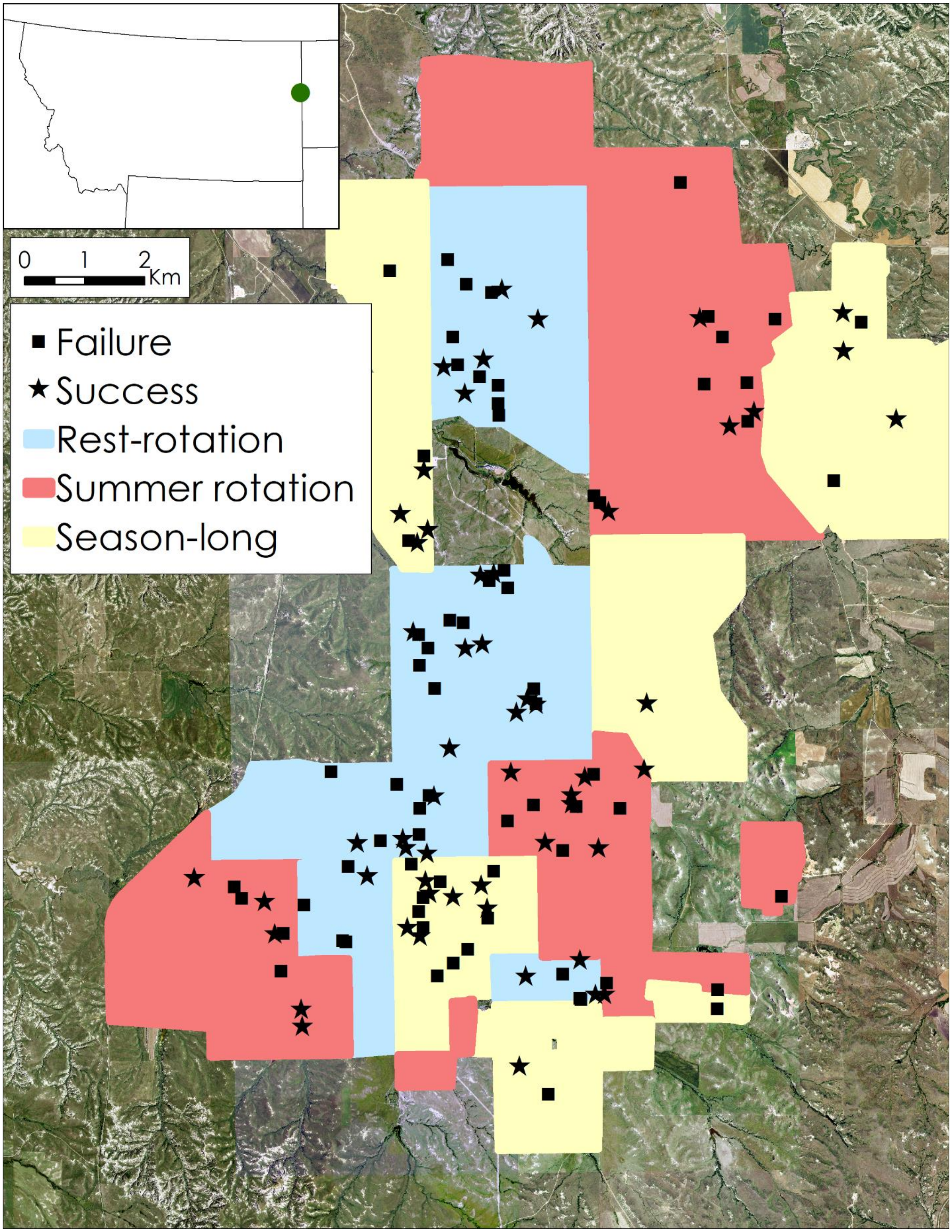

Figure 2. Locations of successful (stars) and failed (squares) sharp-tailed grouse nests in 2016–17 in relation to different grazing treatments.

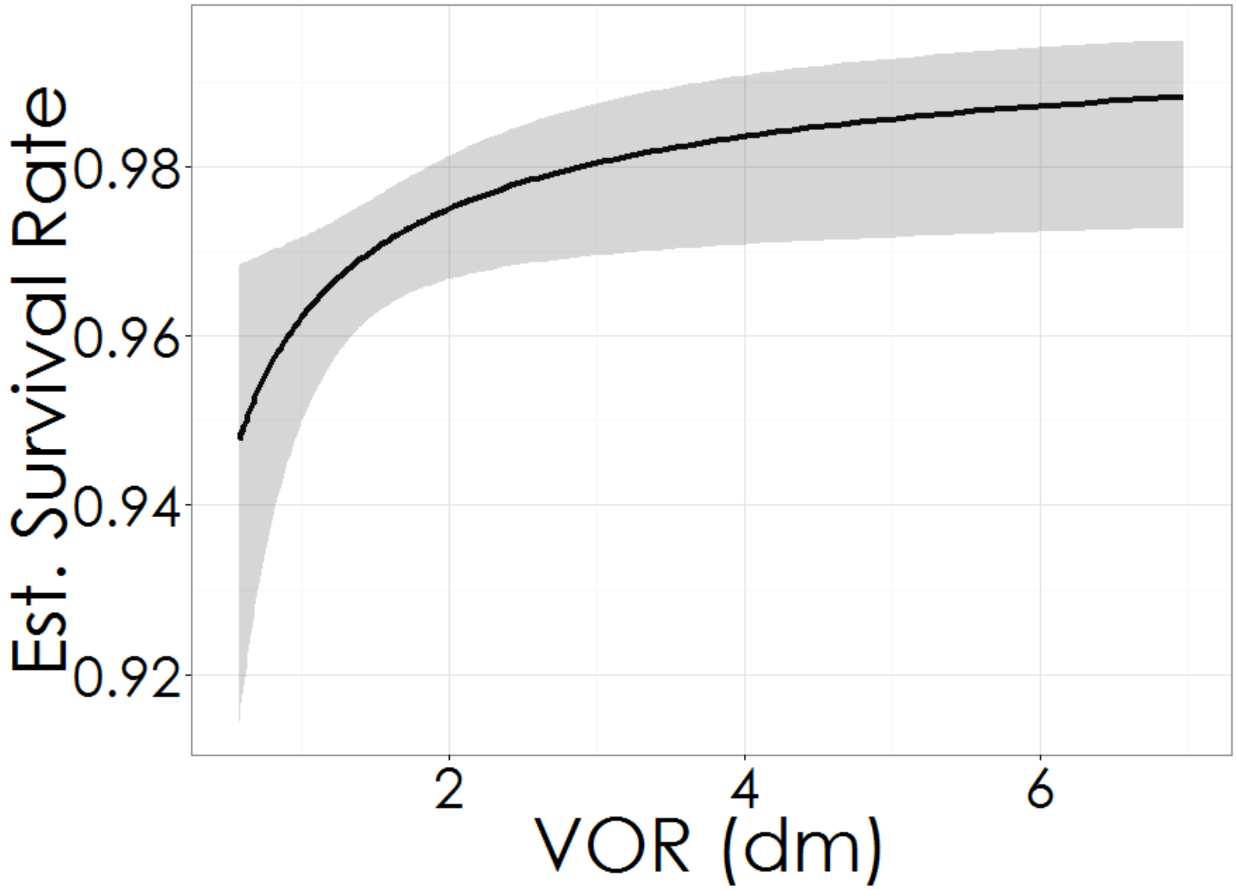

Figure 3. Estimated daily nest survival in relation to average visual obstruction measured within 6 m of the nest bowl, with 85% confidence intervals shown in grey.

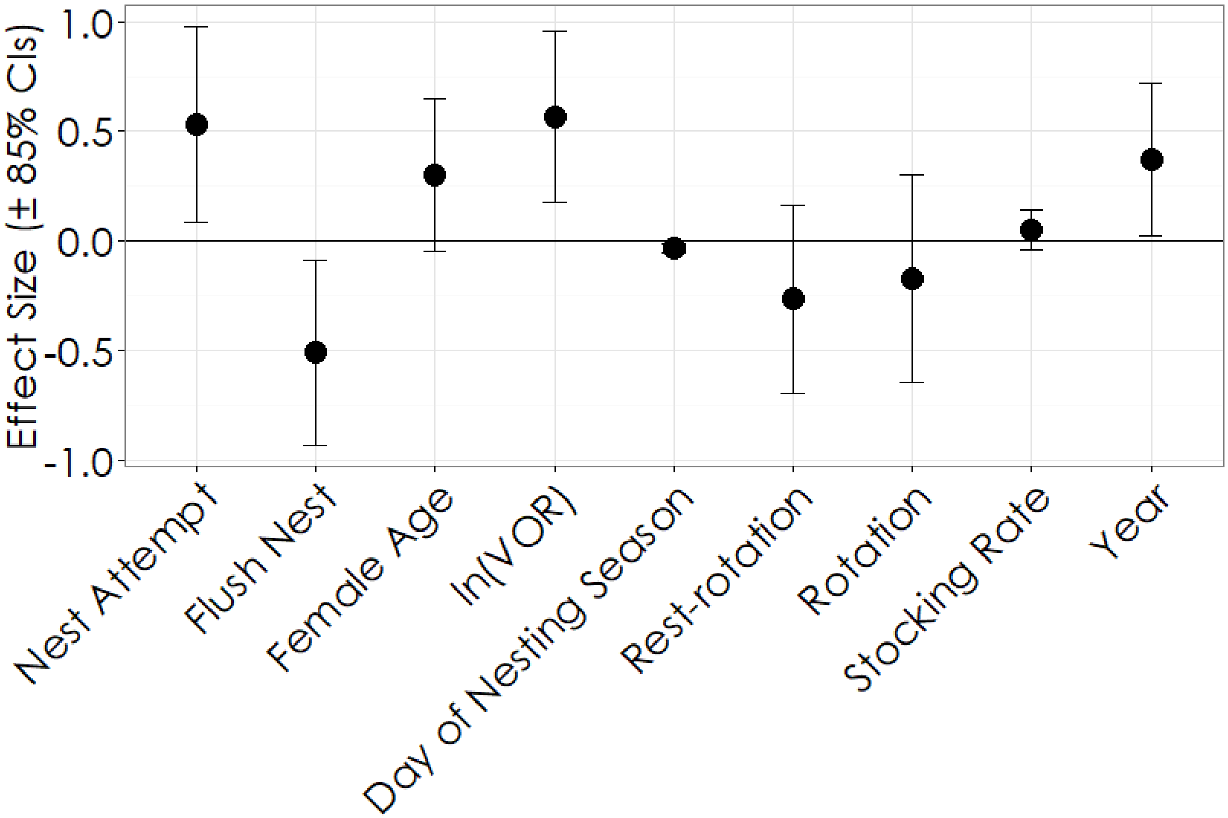

Figure 4. Effect size (*β* ± 85% confidence intervals) for each variable in the analysis examining sharp-tailed grouse nest survival in 2016–17. Parameters for each grazing system represent effect sizes in relation to the reference category of season-long grazing.

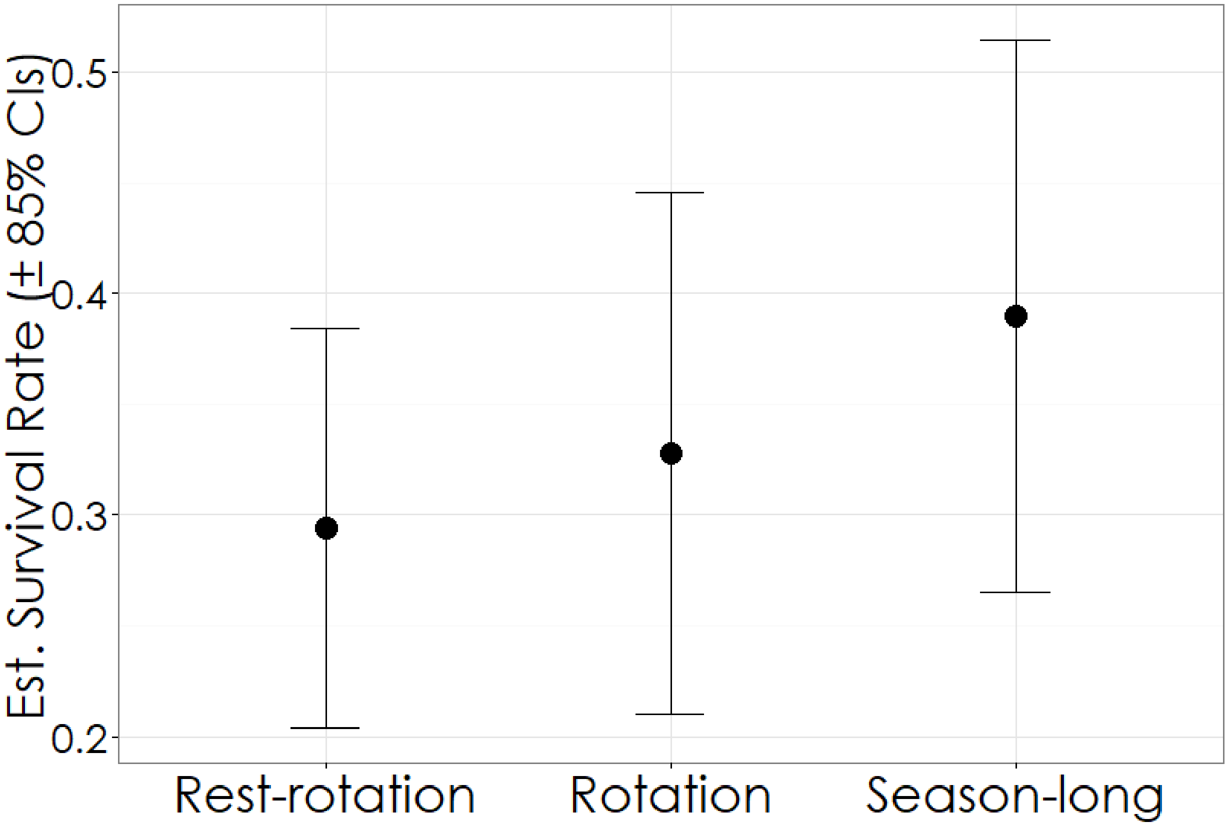

Figure 5. Estimated survival (± 85% confidence intervals) of sharp-tailed grouse nests in each of the three grazing treatments in 2016–17.

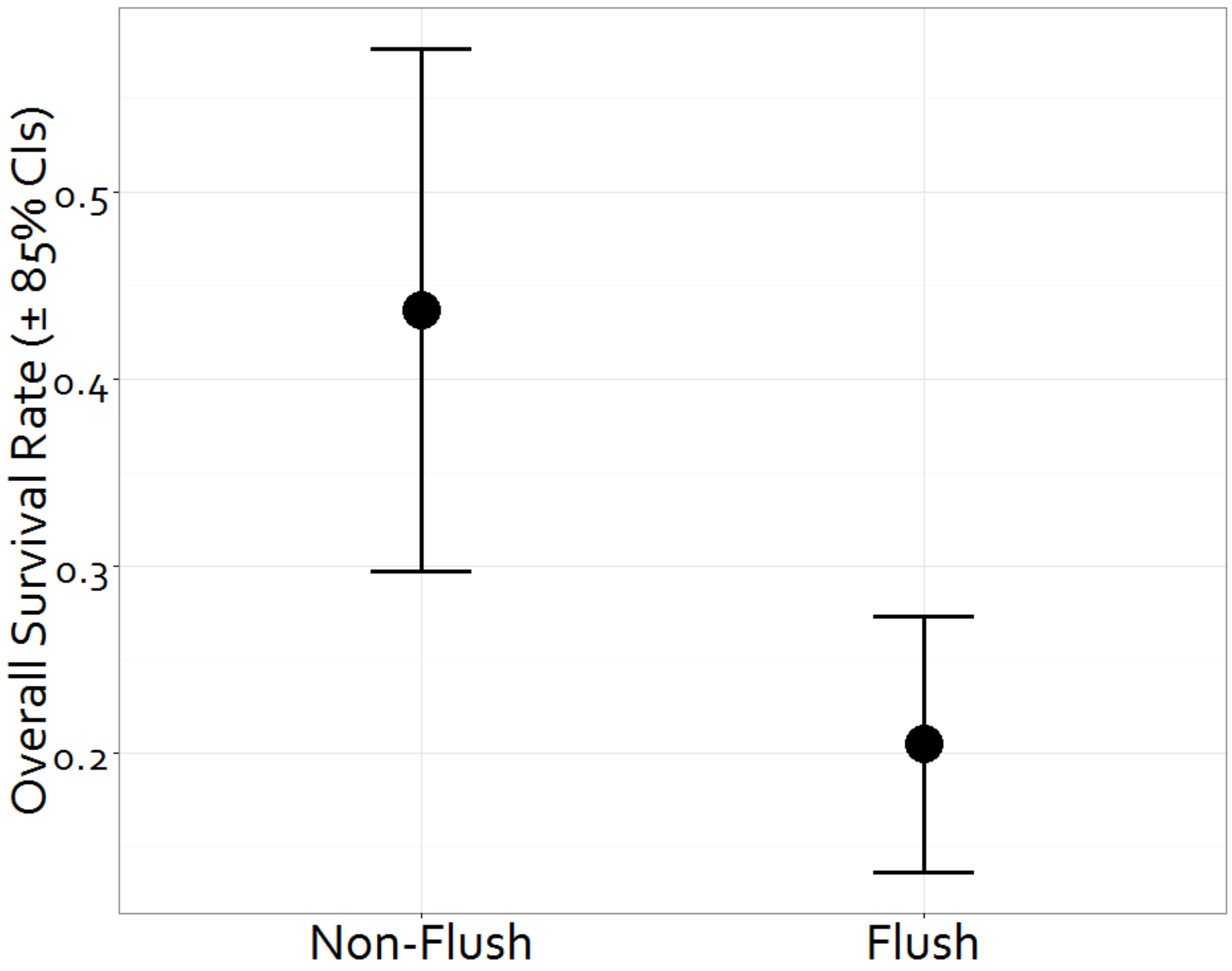

Figure 6. Estimated survival rates (*β* ± 85% confidence intervals) for sharp-tailed grouse nests during the 37-d nesting period for females that were flushed at least once off the nest compared to females that were never flushed off the nest.

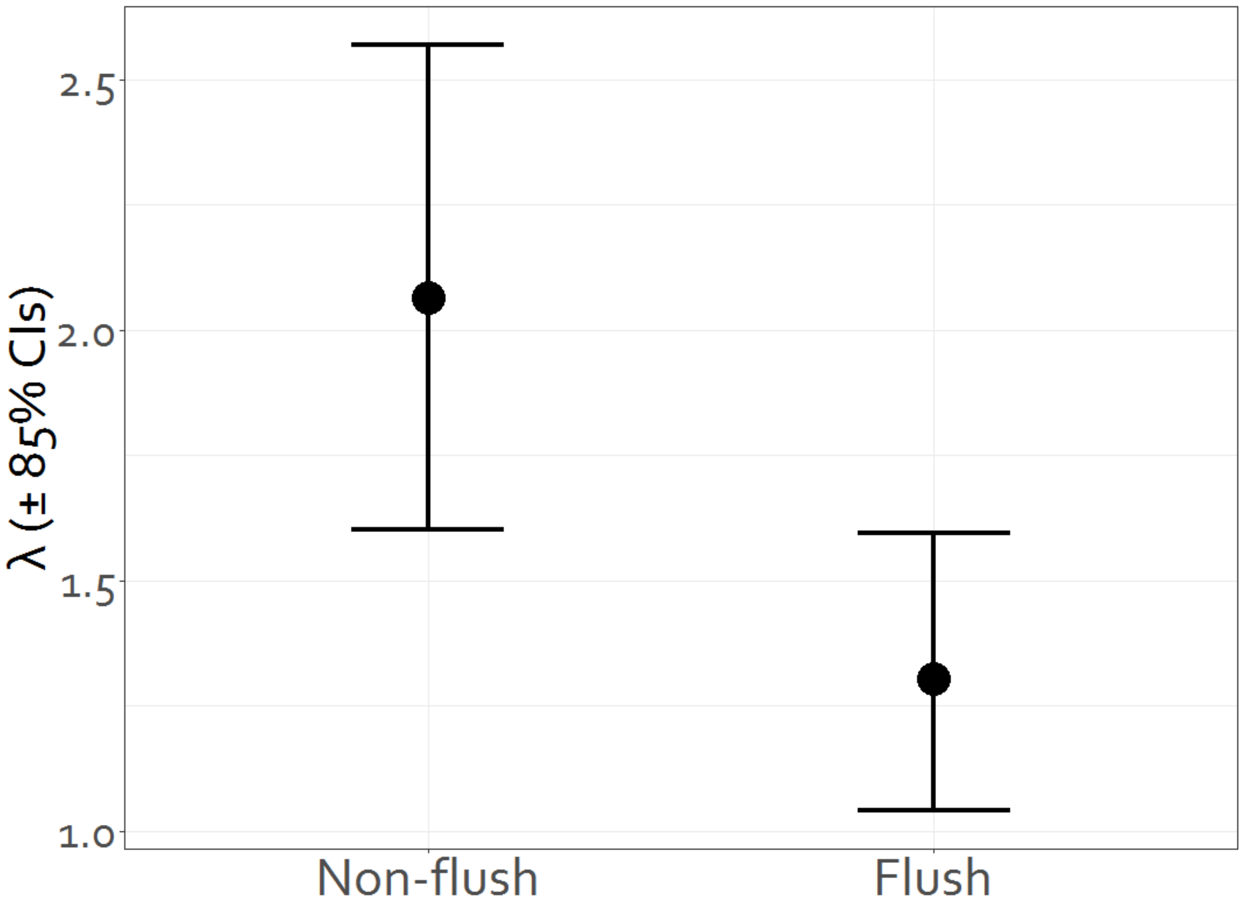

Figure 7. Estimated population growth rates (β ± 85% confidence intervals) for female sharp-tailed grouse that were flushed at least once off the nest compared to females that were never flushed off the nest. Population growth rate was estimated using deterministic matrix models with vital rates estimated for our study population and the only parameter that differed between the two groups was nest survival.

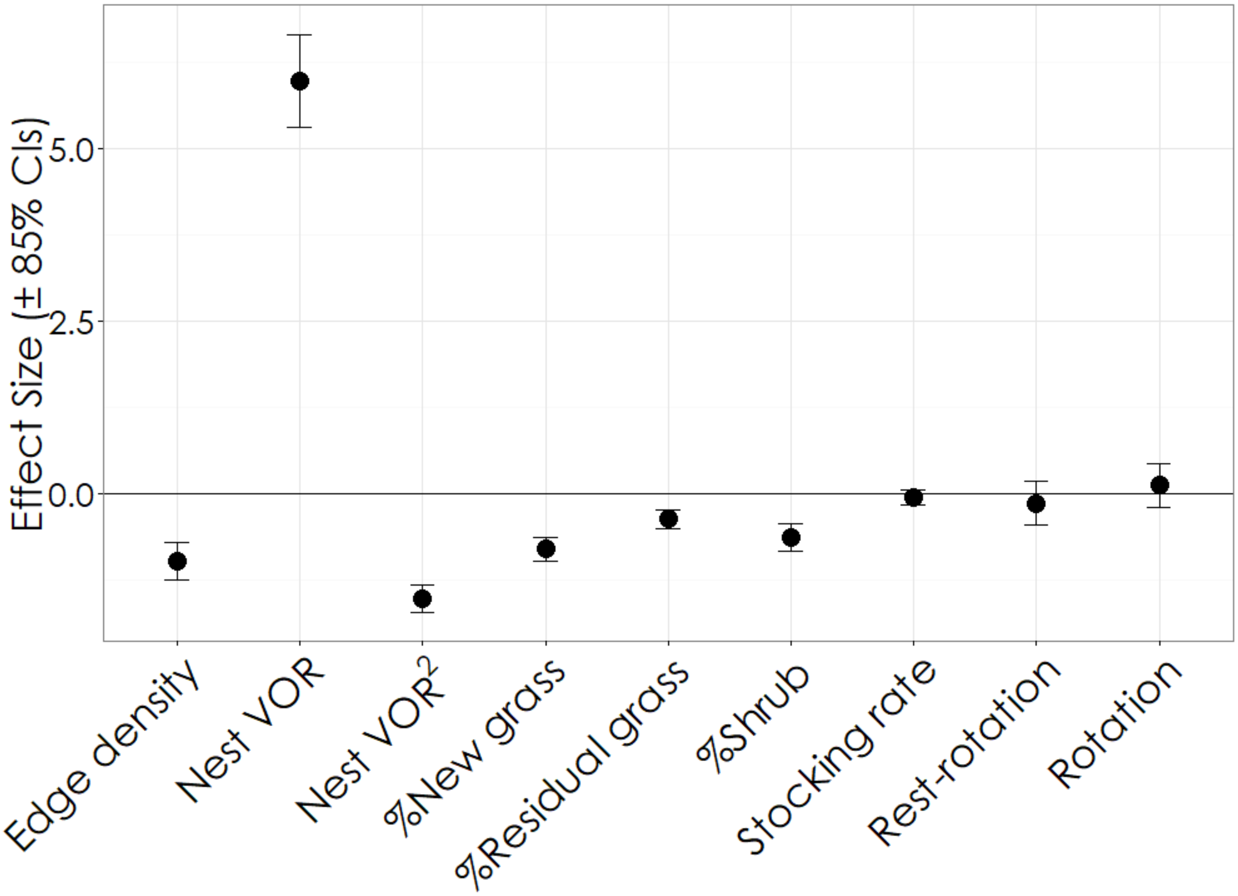

Figure 8. Standardized effect size (*β* ± 85% confidence intervals) for each variable in the analysis examining sharp-tailed grouse nest site selection in 2016–17. Parameters for each grazing system represent effect sizes in relation to the reference category of season-long grazing.

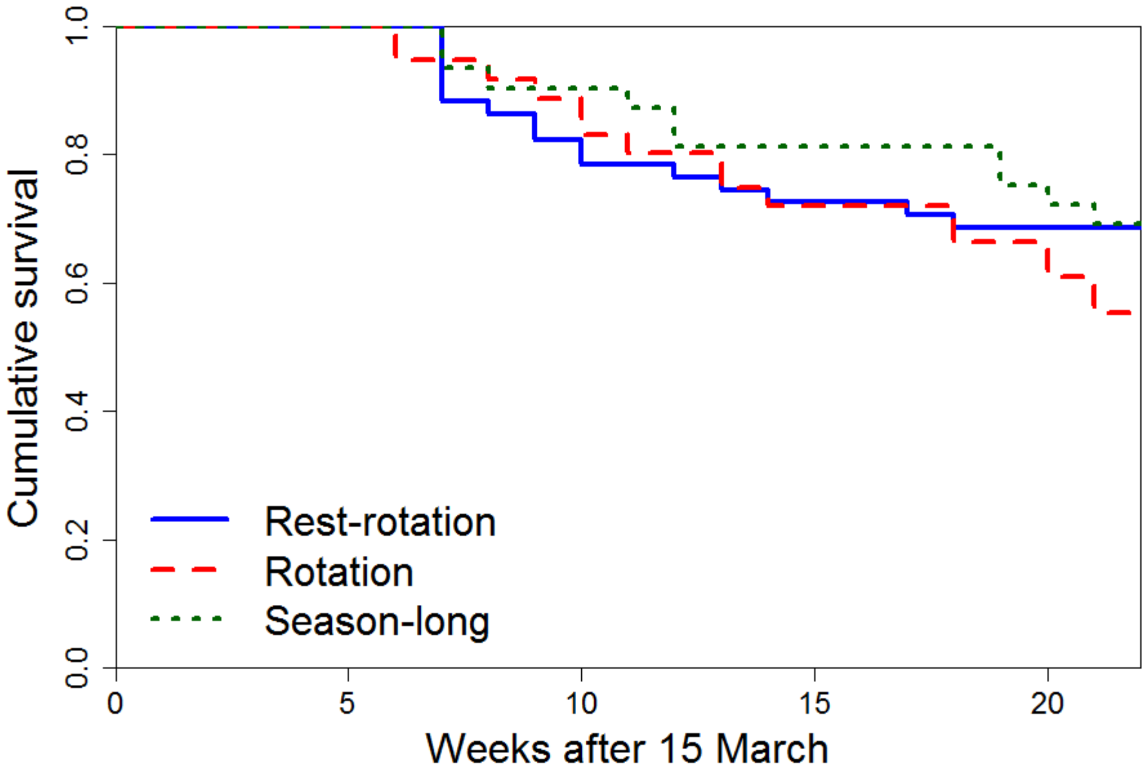

Figure 9. Kaplan-Meier plot of the cumulative survival of radio-marked female sharp-tailed grouse during the breeding season (March – August) in each of the three grazing systems. Females were assigned to the grazing system in which their home range was centered or the system which contained the majority of locations if there were insufficient locations to calculate a home range.

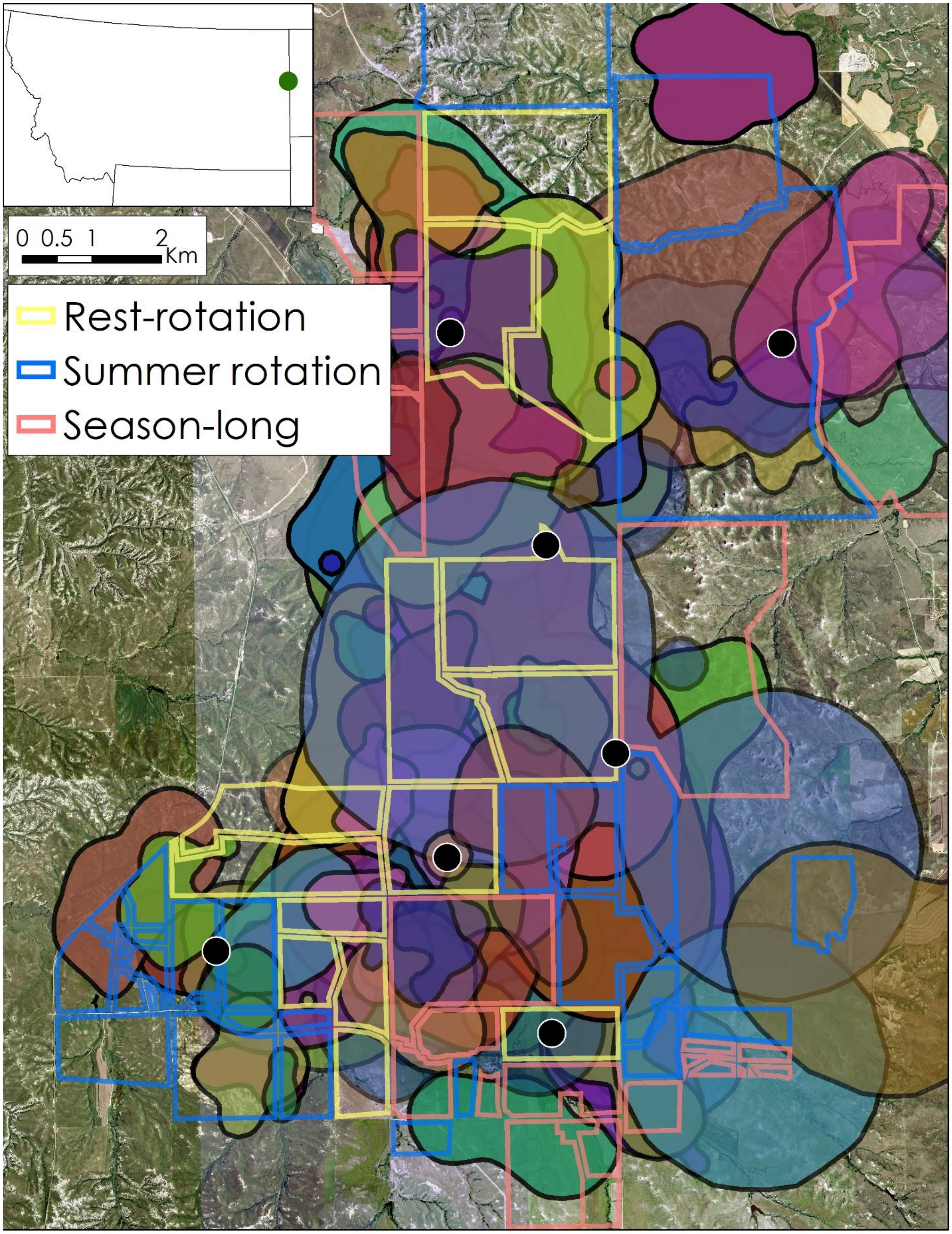

Figure 10. Breeding season home ranges calculated for 53 female sharp-tailed grouse in 2017. Grazing treatments and capture leks (black circles) are shown for reference.

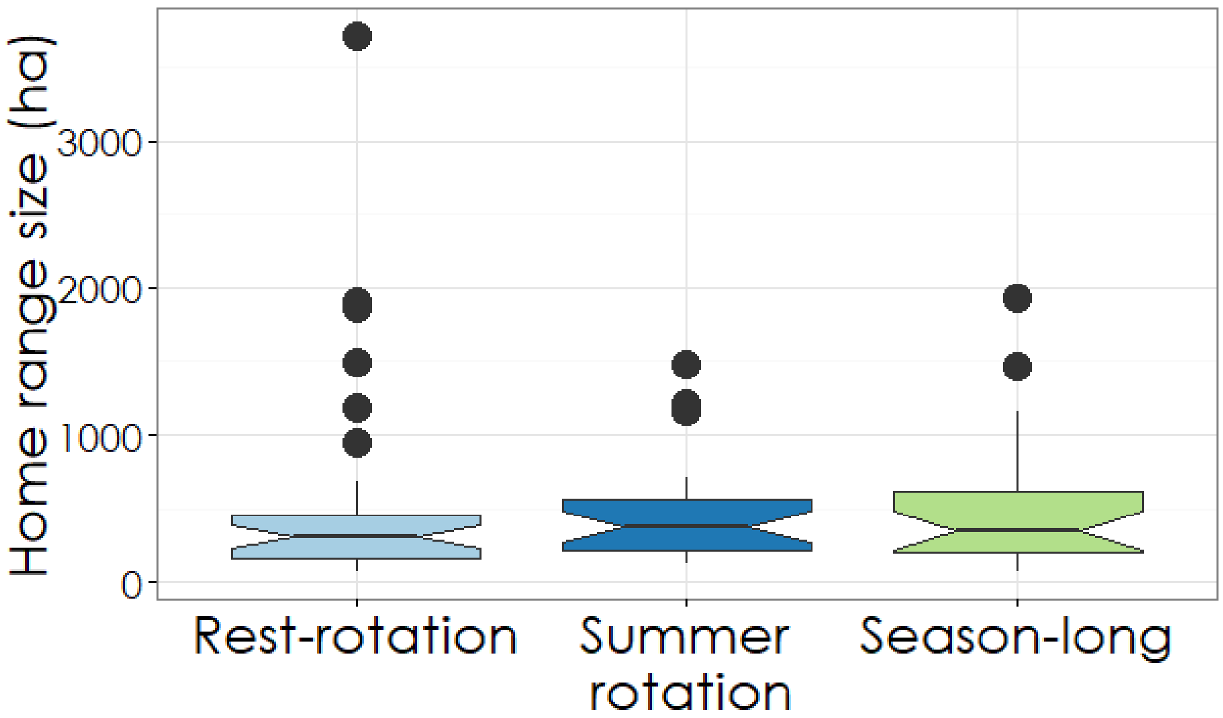

Figure 11. Boxplot of female sharp-tailed grouse breeding season home range size by grazing treatment with points representing outliers. An individual female's home range was assigned to a grazing treatment according to the system at the home range centroid.

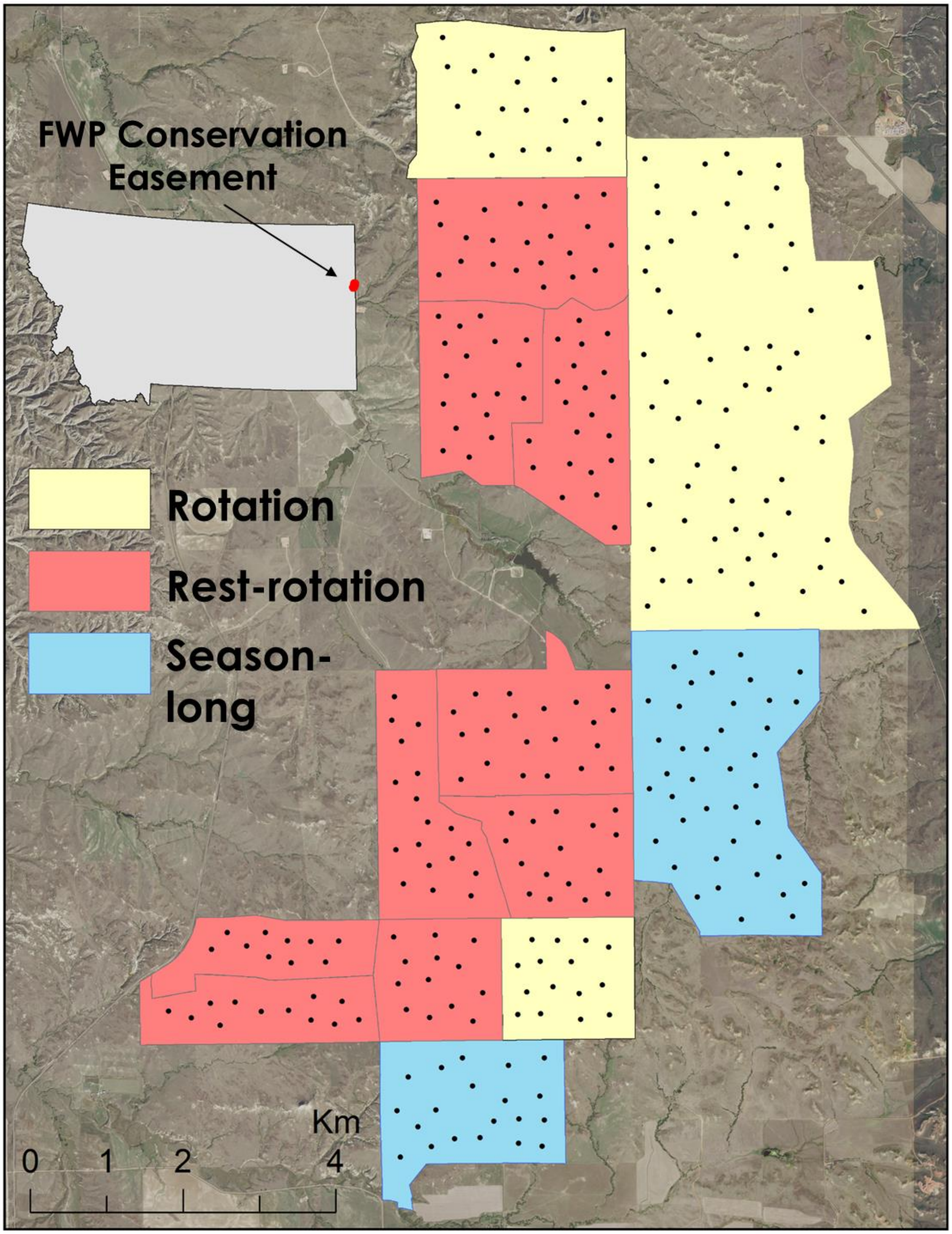

Figure 12. Bird point count survey locations on the Buxbaum conservation easement managed under rest-rotation grazing, and on adjacent private and federal lands managed under season-long and intensive summer-rotational grazing in Richland County, Montana.

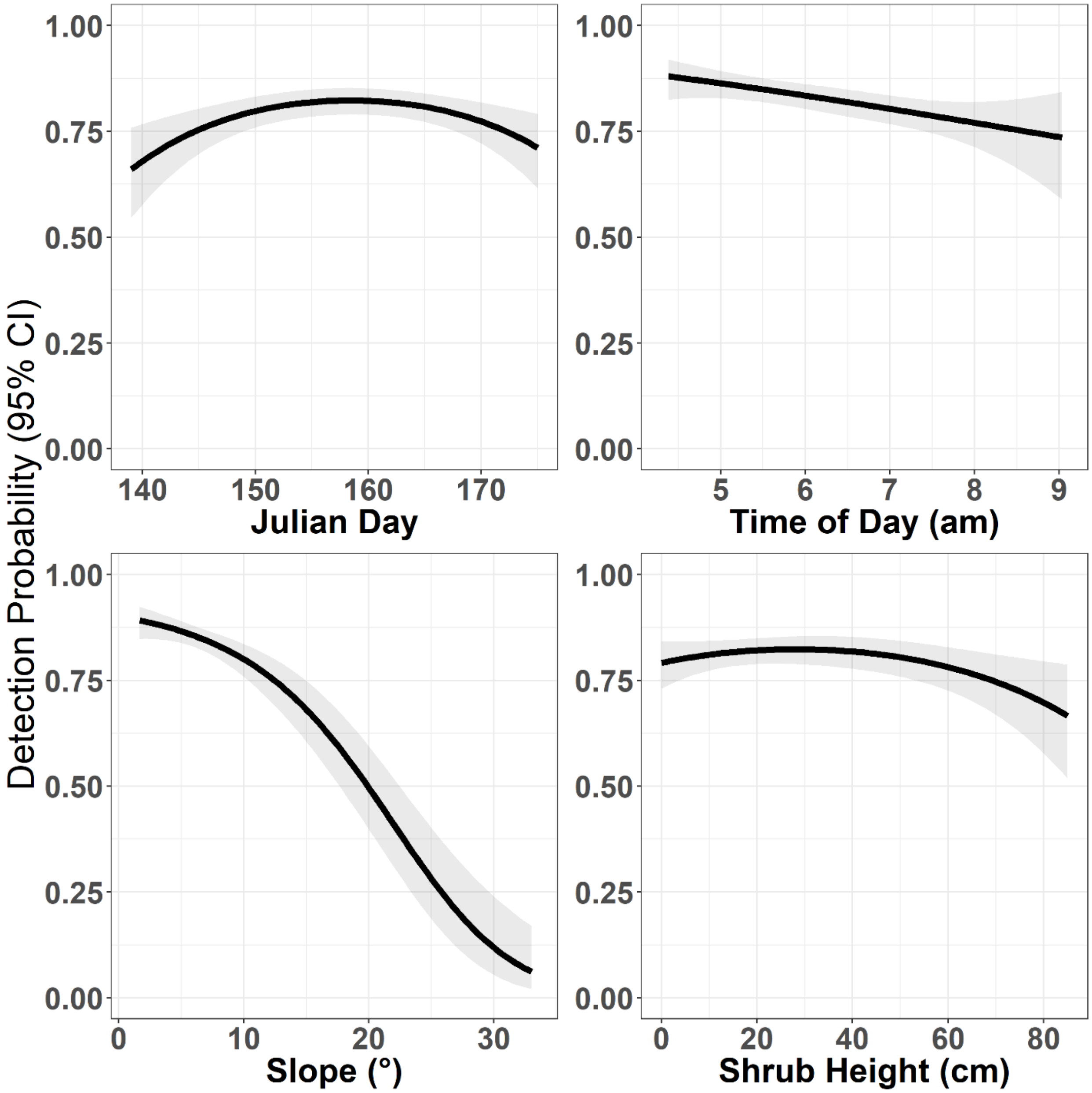

Figure 13. Grasshopper sparrow detection probabilities were most influenced by Julian day, time of day, slope, and shrub height.

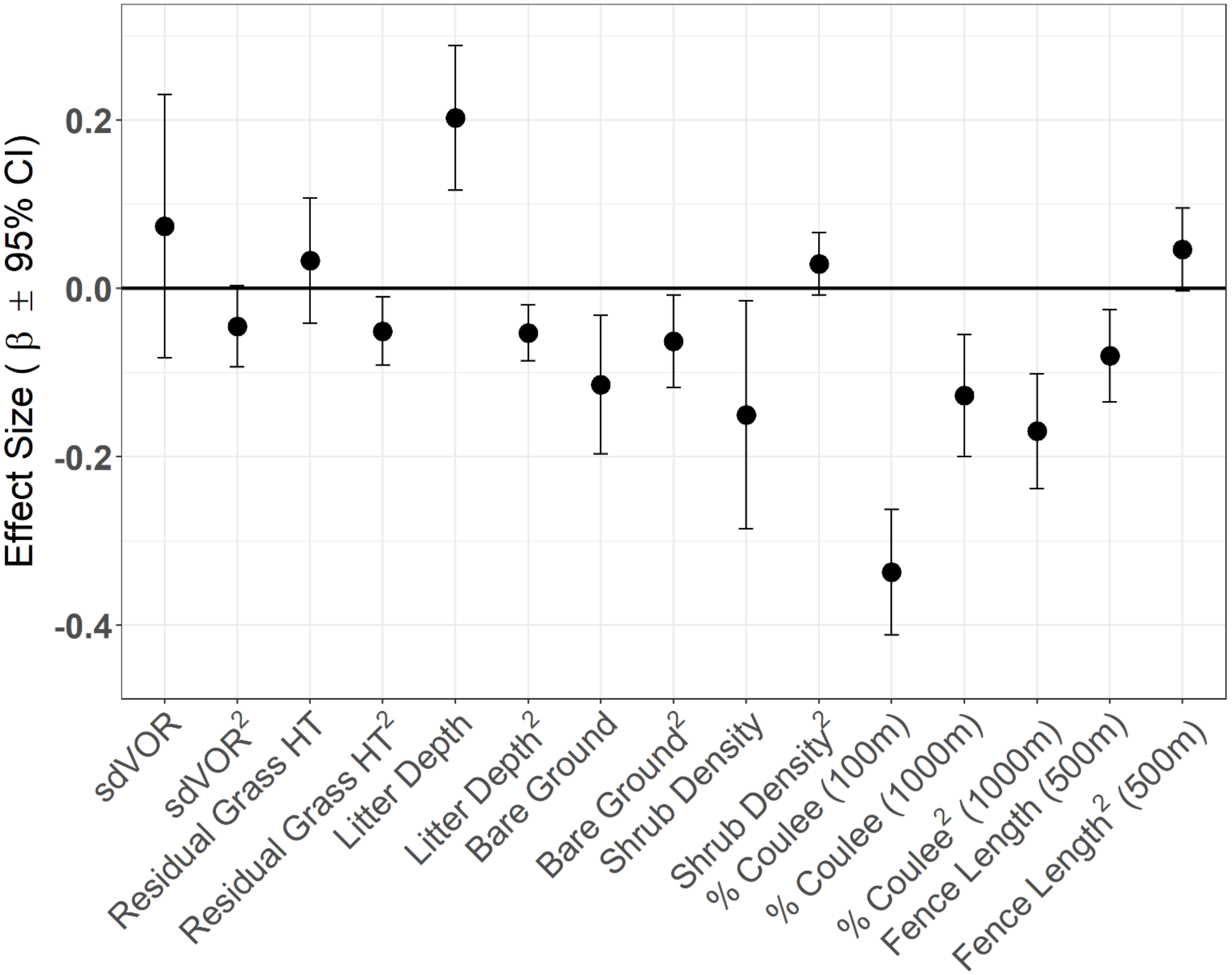

Figure 14. Effect sizes (*β* ± 95% CI) from the top local-scale habitat model and the top landscape-scale habitat model describing grasshopper sparrow abundance. Strong support was shown for those effects which 95% CI do not overlap 0. Effect sizes are comparable, as all covariates are standardized. ^{2} Variable indicates support for a quadratic effect and represents the quadratic component of the variable. sdVOR = standard deviation of visual obstruction reading.

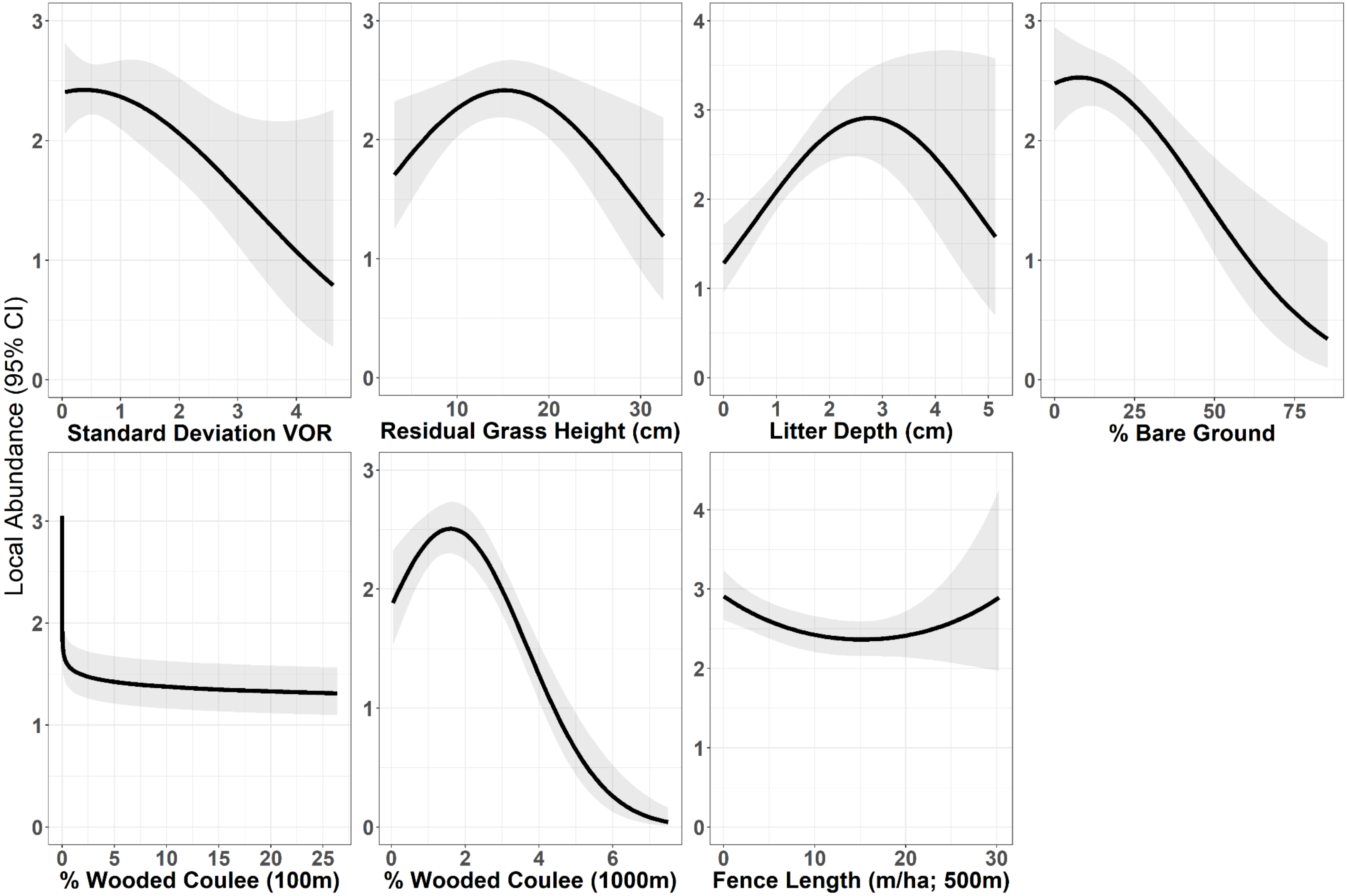

Figure 15. The top habitat model describing local abundance of grasshopper sparrows included strong support for the standard deviation of visual obstruction reading (VOR), residual grass height, litter depth, bare ground coverage, and wooded coulee coverage. The top landscape-scale habitat model describing grasshopper sparrow abundance included support for the proportion wooded coulee within a 1000m radial area from the survey point and the amount of fence within a 500m radial area from the point.

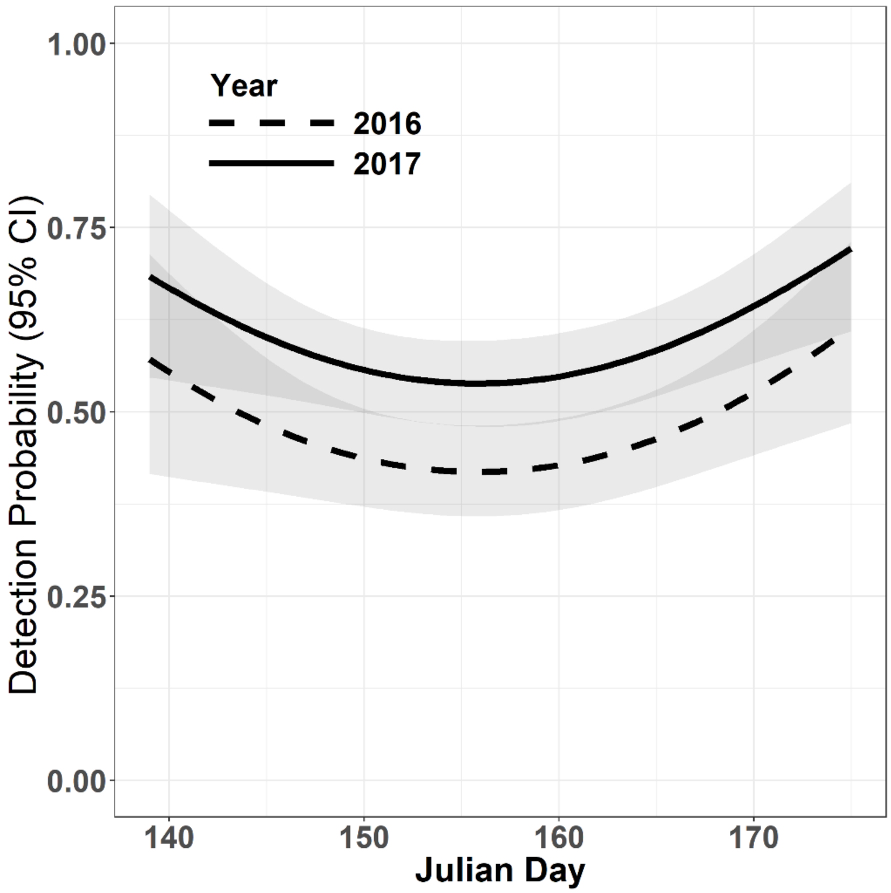

Figure 16. Vesper sparrow detection probabilities were most influenced by year and Julian day.

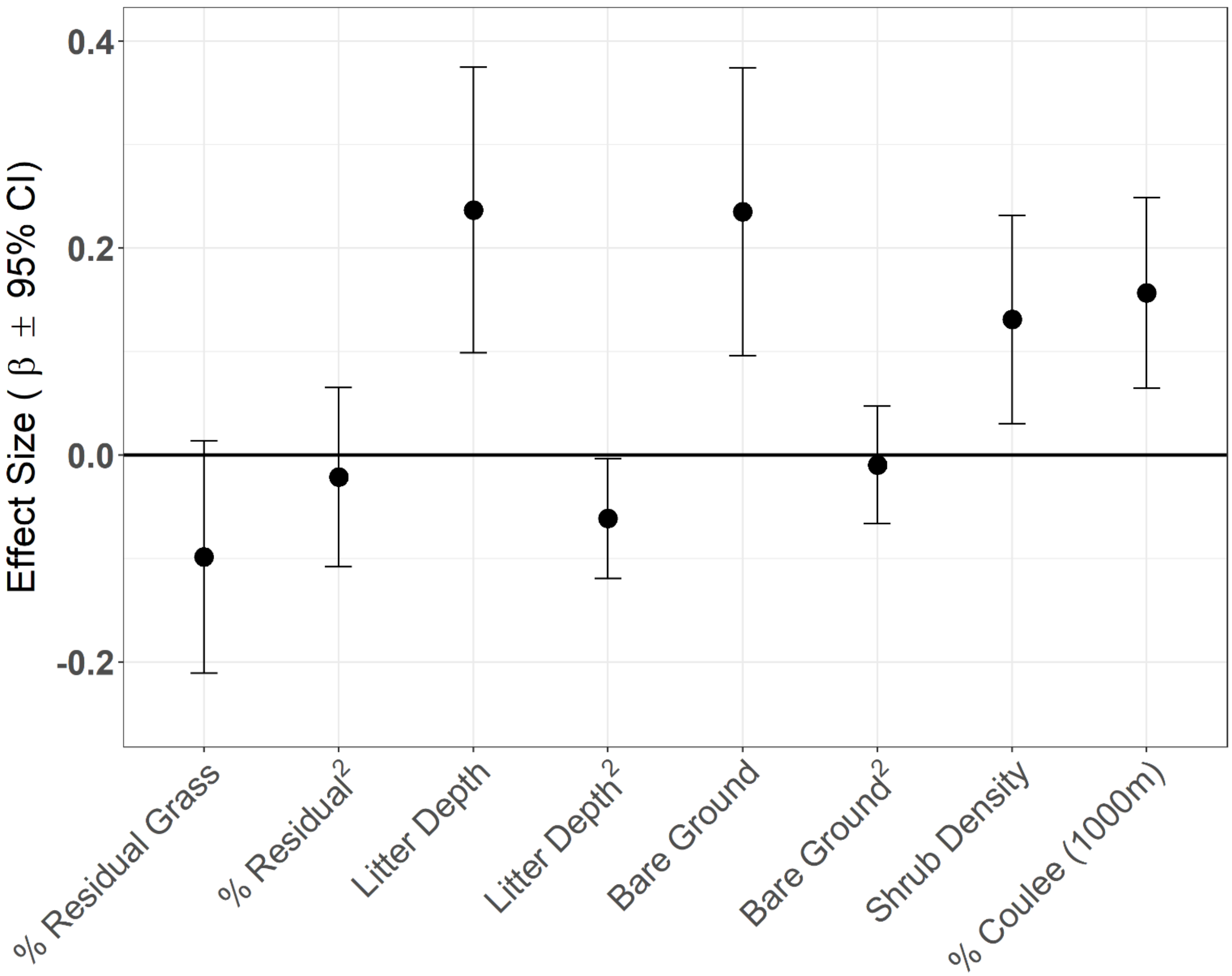

Figure 17. Effect sizes (*β* ± 95% CI) from the top local-scale habitat model and the top landscape-scale habitat model describing vesper sparrow abundance. Strong support was shown for those effects which 95% CI do not overlap 0. Effect sizes are comparable, as all covariates are standardized.

[2] Variable indicates support for a quadratic effect and represents the quadratic component of the variable.

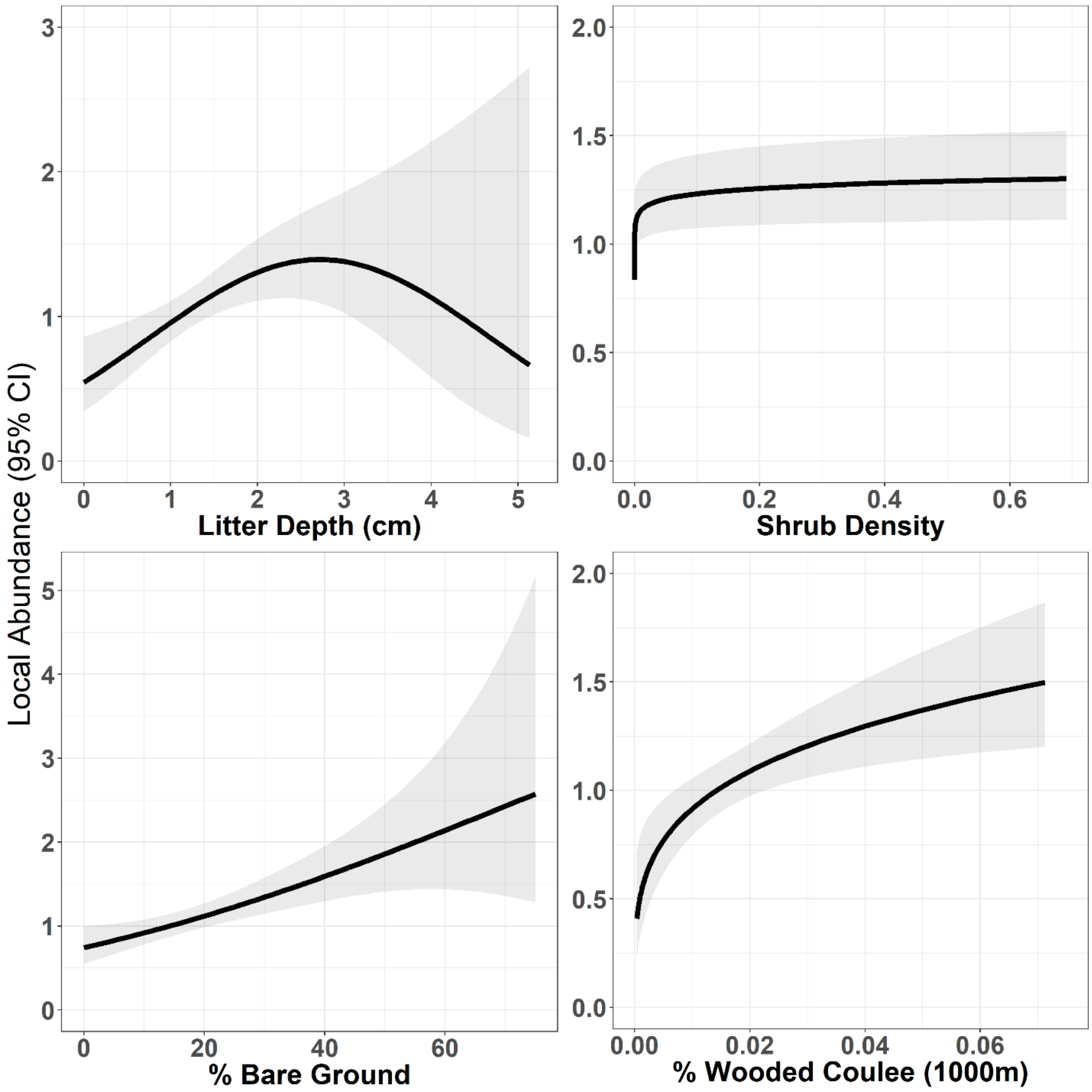

Figure 18. The top habitat model describing local abundance of vesper sparrows included strong support for litter depth, bare ground, shrub density, and wooded coulee coverage. The top landscape-scale habitat model describing vesper sparrow abundance included strong support for the proportion wooded coulee within a 1000m radial area from the survey point.

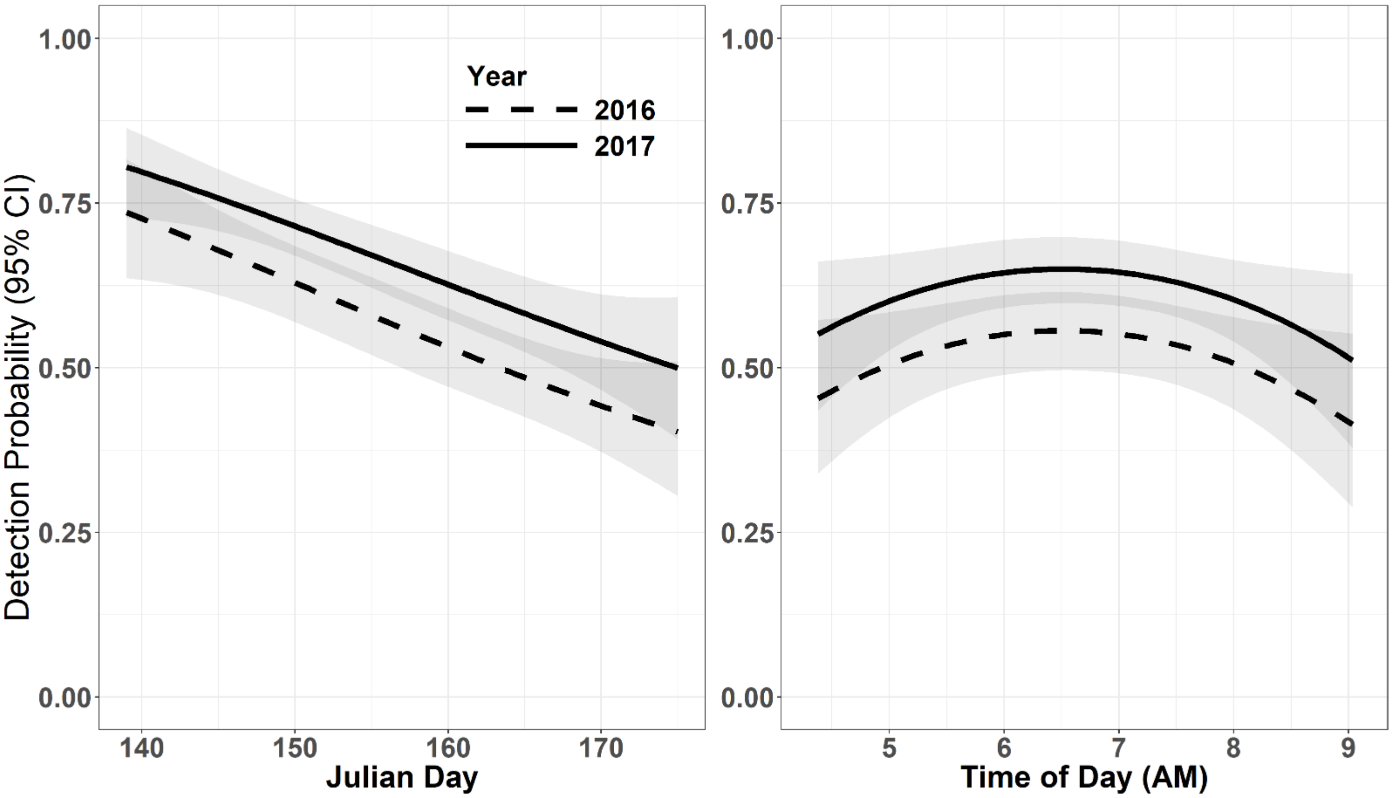

Figure 19. Western meadowlark detection probabilities were most influenced by year, Julian day, and time of day.

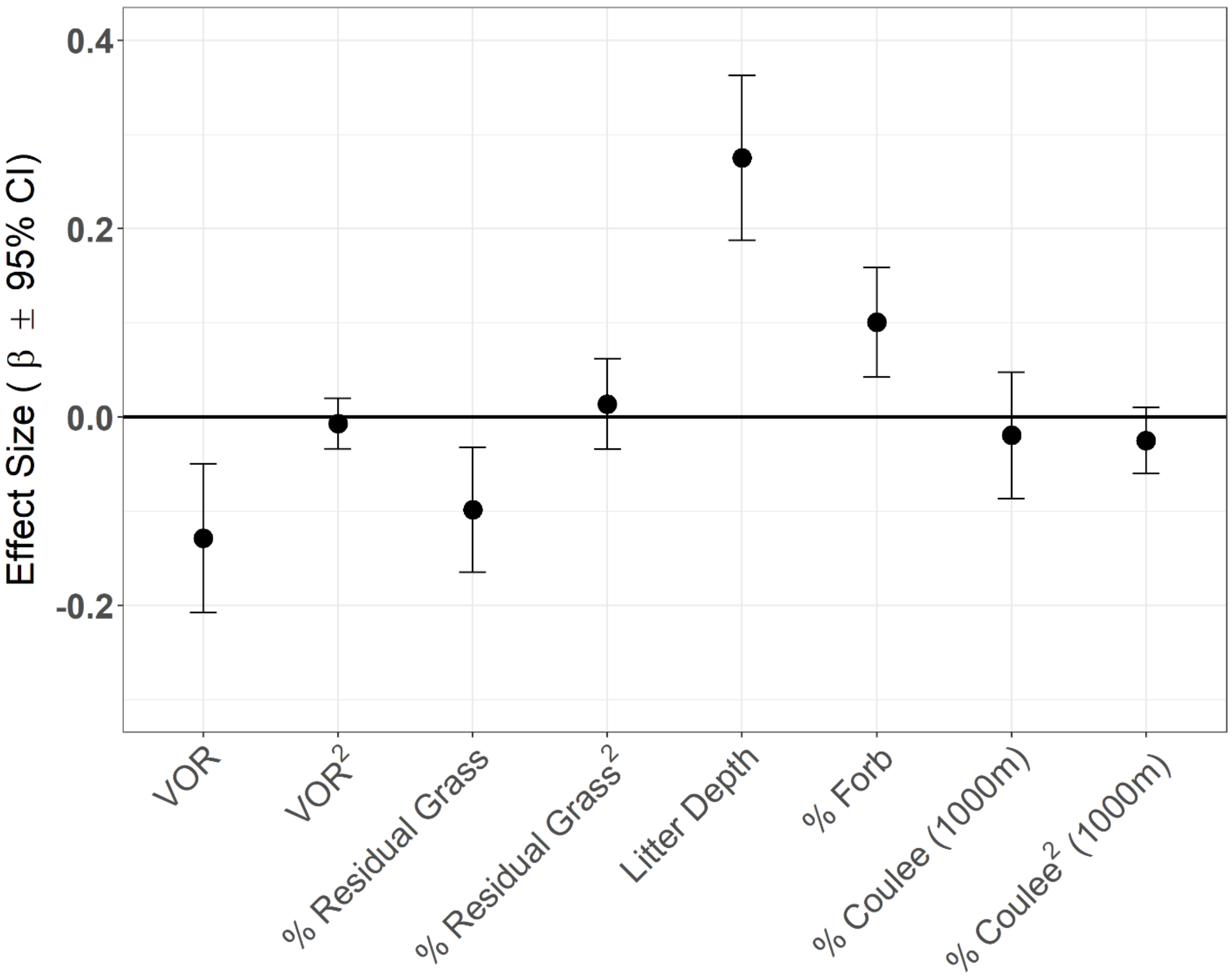

Figure 20. Effect sizes (*β* ± 95% CI) from the top local-scale habitat model and the top landscape-scale habitat model describing western meadowlark abundance. Strong support was shown for those effects which 95% CI do not overlap 0. Effect sizes are comparable, as all covariates are standardized.

^{2} Variable indicates support for a quadratic effect and represents the quadratic component of the variable.

Abbreviations: VOR, Visual Obstruction reading.

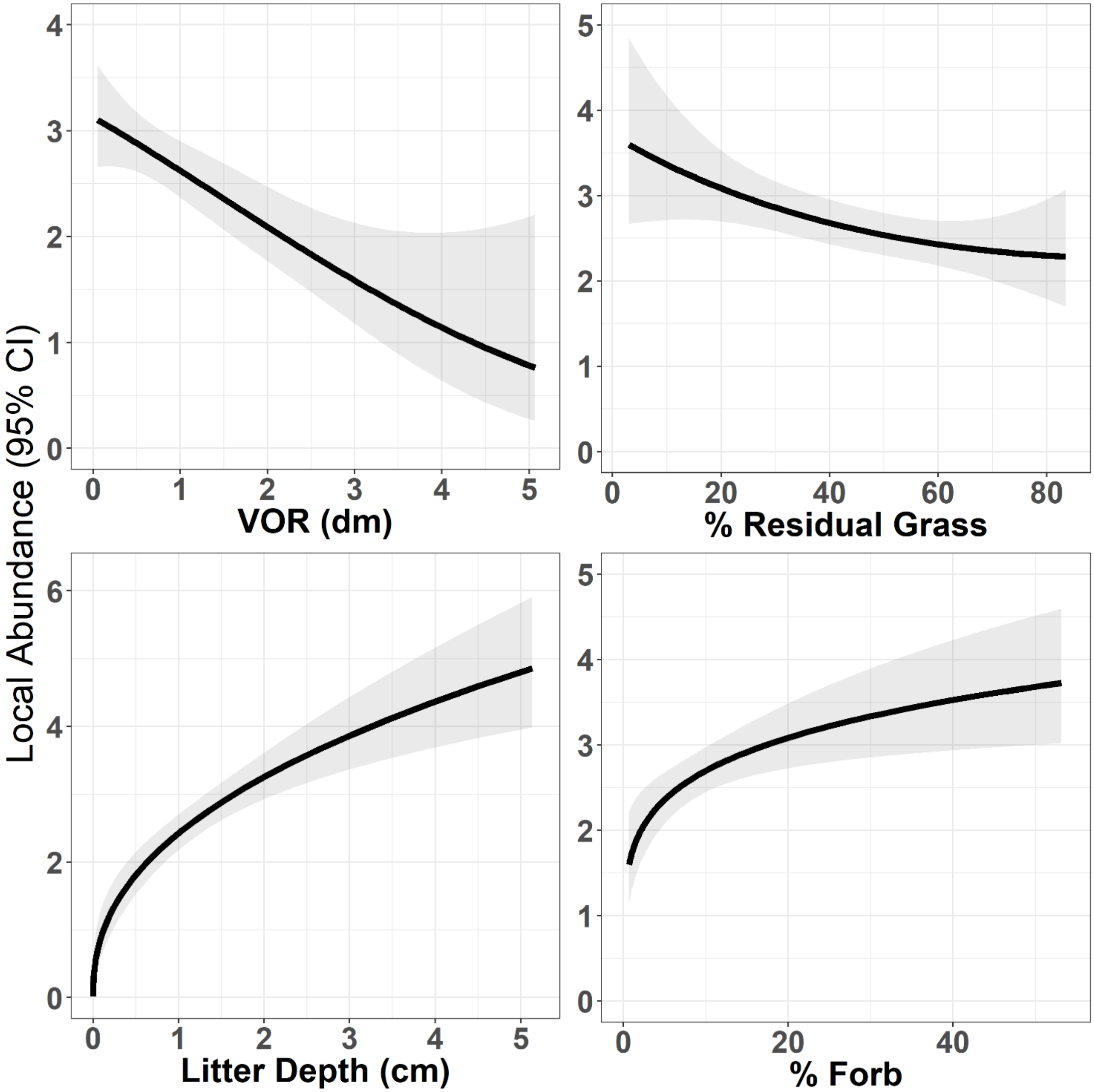

Figure 21. The top habitat model describing local abundance of western meadowlarks included strong support for visual obstruction (VOR), residual grass coverage, litter depth, and forb coverage. The top landscape-scale habitat model describing vesper sparrow abundance had support over the null model and included proportion coulee (1000m scale), but did not show strong support for the effect.

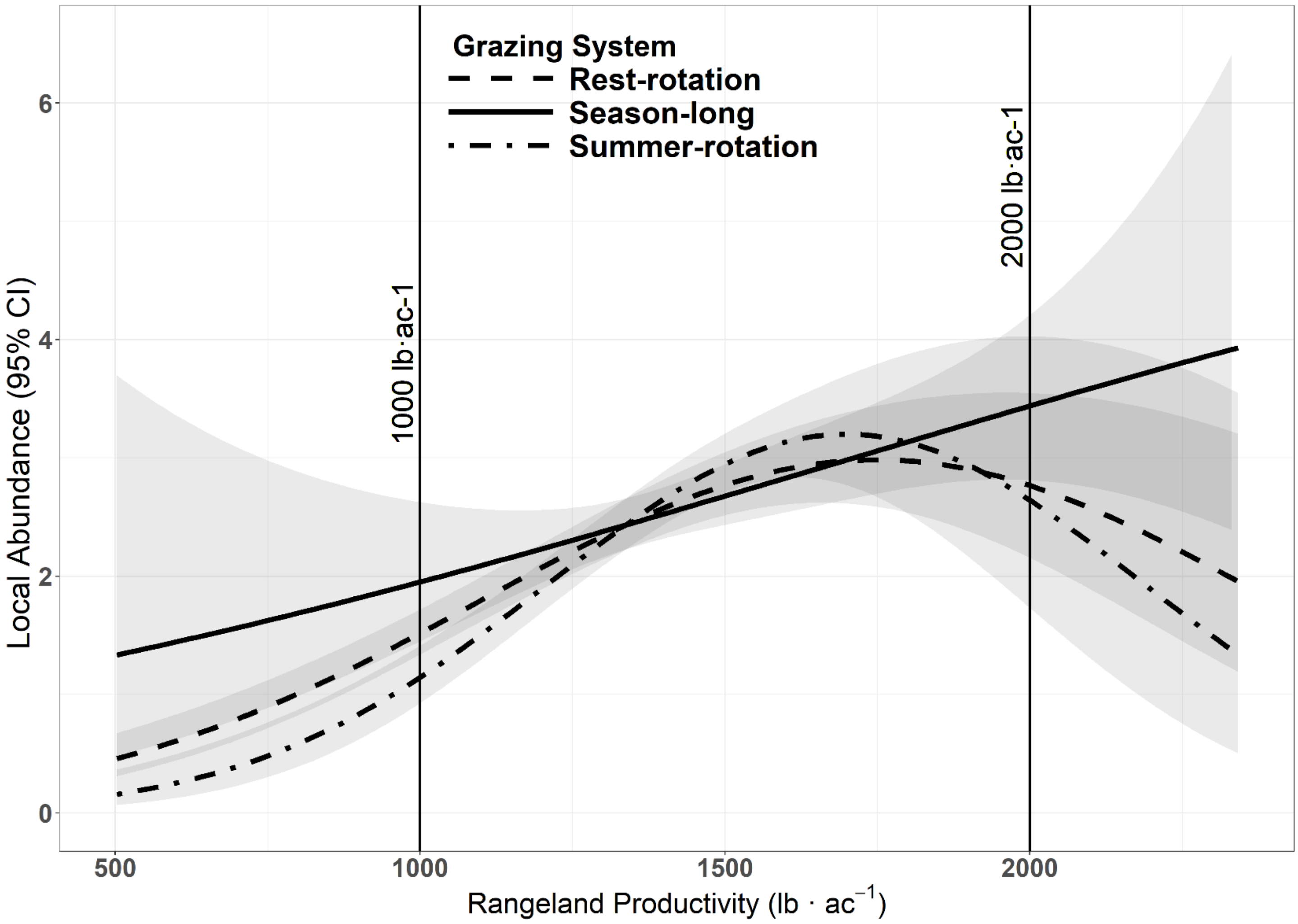

Figure 22. The top management-level model for grasshopper sparrow included an interaction between grazing system and a quadratic effect of rangeland productivity, with the main quadratic effect of stocking rate. Grasshopper sparrow abundance tended to increase with rangeland productivity; abundance was similar across grazing systems when productivity was moderate (1000 – 2000 lb · ac^{-1}), but tended to be highest in pastures managed with a season-long grazing system when rangeland productivity was low (<1000 lb · ac^{-1}) and high (>2000 lb · ac^{-1}).

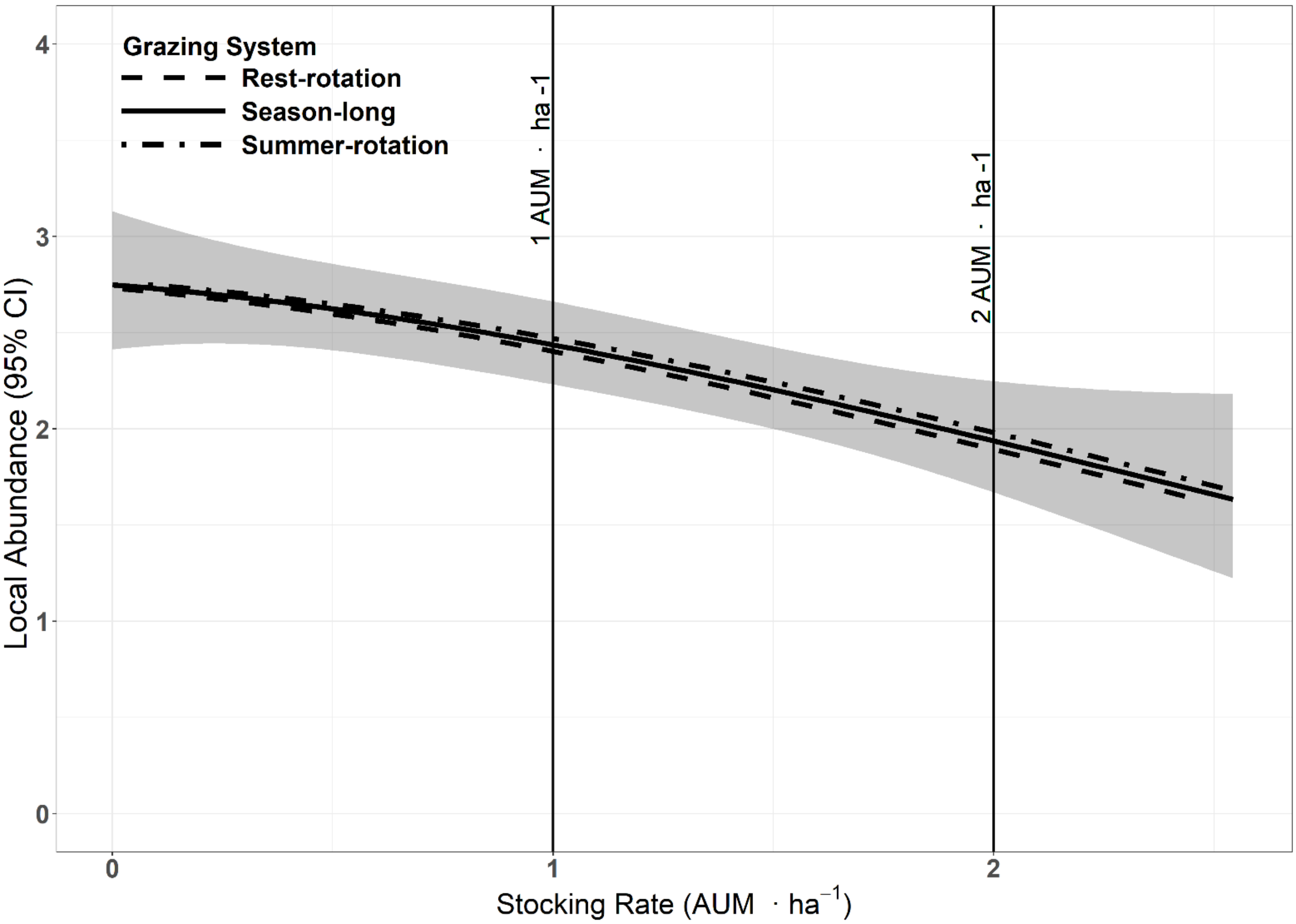

Figure 23. The top management-level model for grasshopper sparrow included an interaction between grazing system and a quadratic effect of rangeland productivity, with the main quadratic effect of stocking rate. Grasshopper sparrow abundance showed a negative response to increasing stocking rate, and the effect was similar among grazing systems.

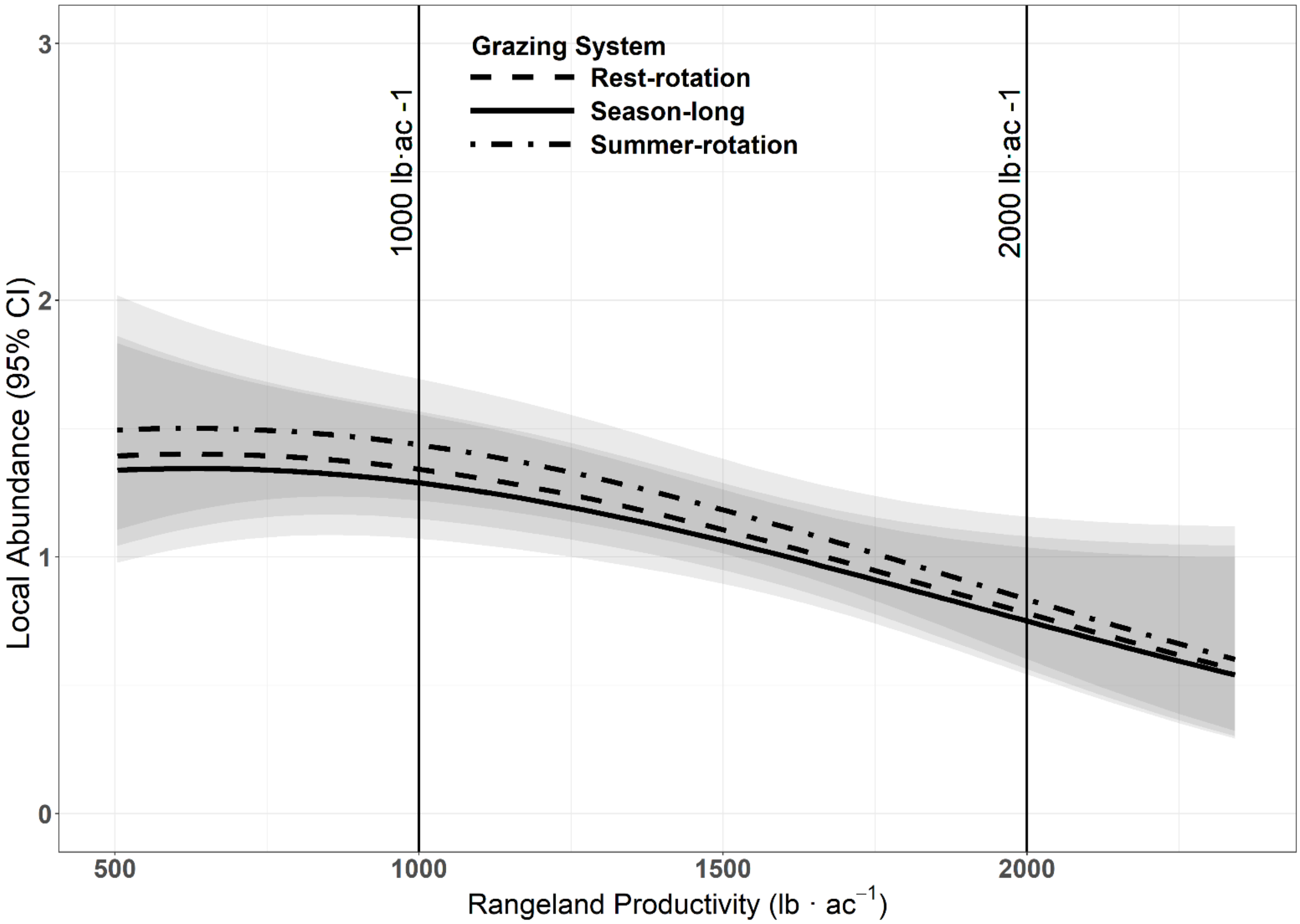

Figure 24. The top management-level model for vesper sparrow included an interaction between grazing system and the quadratic effect of stocking rate, with a main quadratic effect of average rangeland productivity, and the next top model included main effect of grazing system and main quadratic effect of stocking rate (Table 22). Vesper sparrow abundance decreased with increasing average rangeland productivity beyond 1000 lb · ac^{-1}, and the effect was similar among grazing systems.

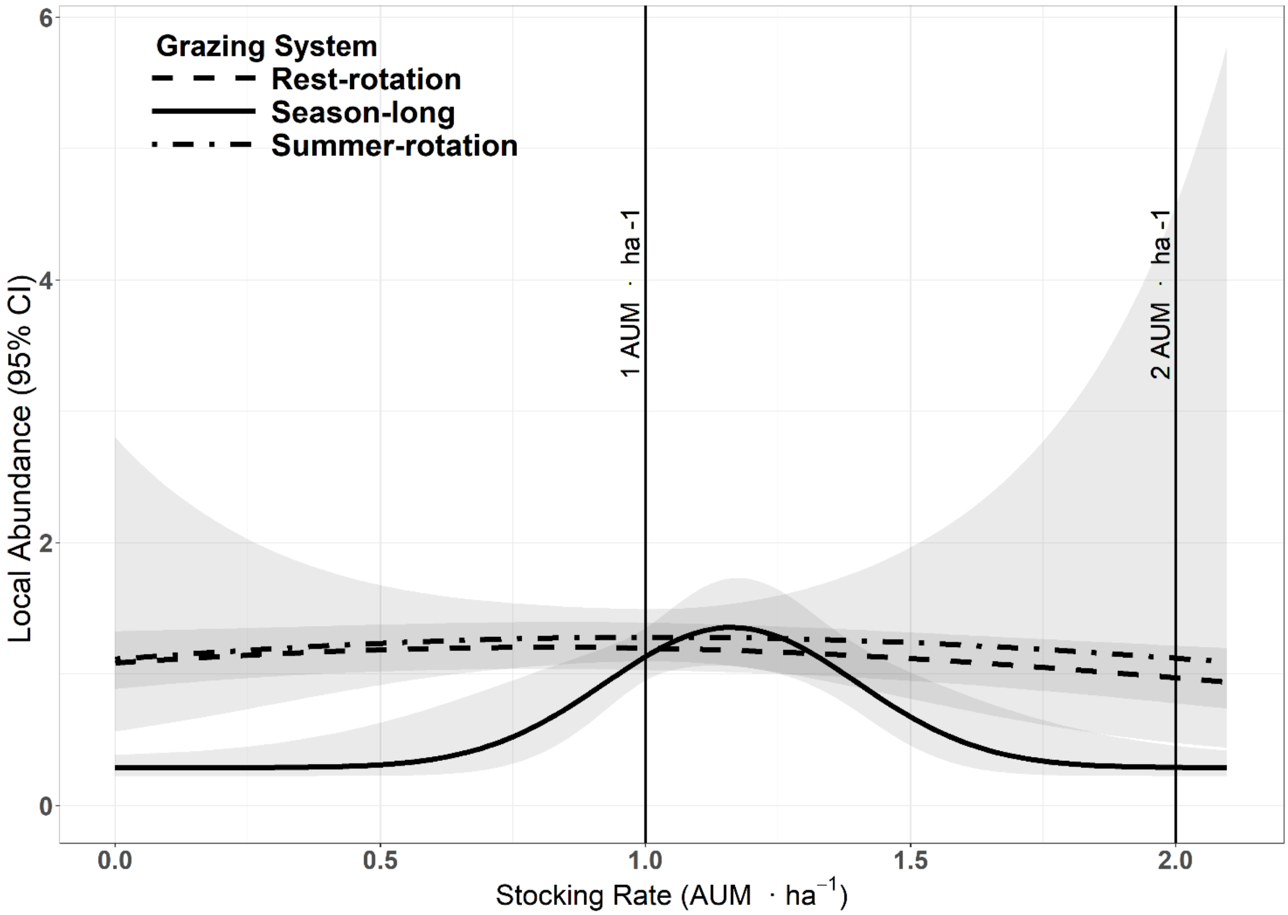

Figure 25. The top management-level model for vesper sparrow included an interaction between grazing system and the quadratic effect of stocking rate, with a main quadratic effect of average rangeland productivity. Vesper sparrow abundance showed a weak quadratic response to stocking rate, increasing with increasing stocking rate up to approximately 1 AUM · ha^{-1}, and decreasing beyond this rate. An interaction between grazing system and stocking rate was supported; however, season-long grazing systems were only stocked at rates between 1 and 2 AUM · ha^{-1}, so the resulting quadratic effect of stocking rate on vesper sparrow abundance within season-long grazing systems is uninformative.

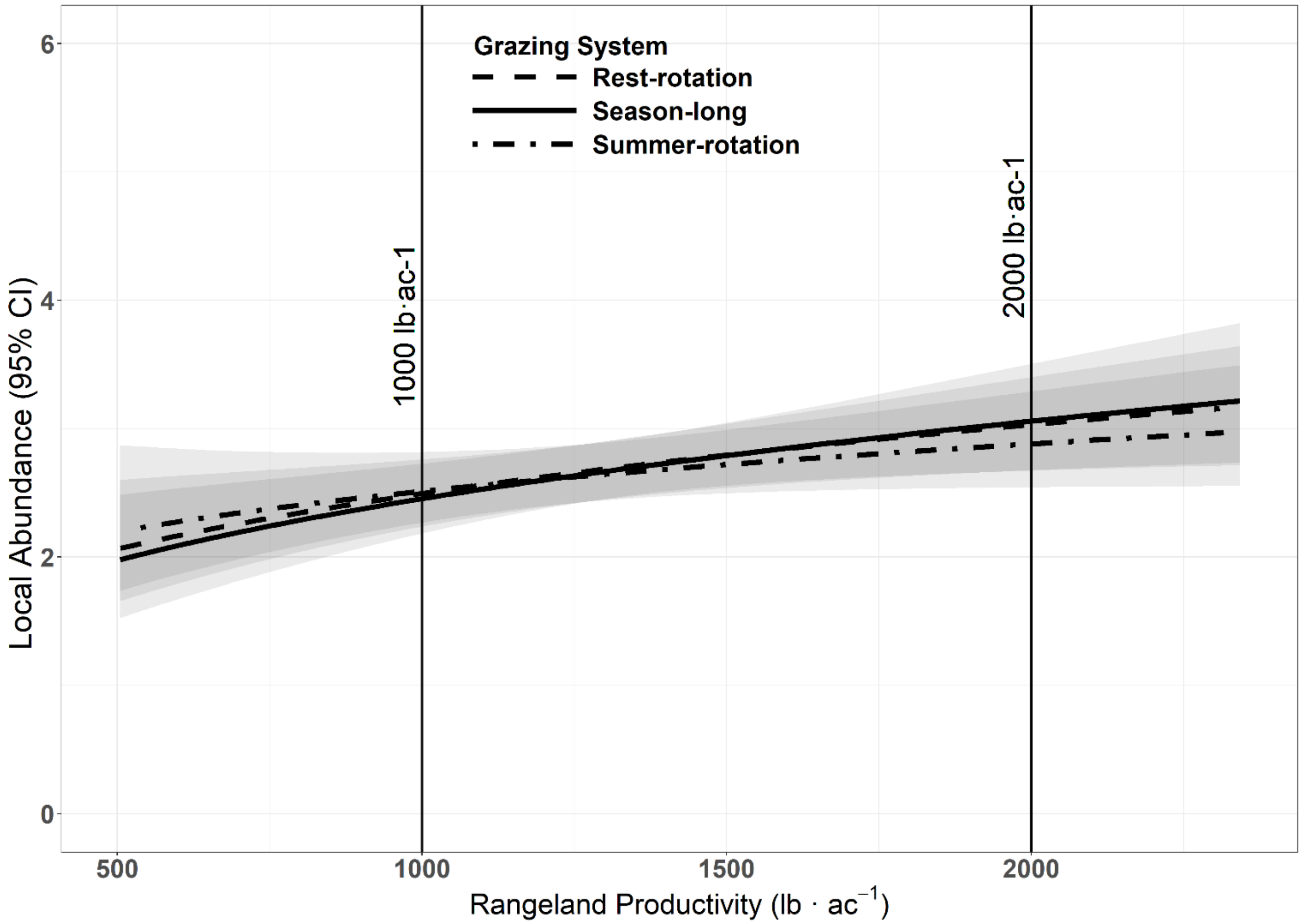

Figure 26. The top management-level model for western meadowlark included main effect of rangeland productivity and main quadratic effect of stocking rate, and the next top model which was also supported with $\Delta AIC_c < 2$ included an interaction between grazing system and rangeland productivity, with the main quadratic effect of stocking rate. Western meadowlark abundance showed a positive pseudo-threshold response to rangeland productivity, where abundance increased with increasing productivity, and the effect was similar among grazing systems.

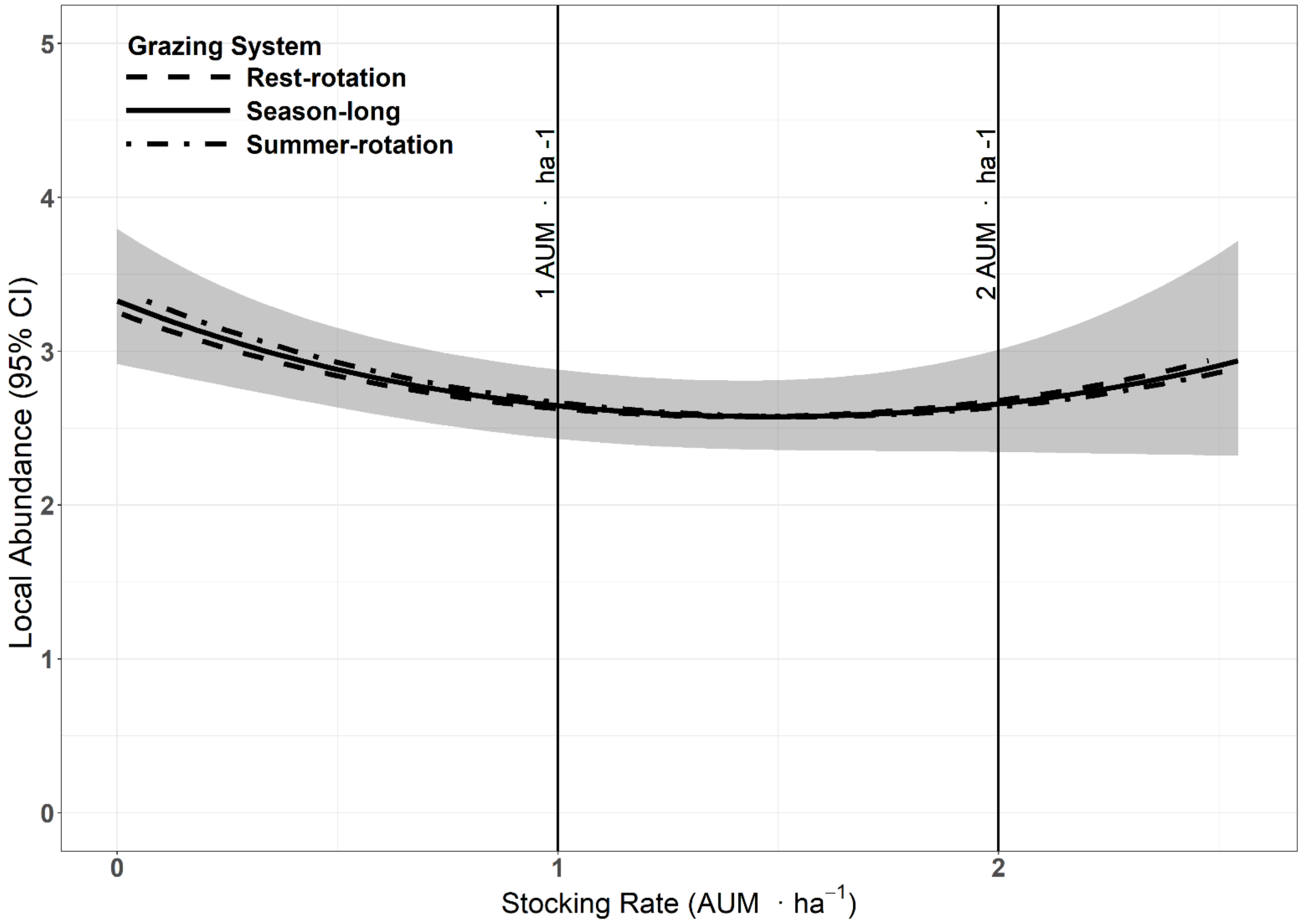

Figure 27. The top management-level model for western meadowlark included main effect of rangeland productivity and main quadratic effect of stocking rate, and the next top model which was also supported with $\Delta AIC_c < 2$ included an interaction between grazing system and rangeland productivity, with the main quadratic effect of stocking rate. Western meadowlark abundance showed a weak quadratic response to stocking rate, where abundance decreased with increasing stocking rates from 0 to 1 AUM · ha^{-1}, with increases in stocking rate beyond 1 AUM · ha^{-1} not strongly affecting western meadowlark abundance.