EFFECTS OF CATTLE GRAZING ON GRASSLAND BIRD ASSEMBLAGES IN SOUTH TEXAS GRASSLANDS

A Thesis

by

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ABSTRACT

Many assemblages of birds have declined globally in the past several decades, with grassland bird species experiencing the most substantial declines. One of the leading hypotheses that explains this decline is the conversion of native grasslands to agriculture and ranching landscapes. Using point count data from 2015 to 2019, my goal was to perform a before-after-control-impact (BACI) assessment to determine the response of local grassland bird assemblages to various grazing regimes in south Texas. Continuous grazing treatments appeared to foster grassland bird richness, though Brown-headed Cowbird, Dickcissel, Grasshopper Sparrow, Northern Bobwhite, and Scissor-tailed Flycatcher densities were not impacted by grazing treatment. Eastern Meadowlark densities were impacted by grazing treatment, with a rotational moderate regime supporting the highest density. Both moderately and highly stocked rotational regimes may be beneficial for Eastern Meadowlarks, though lower cattle densities appeared to be most favorable. I recommend that ranchers in Texas implement rotational grazing systems as opposed to continuous systems in an effort to support Eastern Meadowlark density and foster grassland bird conservation.

DEDICATION

My thesis is dedicated to everyone who has helped me through my academic journey, including my mom and dad who have invested in my education, my husband who has been by my side throughout this process, my grandma who has baked me cookies to fuel my studies, and especially to all of my teachers, professors, and advisors who have inspired and guided me every step of the way.

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Contributors

This work was supervised by a thesis committee consisting of Doctor Michael Morrison [advisor] of the Department of Rangeland, Wildlife and Fisheries Management, Doctor Fred Smeins of the Department of Rangeland Ecology and Management, and Doctor Tyler Campbell of The East Foundation.

The data analyzed in this thesis was provided by The East Foundation. The analyses section was conducted with the guidance of Doctor Brian Pierce of the Texas A&M Natural Resources Institute. All other work conducted for this thesis was completed by the student independently.

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1. INTRODUCTION

Of the many groups of birds that have declined globally in recent decades, grassland birds have declined more than any other avian assemblage in North America (Peterjohn and Sauer 1999, Barlein 2016, Ethier et al. 2017, Rosenberg et al. 2017, Stanton et al. 2018). I defined grassland birds as species that have affinities towards grasslands and grass-shrub vegetation, whether it be year-round or select seasons, and whether it is for nesting, feeding, or some other purpose. There are several existing hypotheses that explain the substantial decline observed in grassland bird assemblages. In the United States, some of the biggest contributing factors are habitat loss and fragmentation, replacement of prairie ecosystems with agricultural landscapes, and deterioration of western U.S. rangelands (Brennan and Kuvlesky 2012).

Native grasslands once covered 1.5 million km^2 of the American continent; tallgrass, mixed-grass and short-grass prairies now cover only 1%, 20%, and 30% of their historical range, respectively (Knopf 1994, Sierra-Corona et al. 2015). Furthermore, the remaining grasslands across the country are now highly fragmented relative to their historic state of continuity (Cassidy and Kleppel 2017). It is widely agreed upon that loss and degradation of grasslands is the most significant source of grassland bird decline in the U.S. and around the world (Ethier et al. 2017, Stanton et al. 2018). However, it is important to note that there is not a single cause responsible for the decline, but rather it is the result of many factors accumulating that inhibit grassland birds (Brennan and Kuvlesky 2012). As native grasslands declined in distribution, so did many grassland bird species that depended upon them for survival (Brennan and Kuvlesky 2012, Ethier et al. 2017, Stanton et al. 2018). Simultaneously, agricultural landscapes increased in distribution and served as surrogates of grasslands for many grassland bird species, being one of the remaining environments for them to live (Cassidy and Kleppel 2017).

Bison (*Bison bison*) were one of the most numerous and influential grazers across the continent, but today occur in small numbers compared to their historic abundance (Kohl et al. 2013). Cattle now represent part of the ecological role that bison once held. In grassland ecosystems, disturbances such as grazing can produce patchiness throughout the vegetation, which creates a more heterogenous environment (Fuhlendorf et al. 2006). Previous studies have suggested that heterogeneity may be the precursor to biodiversity, and in some cases, it should be the very basis for management decisions regarding conservation (Christensen 1997, Ostfeld et al. 1997, Wiens 1997, Fuhlendorf and Engle 2001). Furthermore, researchers have found that systems with higher biodiversity have stronger resilience and increased productivity (Petersen et al. 1998). Sanderson et al. (2007) noted that a more diverse landscape is an unofficial form of an insurance policy. For example, if a drought resulted in the loss of vegetation species important to cattle ranching, having other similar species would help ensure that cattle have other forage compared to an environment with fewer species. Fostering biodiversity in grasslands results in greater ecosystem stability in response to disturbances, increased forage production, and reduced invasion by exotics (Sanderson et al. 2007). With this in mind, managing ranches for a more diverse landscape can be beneficial for both grassland bird assemblages and cattle operations. A more diverse plant community, for example, would provide a wider range of habitats for specific bird species and would allow cattle a greater range of forage.

Jensen (2001) argued that cattle operations and conservation may even go hand-in-hand, specifically when it comes to preserving large areas of land for grassland birds. To successfully design and implement ecologically sustainable livestock management practices, it is crucial to understand relationships between livestock and cohabitant birds (Derner et al. 2009). Though cattle management has the potential to support grassland birds, it can also negatively impact them when grasslands are degraded because birds require certain sizes and structures of grass for successful reproduction (Bleho et al. 2014, Bilotta et al. 2007, Andrade et al. 2015). When managed properly, cattle ranching can result in a diverse landscape mosaic that supports a variety of species (Skinner et al. 1984, Hull 2002, Powell 2006, Stroppel 2009). A non-profit agricultural research organization in south Texas called the East Foundation has implemented a project to investigate the effects of grazing regime on local grassland bird assemblages in an effort to guide future management and conservation practices.

The San Antonio Viejo (hereafter SAV) ranch was located in south Texas, part of Jim Hogg and Starr counties, and is owned by the East Foundation (Figure 1). They manage this working laboratory as an agricultural research organization that fostered science, education, and outreach. To improve knowledge of wildlife and vegetation response to grazing, the East Foundation created a large-scale, long-term grazing demonstration project called the Coloraditas Grazing Research and Demonstration Area (henceforth CGRDA). The goal of the demonstration area was to test regional stocking rate paradigms of high (1 Animal Unit [AU]/14 ha) and moderate (1 AU/20 ha) within common grazing systems: continuous and rotational grazing (Figure 2). Under rotational systems, ranchers rotate cattle herds between pastures to let vegetation recover from grazing, where continuous systems allow cattle to continuously graze pastures. I defined grazing regime as the combination of stocking rate and grazing system. The purpose of the demonstration was to monitor the long-term effects of these grazing systems and stocking rates on local wildlife and vegetation, including grassland bird species. Commencing in 2014, the demonstration aimed to mimic real-world scenarios by holding constant yearlong stocking rates within the CGRDA while allowing for realistic flexibility in management decisions such as time of rotation, workings, and providing supplemental feed.

Brennan and Kuvlesky (2012) argued that successful strategies to stabilize and increase grassland bird assemblages must include agricultural, prairie, rangeland, and forested areas; my study contributed by focusing on rangeland research and conservation. Impact assessments are beneficial research tools (Offer et al. 2003, Terlizzi et al. 2005) utilized in rangeland research (Read and Anderson 2000, Larson et al. 2016) that allow scientists to factor out preexisting differences between treatments as only changes from baseline conditions are of interest (Stanley and Knopf 2002). Simply put, impact assessments separate the signal from the noise (Morrison et al. 2008). I used this structured process to determine whether magnitude of change differed over time between treatments (Stanley and Knopf 2002) to analyze grazing impact on local grassland bird assemblages. I used framework of a before-after-control-impact (henceforth BACI) assessment (Stewart-Oaten et al. 1986) with an added component of spatial replication to make it a "beyond BACI" assessment (Underwood 1992).

2. GOALS AND OBJECTIVES

My research goal was to assess the impact of various grazing regimes on local grassland bird assemblages. To do this, I utilized a beyond before-after-control-impact (BACI) assessment. In regard to grassland bird richness and density, I tested the null and alternate hypotheses that:

- H_0: There is no statistically significant difference between any 2 grazing treatments for any species and thus, grazing regime does not impact local grassland bird assemblages
- H_a: There is a statistically significant difference between at least 1 combination of grazing treatments for at least 1 species and thus, grazing regime does impact local grassland bird assemblages

Individual objectives were to 1) Use available data to estimate grassland bird richness for representative assemblage, 2) Generate models to estimate density for select representative grassland bird species, 3) Identify how grazing regime affected grassland bird assemblages, 4) Consider if any statistically significant outcomes reflected potential biological significance, and 5) Use my results to inform management of grassland birds.

3. METHODS

Study Area

My study site was the SAV ranch, a property of the East Foundation, in Jim Hogg and Starr counties, Texas, USA (lat: 26.956671, long: − 98.835408; Figure 1). My project boundary encompassed the entire ranch, approximately 604 km^2 (60,496 ha). The CGRDA pastures represented the treatment sites (7,502 ha; 75.02 km^2) and the remainder of the ranch represented the reference sites (52,994 ha; 529.94 km^2). Approximately half of the ranch was in the Coastal Sand Plain level 4 ecoregion (Figure 3), which contained grasslands intermixed with vegetation such as *Opuntia engelmannii var lindhiemerii* (Texas prickly pear), *Prosopis glandulosa* (honey mesquite), *Celtis ehrenbergiana* (granjeno), *Schizachyrium scoparium var. littorale* (seacoast bluestem), and sand dunes. The other half of the ranch was in the Texas-Tamaulipan Thornscrub level 4 ecoregion (Figure 3), which included shrub species such as *Prosopis glandulosa* (honey mesquite), *Acacia rigidula* (blackbrush), *Croton capitatus* (woolly croton), *Monarda fruticulosa* (spotted beebalm), *Leucophyllum frutescens* (cenizo), and *Acacia berlandieri* (guajillo) (Diamond and Fulbright 1990, Fulbright et al. 1990, McLendon et al. 2013 unpublished report).

Elevation ranged from 52 m on the eastern edge to 64 m on the western edge of the ranch. In this region, the mean annual temperature was 22˚C (with annual fluctuations between 7˚C and 36˚C) and the regional average annual precipitation was 57 cm (National Oceanic and Atmospheric Administration [NOAA] 2016, Davis et al. 2019, Figure 4). A 3-year drought in the area lasting from 2011 to 2013 preceded this study. The mean monthly Palmers Drought Severity

Index did not rise above a mild drought for any month, and severe to extreme drought persisted in 28 of the 36 months from 2011 to 2013 (Palmer Drought Severity Index 2018).

Trained technicians collected data used in this study for an underlying monitoring project to determine sample strategy for small mammals, birds, and herpetofauna (Baumgardt et al. 2019a, Baumgardt et al. 2019b). Prior to the initiation of the monitoring project in 2014, the CGRDA lacked the current cross fencing infrastructure and was grazed continuously at a stocking rate of 1 AU/ 12 ha. In March 2014, the CGRDA was deferred from grazing for approximately 1.75 years to allow the range to recover from drought and intense grazing. Data from 2015 represents the pre-treatment (hereafter “before”) phase of the impact assessment. The East Foundation introduced 435 first year, same-aged, bred Santa Gertrudis cross heifers onto the CGRDA from 3 to 16 December 2015 to initiate the treatment phase (hereafter “during”). Ranchers bred cows each spring with weaning and palpation occurring each fall to keep treatments stocked consistently. The foreman decided to rotate the herds in the rotational treatments based on his visual assessment of forage standing crop and cattle body condition.

Due to a prolonged period of drought in 2017 (Figure 4), cattle were removed from the landscape from May 2018 to the present in an effort to limit economic loss on cattle and to allow the native vegetation to grow without an added stressor. The technicians maintained the sampling protocol described below throughout this time, which allowed the team to collect data when the cattle were removed from the landscape, representing the post-treatment (hereafter “after”) phase of the impact assessment. Though 2018 data represented the after phase, 2019 data also was included in the analysis to see how trends continued after the first year of post-treatment data and to potentially see any lag effects such as recovery from a potential impact (Graham et al. 2007, Drever et al. 2008).

Study Design

My study design was based on a BACI assessment (Grzybowski 1982, Stewart-Oaten et al. 1986) with the addition of spatial replication through additional treatment sites (Underwood 1992) to make it a "beyond BACI" assessment. Though point count data were collected in 2014, this initial year was intended to be a trial run for the research crew to determine the most efficient methods for future sampling. In an effort to reduce introduced bias from protocol changes, I omitted data from 2014. Thus, the following sections also exclude this initial trial run year to better convey the design and protocol relevant to my thesis. My data consisted of collections characterized at 3 time points: pre-treatment data (2015; before grazing), treatment data (2016 to 2017; during grazing), and post-treatment data (2018 to 2019; after grazing). The elements for my project were individual grassland birds that made up the point count data. Based on my overall goal of assessing grassland bird assemblages in Texas, I utilized the sample population from the SAV ranch to extrapolate potential impacts of varied grazing regimes to the greater area. Due to the characteristics of the SAV ranch, I considered this an appropriate representation of the general landscape and vegetation type of the greater area (Gould et al. 1960), thus making it an appropriate reference site for the proposed target population extrapolation.

The treatment sites were composed of transects within the CGRDA pastures (76.89 km^2; 7,689 ha) on the north end of the ranch. These sites consisted of 4 constantly held treatments applied to 10 sub pastures designated as observational units. The western 4 pastures were under continuous and the eastern 6 under rotational grazing systems. Half of the pastures under each grazing treatment contained a moderate stocking rate of 1 Animal Unit (AU)/20 ha and the other half were held under a high stocking rate of 1 AU/14 ha; the East Foundation categorized these stocking rates as moderate and high according to uncited regionally paradigmatic definitions. Thus, within the continuous grazing system pastures (*n*=4), there were 2 (19.25 km^2; 1925 ha) held under a high and 2 (14.0 km^2; 1400 ha) under a moderate stocking rate. Similarly, the rotational treatments contained 3 pastures (21.13 km^2; 2113 ha) held under a high stocking rate and 3 other pastures (20.64 km^2; 2064 ha) held under a moderate stocking rate (Figure 5). Throughout the experiment from 2015 to 2019, there were 10 treatment transects (1 per treatment pasture).

Reference sites were composed of transects located within the remainder of the SAV ranch that contained pastures under varying grazing regimes. Ranchers managed reference sites under typical regional ranching protocol using a flexible range of stocking rates and a combination of continuous and rotational grazing systems throughout sampling. I selected these specific reference sites to create a baseline for comparison that represented management outside the treatment pastures (i.e. they represented change that occurred outside the controlled treatment areas). Specific grazing regimes and workings of pastures within the reference sites were irrelevant for this impact assessment; I simply used the reference area containing typical ranching practices as a baseline comparison to test which constantly held grazing regime in the treatment area benefited grassland birds the most. Due to the underlying project across the ranch, some factors were adjusted annually based on available personnel, time, funding, and requirements of the ongoing monitoring. The number of reference transects fluctuated between 5 and 20 from 2015 to 2019 with 15 transects as the median (Table 2).

Representative Assemblage

Some species have lower detection probabilities than others, which can result in technicians missing them during the survey process, if not surveyed for long enough periods (Nichols 1992). Based on the fact that use of raw species counts as a measure of species richness and density is often biased low (Calder 1990, Boulinier et al. 1998, Diefenbach et al. 2003), I strategically selected a range of species as a representative assemblage to more accurately assess grassland bird changes. Similar to how scientists select certain lab species to model the human system (Gelfand 2002), I used data from selected species under the assumption that they may provide information to a greater assemblage. Shine and Bonnet (2002) described model organisms as taxa selected based on some characteristic to answer a question; I used data from 6 species based on the diversity that they collectively represented to help answer if grazing regime impacted grassland bird assemblages.

Of the 92 breeding grassland bird species detected on the ranch from 2015 to 2019, I strategically chose 6 that represented an array of avian characteristics to maximize possibility of finding an impact on any certain group of birds (Figure 6; Table 1). To represent resident species year-round on the ranch, I selected the Eastern Meadowlark (*Sturnella magna*) (overview of species: Lockwood and Freeman 2014). I chose the Scissor-tailed Flycatcher (*Tyrannus forficatus*) to represent species that are only on the ranch for their breeding season (overview of species: Nolte and Fulbright 1996). To represent species that were seen frequently on the ranch, I selected the Dickcissel (*Spiza americana*) (overview of species: Dechant et al. 2002a), and to represent those rarely observed I selected the Grasshopper Sparrow (*Ammodramus savannarum*) (overview of species: Dechant et al. 2002b). I chose the Brown-headed Cowbird (*Molothrus ater*) to represent the unique group of grassland birds that are nest parasites (overview of species:

Shaffer et al. 2003), and I chose the Northern Bobwhite (*Colinus virginianus*) to represent grassland gamebird species (overview of species: Hernández and Guthery 2012; sympatric study: Bruno 2018). I chose these species to represent a variety of diverse grassland bird groups in an effort to detect an impact of grazing regime on any group of grassland birds. For example, it is possible that the resident Eastern Meadowlark behaves similarly to other residents and therefore, I could apply inferences to other species.

Data Collection

From 2015 to the present, a trained field crew has been collecting data for a monitoring project using point transect distance sampling (Hutto et al. 1986, Buckland et al. 1993). The goal was to develop a robust method for monitoring bird assemblages to provide management recommendations to landowners and biologists (Baumgardt et al. 2019a). The research crew randomly established center points of transects, each of which located at a minimum of 400 m from the edge of individual pastures or ranch boundaries to restrict observations to the specific areas under survey (Figure 5). After oversampling in 2014, Baumgardt et al. (2019a) determined that 4 visits were sufficient to detect changes in the species of interest (Baumgardt et al. 2019a, Baumgardt et al. 2019b). Each transect was sampled a minimum of 4 times based on recommendations from Baumgardt et al. (2019a; Table 2). In 2015, technicians conducted surveys from the third week of April to the third week of July. From 2016 to 2019, sampling began at the same time but ended during the last week of June.

For all years, transects were square or rectangular in shape but varied in the number of points per transect. The crew modeled this configuration after the United States Department of Agriculture's Multiple Species Inventory and Monitoring Program (Manley et al. 2006). From 2015 to 2017 each transect contained 12 points, and from 2018 to 2019 the number increased to

16. Though 12 points was sufficient to detect changes in species of interest, the crew increased the number of points due to more personnel and higher efficiency (Baumgardt 2019a; Table 1); this change did not bias the data that they subsequently collected. Observation points were spaced 400 m apart (to minimize likelihood of sampling same individuals at multiple locations) in a circuit such that the first point was 400 m from the last point. The point arrangement allowed the starting point for each transect to be alternated so that the same 2 points were not always the start or end point for the transect, which should better facilitate determination of site-specific effects from time of day on detection probability estimates. Additionally, the configuration maximized efficiency as little time was wasted returning to vehicles upon completion, thus allowing for increased data collection.

Technicians used a 10-minute duration for each point, such that they could survey each transect in its entirety in a single day between 0.5 hr before sunrise (or until lighting conditions were favorable) and ~ 1200 hrs. The purpose of this timeframe was to survey when the majority of birds were active in the mornings, while arriving early enough to sample some nocturnal species as well. Upon reaching each point, observers waited 2 minutes before conducting surveys to allow birds to settle after initial disturbance from entering the observation point (Rosenstock et al. 2002), and they did not survey during inclement weather (Baumgardt et al. 2019a). Technicians used a form of double sampling from 2015 to 2017 where they collected data simultaneously but independently of each other. I could not analyze data in a double sampling platform because there was no way to match or separate detections between observers. To scale back monitoring efforts and direct focus more towards impact assessments, technicians switched to a single observer method in 2018 (Table 2). Changes in sampling protocol did not bias results

I report herein because all methods met or exceeded the detection and sampling criteria specified in Baumgardt et al. (2019a, 2019b).

Technicians also recorded all birds observed by sight or sound (Hutto et al. 1986) within a 200 m radius using binoculars as necessary. They collected data for detected individuals that included: species code and distance (with a rangefinder); which period the observer was in (increments of 2 minutes; $n = 4$); if the bird was in a flock or flying over (flocks were treated as single observations with number of individuals estimated and location of the center of the flock used for estimating distance; flyovers were not assigned a distance); how many individuals (if > 1); and any other notes pertaining to the observation. From 2015 to 2018, observers recorded all individuals that were just seen, or both seen and heard as visual observations, while individuals solely heard were recorded as aural observations. In 2019, they made a modification to the protocol that allowed the observers to record "both" for individuals seen and heard. Once they completed the survey at a point, they immediately walked to the next point with the aid of a handheld global positioning system (GPS) unit and repeated the process until they surveyed all points.

Analyses

I estimated species richness and species density values per year per treatment. I estimated richness values with Microsoft Excel and R (R Core Team 2017), and density values using Program Distance, version 7.4 (Thomas et al. 2010). Distance is a comprehensive computer software program that provides a platform for analyzing distance data and can give users density estimates by region or treatment while accounting for variable detection probability (Buckland et al. 1993, Johnson et al. 2010, Thomas et al. 2010). I used Program Distance to calculate density estimates of 6 species stratified by treatment and year and I analyzed each species in individual analyses. Within Distance, I assessed data fit to the following models: uniform key with cosine adjustments; half-normal key individually and with cosine and Hermite polynomial adjustments; and hazard-rate key individually and with simple polynomial adjustments (Thomas et al. 2010). I used the 4 following methods for each species to maximize potential of selecting the best model fit of the detection function: (1) global and (2) stratified multiple covariate distance sampling (henceforth MCDS) models, and (3) global and (4) stratified conventional distance sampling (henceforth CDS) models (Marques et al. 2007). Stratified methods generated different models using data separated by stratum (individual treatment years), and global methods generated a single model using all of the data (combined treatments and years). Selecting global models may have introduced bias due to ignoring potential differences between strata.

I selected suitable yet parsimonious detection functions by choosing models according to Akaike's information criterion (henceforth AIC), which takes into account the model fit and number of parameters (i.e. model complexity; Buckland et al. 1993: 75-76). Lower ΔAIC values indicated better model fit with fewer parameters. I also examined the results of 3 goodness-of-fit tests: Kolmogorov-Smirnov (K-S), Cramer VonMises (CvM unif), and Cramer VonMises (CvM cos), specifically focusing on the visual model fit near the origin of the detection function (Massey Jr. 1951: 69, Buckland et al. 1993: 76-77). High goodness-of-fit test results ($P > 0.05$) indicated that the data matched well with the expected values and they were not significantly different. In addition, I noted the coefficient of variation (CV) values as low values indicated that the standard deviation was small compared to the mean and thus, made it easier to detect statistical significance compared to models with higher CV values (Conquest 1983: 209). After selecting the best fit model in Distance, I generated detection probabilities ($\hat{p}$) and density estimates (D) for each species in each treatment each year. When I selected global detection function models, I then applied that model to the observations within each stratum (treatment x year) to calculate density estimates. Due to the underlying monitoring project not being designed for detections based on treatment x year stratification, some of the representative species had low numbers of observations per stratum (Table 1).

For the following statistical tests using both the density and richness data, I set alpha at 0.05 and compared test results to determine statistical significance. I determined if an analysis of variance (henceforth ANOVA) test could ensue based on whether the following assumptions of ANOVA were met: (1) independence, (2) normality, and (3) equality of variances (Lee and Ahn 2003: 988). I considered each treatment site a unique observational unit and assumed that the species within each was affected independently by the given treatment. Even though I used a single global detection model to calculate density estimates by treatment x year, I worked under the assumption of independence. To test if the data were normally distributed, I performed a Shapiro-Wilk test (Shapiro and Wilk 1965), where a P-value > 0.05 indicated normal distribution. If the result was $P < 0.05$, I attempted to transform the data by adding a constant (1), transforming by $\log_{10}$, and performing the test again.

I proceeded with a Levene's test (Levene 1960; Lim and Loh 1996) to determine if the assumption of equal variance was violated. Levene's test results where the P-value was < 0.05 indicated non-equal variance, and thus, I transformed the data in the same way as previously mentioned in attempt to change the test outcome. If none of the assumptions were violated, I then proceeded with an ANOVA test. To increase the power of my results, I used 2 single factor ANOVA tests instead of 1 2-way ANOVA to separately test the potential differences between treatments and between years (Park and Schultz 1999). An ANOVA F-statistic greater than the critical value indicated a difference between groups, and the P-value indicated if that difference was statistically significant. For species that could undergo ANOVA tests, I followed with a 2-way ANOVA table in R to further represent the data and signify whether differences occurred between treatments, years, or treatment years.

If any assumption of the ANOVA test was violated, I instead used a Friedman's rank sum test, which is a 1-way repeated measures non-parametric statistical test (Ratto et al. 1998: 525). Significant *P*-values indicated that statistically significant differences between means of richness or density estimates occurred. After either an ANOVA or Friedman's rank sum test, I performed a Wilcoxon signed-rank Pairwise Comparison test (Woolson 2005: 1, Rosner et al. 2006) to determine where potential differences occurred and if they were between treatments or years. This test identified treatment combinations whose richness or density values contributed significantly to the previous test's implication of treatment differences. Ideally, this test should have revealed a pattern that indicated a consistent impact within a particular treatment (i.e. if both CH and CM treatments were different than the control, then a continuous grazing system may have had an impact). For both the density and richness estimates, I created line graphs to visually assess the data (Shah et al. 1995). I looked for any treatment years in which the data trajectories did not follow the same pattern (indicating differential change in magnitude; Latta et al. 2011). Due to some line graph plots being difficult to interpret when line overlapping occurred, I also created box plots as an additional method of graphical analyzation (Williamson et al. 1989). After I obtained all of the results, I inferred whether any statistically significant differences may have signified biologically significant changes by evaluating the specific species and their natural histories (EFSA Scientific Committee 2011).

4. RESULTS

Richness

A Shapiro-Wilk test indicated that the data were not normally distributed ($P = 0.000$). I transformed the data by $\log_{10}$, and a Shapiro-Wilk test also indicated that the transformed data were not normally distributed ($P = 0.000$). A Levene's test indicated that the data did have equal variance ($P = 0.392$), but despite this, the first ANOVA assumption of normal distribution was violated and therefore I used Friedman's test instead of an ANOVA. The Friedman's Rank Sum test indicated that there were statistically significant differences between the means of at least 2 combinations in the richness dataset ($P < 0.001$). A Wilcoxon signed-rank pairwise comparison test (Table 3) generated significant *P*-values between continuous moderate and rotational moderate regimes ($P = 0.046$), continuous moderate and the reference ($P = 0.046$), as well as between continuous high and the reference ($P = 0.046$).

To visually assess the richness data set, I created line graphs of both rotational and continuous regimes compared to the references (Figure 7). Rotational treatments maintained higher richness values ($\bar{x} = 5.5$) through time compared to the continuous treatments ($\bar{x} = 4.8$). In addition, rotational treatments displayed higher or equal richness estimates, and continuous treatments displayed lower or equal estimates relative to the references for all years. The continuous high regime deviated from the expected richness values during the treatment years in 2016 and 2017, producing 1 species greater than expected when compared to the reference, suggesting that a continuous high regime increased grassland bird richness when the cattle were introduced to the landscape. The continuous moderate regime increased from 2018 to 2019 where the reference did not, and had 1 more species than expected in 2016, further suggesting that the continuous treatments fostered grassland bird richness.

Due to some line segments overlapping, I created a box plot to visually display the richness data (Figure 8). The box plot depicted rotational treatments collectively having an equivalent interquartile range compared to the reference, and the continuous treatments having lower values relative to the other treatments. Lastly, I created an additional line graph and a bar graph to visually assess the compiled richness data (Figure 9). The collective line chart depicted continuous treatments having lower and rotational treatments having higher richness values relative to the reference, similar to Figure 7 but on a singular graph for easier comparison. The bar chart also showed that the continuous treatments collectively had lower or equal richness values relative to all other treatments for all years.

Density

Northern Bobwhite

Through visually assessing the Northern Bobwhite data set and examining the goodness-of-fit test results in Distance, I determined the appropriate truncation distance to be 165 m. Using a 165 m truncation, I assessed data fit with several detection function models (Table 4), ultimately selecting a CDS global hazard rate key model with 0 adjustment terms (Figure 10). Even though the stratified models had lower ΔAIC values, upon visual examination, many of the strata contained heaping that resulted in potential biased model fit. In attempt to fill some of the heaping, I chose the best global model instead of a stratified one, though this choice may also have introduced bias due to potential differences between strata being ignored. This model had the following goodness-of-fit test results: K-S: 0.000, CvM unif: 0.001, CvM cos: 0.001. The model resulted in a detection probability ($\hat{p}$) of 0.41, %CV ($\hat{p}$) of 0.91, effective detection radius

(EDR) of 105.7 m, and %CV (EDR) of 0.45. Density estimates and other statistics are reported in Tables 5 and 31.

A Shapiro-Wilk test indicated that the data were normally distributed (P = 0.097) and a Levene's test indicated that the data had equal variance (P = 0.110). Because neither ANOVA assumption was violated, I used an ANOVA test to determine where differences occurred within the data set. The ANOVA test comparing treatments indicated that there was not a difference between treatments (F = 1.023, CV = 2.866, P = 0.420; Table 6). The ANOVA test comparing years indicated that there was a significant difference between mean density estimates (F = 10.149, CV = 2.866, P = 0.000; Table 7). An ANOVA table supported these findings, suggesting that there was a statistically significant difference between years but not treatments or treatment years (Table 8). For a visual comparison of the statistical test results, I also created line graphs (Figure 11) and a box plot (Figure 12). The visual aids agreed with the statistical results because density estimates appeared to similar across treatments.

Grasshopper Sparrow

Although a robust model fit was impossible with the grasshopper sparrow data set due to so few observations (n = 185) and some years lacking observations, I proceeded to obtain the best possible density estimates. Through visually assessing the data set and examining the goodness-of-fit test results in Distance, I selected the optimal truncation distance of 90 m due to heaping at 100 m. Using a 90 m truncation, I assessed data fit with several detection function models (Table 9), ultimately selecting a MCDS global hazard rate key model with 0 adjustment terms (Figure 13). Even though the stratified models had ΔAIC of 0.00 and 0.09 and better goodness-of-fit results, they were biased due to the small sample sizes in many treatment years. I selected the best global model because it was more informed and therefore likely a better depiction of reality compared to stratified models. The MCDS global model had the following goodness-of-fit test results: K-S: 0.000, CvM unif: 0.005, CvM cos: 0.005. This model resulted in a detection probability ($\hat{p}$) of 0.54, %CV ($\hat{p}$) of 9.82, effective detection radius (EDR) of 66.05 m and %CV (EDR) of 4.91. Density estimates and other statistics are reported in Tables 10 and 31.

A Shapiro-Wilk test indicated that the data were not normally distributed ($P < 0.000$). I transformed the data and a Shapiro-Wilk test also indicated that the data were not normally distributed ($P < 0.000$). A Levene's test indicated that the data did not have equal variance ($P = 0.013$). Due to both assumptions being violated, I used a Friedman's rank sum test which indicated that there were statistically significant differences between at least 2 of the means ($P < 0.001$). I then used a Wilcoxon signed-rank test for a pairwise comparison of the data to determine where differences occurred. The treatment comparison resulted in no statistically significant differences (Table 11), but the year comparison resulted in statistically significant differences between 2016 and 2017, 2017 and 2018, and 2017 and 2019 (Table 12). For a visual comparison of the statistical test results, I also created line graphs (Figure 14) and a box plot (Figure 15).

Dickcissel

Through visually assessing the Dickcissel data set and examining the goodness-of-fit test results in Distance, I selected the optimal truncation distance of 125 m. Using a 125 m truncation, I assessed data fit with several detection function models (Table 12) and selected a CDS global hazard rate key model with 0 adjustment terms (Figure 16). Although stratified models had lower ΔAIC values and better goodness-of-fit test results, many treatment years lacked observations and others had very few; this likely resulted in skewed results and therefore, I chose the best global model. This model had the following goodness-of-fit test results: K-S: 0.001, CvM unif: 0.025, CvM cos: 0.050, and resulted in a detection probability ($\hat{p}$) of 0.37, %CV ($\hat{p}$) of 3.21, effective detection radius (EDR) of 72.9 m and %CV (EDR) of 4.28. Density estimates and other statistics are reported in Tables 14 and 31.

A Shapiro-Wilk test indicated that the raw data and transformed data were not normally distributed ($P < 0.000$ for both results). A Levene's test indicated that the data did have equal variance ($P = 0.109$). Despite this, the ANOVA assumption of normal distribution was violated and thus, I used a Friedman's rank sum test which indicated that there were statistically significant differences between at least 2 of the means ($P < 0.001$). I then used a Wilcoxon signed-rank test for a pairwise comparison of the data to determine where differences occurred. The treatment comparison resulted in no statistically significant differences (Table 12), but the year comparison resulted in statistically significant differences between 2015 and all other years, as well as between 2017 and 2018 (Table 13). For a visual comparison of the statistical test results, I created line graphs (Figure 17) and a box plot (Figure 18), which clearly depicted 2015 as an anomaly year with relatively high species density.

Brown-headed Cowbird

Through visually assessing the Brown-headed Cowbird data set and examining the goodness-of-fit test results in Distance, I selected the optimal truncation distance of 125 m. Using a 125 m truncation, I assessed data fit with several detection function models (Table 17) and chose a CDS stratified model (global depiction: Figure 19) because there was not much heaping occurring per stratum (treatment x year) and there were enough observations in each stratum to provide reliable density estimates. In fact, stratification actually reduced the amount of visual heaping. This model had the following goodness-of-fit test results (K-S: 0.001-0.998, CvM unif: 0.007-0.950, CvM cos: 0.003-0.950). This model resulted in a detection probability ($\hat{p}$) between 0.13 and 0.66 and an effective detection radius (EDR) between 44.7 m and 101.6 m. Density estimates and other statistics are reported in Tables 18 and 31.

A Shapiro-Wilk test ($P = 0.364$) and Levene's test ($P = 0.330$) indicated that the data were normally distributed and had equal variance. Due to neither of the assumptions being violated, I used an ANOVA test to determine where differences occurred within the data set. The ANOVA tests comparing treatments and years indicated that there was not a difference between treatments ($F = 0.592$, $CV = 2.866$, $P = 0.672$; Table 19) nor years ($F = 2.416$, $CV = 2.866$, $P = 0.083$; Table 20). An ANOVA table agreed with these findings, suggesting that there were not statistically significant differences between years, treatments, or treatment years (Table 21). For a visual comparison of the statistical test results, I also created line graphs (Figure 20) and a box plot (Figure 21).

Eastern Meadowlark

Through visually assessing the data set and examining the goodness-of-fit test results in Distance, I selected the optimal truncation distance of 165 m. Using a 165 m truncation, I assessed data fit with several detection function models (Table 22), ultimately selecting a CDS global hazard rate key model with 0 adjustment terms (Figure 22). Stratified models had better model fit for some strata, however visual examination revealed that many models were informed by few observations and thus, I selected the more informed global model. The best global model used conventional distance sampling. This model had the following goodness-of-fit test results: K-S: 0.000, CvM unif: 0.001, CvM cos: 0.001. This model resulted in a detection probability ($\hat{p}$) of 0.48, %CV ($\hat{p}$) of 2.63, effective detection radius (EDR) of 114.12 m and %CV (EDR) of 1.31. Density estimates and other statistics are reported in Tables 23 and 31.

A Shapiro-Wilk test indicated that the raw and transformed data were not normally distributed (P = 0.000 and 0.001, respectively), and a Levene's test indicated that the data did not have equal variance (P = 0.006). Due to both of the ANOVA assumptions being violated, I used a Friedman's rank sum test which indicated that there were statistically significant differences between at least 2 of the means ($P < 0.001$). I then used a Wilcoxon signed-rank test for a pairwise comparison of the data to determine where differences occurred. Unlike all other analyzed species, the treatment comparison resulted in statistically significant differences between the rotational moderate treatment and all other treatments, as well as between the continuous high treatment and the control (Table 24). The year comparison resulted in no statistically significant differences (Table 25). For a visual comparison of the statistical test results, I also created line graphs (Figure 23) and a box plot (Figure 24).

Scissor-tailed Flycatcher

Through visually assessing the data set and examining the goodness-of-fit test results in Distance, I selected the optimal truncation distance of 180 m. Using a 180 m truncation, I assessed data fit with several detection function models (Table 26), ultimately selecting CDS stratified model (represented with a global model: Figure 25) because observations were plentiful per treatment year ($\bar{x}$ = 118). This CDS stratified model had the following goodness-of-fit test results: K-S: 0.047-0.991, CvM unif: 0.175-0.950, CvM cos: 0.175-0.950. The models resulted in detection probabilities ($\hat{p}$) between 0.10 and 0.31, and effective detection radius (EDR) between 57.7 m and 100.8 m. Density estimates and other statistics are reported in Tables 27 and 31.

A Shapiro-Wilk test indicated that the data were normally distributed (P = 0.156) and a Levene's test indicated that the data did have equal variance (P = 0.840). Due to neither of the

ANOVA assumptions being violated, I proceeded with an ANOVA test. The ANOVA test comparing treatments indicated that there was not a difference between treatments (F = 0.808, CV = 2.866, P = 0.534; Table 28) and The ANOVA test comparing years indicated that there was a significant difference between years (F = 4.696, CV = 2.866, P = 0.008; Table 29). An ANOVA table agreed with these findings, suggesting that there were statistically significant differences between years, but not treatments or treatment years (Table 30). For a visual comparison of the statistical test results, I also created line graphs (Figure 26) and a box plot (Figure 27).

Density Note

It is important to note that all density data sets for all representative assemblage species exhibited heaping or rounding of data (Buckland et al. 2001). The stepping pattern observed in the Q-Q plots (Figures 10, 13, 16, 19, 22, and 25) showed that there was rounding in distance estimates and certain values were used more frequently than others. Because of this, only 7 out of 18 goodness-of-fit test results indicated a significant fit of the data to the model (Dickcissel CvM cos; Brown-headed Cowbird K-S, CvM unif, CvM cos; Scissor-tailed Flycatcher K-S, CvM unif, CvM cos). Furthermore, Brown-headed Cowbird and Scissor-tailed Flycatcher models were stratified so not all of the strata yielded goodness-of-fit results where $P > 0.05$. This means that the models were not optimally fitted, and this introduction of human error should be considered.

5. IMPLICATIONS

Because grassland birds have been declining regionally in Texas and around the world, conservation and management insights are crucial in the fight to preserve them. The leading hypothesis for this decline relates to the loss of grassland vegetation types, which can affect local grassland bird assemblages that rely on this unique vegetation type for survival and reproductive success. Grassland biotic diversity is maintained through fluctuating conditions that favor one group of species and then another (Mentis and Bailey 1990). For some species, common uniform ranching paradigms of continuous grazing and highly stocked systems are at odds with fostering diversity, as they focus on short-term solutions with high gain (Jansen et al. 1999). Implementing a combination of grazing intensities (continuous and rotational systems) with varied stocking rates may allow for the greatest diversity of grassland bird species across a given area, as it would support a variety of potential habitat requirements for a range of species. Even though many species may be unaffected by grazing regime, my study suggested that Eastern Meadowlarks are highly dependent upon rotational grazing systems, though stocking rate did not appear to affect them.

According to the State of the Birds report, the North American Eastern Meadowlark population was estimated to have declined by 74 million birds since 1970 (U.S. NABCI Committee 2019). In 2014, they were listed as a common bird in steep decline, and in 2019 they were considered a species of greatest conservation concern in 26 states (U.S. NABCI Committee 2014, 2019). Furthermore, more than 95% of the Eastern Meadowlark's distribution is on private lands, highlighting the importance of private land practices when it comes to the survival of this species (U.S. NABCI Committee 2011). Implementing rotational grazing systems in south Texas would likely foster density of local Eastern Meadowlarks and potentially other species not evaluated in my thesis, thus aiding in the support of grassland birds as a whole. These management recommendations must be made available to landowners in Texas if implementation is to be undertaken in an effort to conserve grassland birds.

6. CONCLUSIONS

Continuous grazing regimes may have had an impact on local grassland bird richness due to both continuous high and moderate regimes supporting higher richness than expected relative to the reference (Figure 7, Table 3). This would indicate that continuous grazing systems increased grassland bird richness, though this is contradicting to previous research that shows continuous systems as less suitable environments for grassland birds compared to rotational systems (Hull 2002). This contradiction may suggest that my study didn't have enough species to accurately gauge grassland bird species richness, or it is also possible that richness remained high when cattle were introduced because certain species benefit from cattle presence, which will be described further below.

Though the line graphs suggested that higher richness may come from continuous regimes when cattle were introduced to the landscape, the other visual aids, which combine the years, conclude the opposite and agree with the literature. For example, my box plot (Figure 8) depicted the rotational sites (regardless of stocking rate) with a similar pattern to the reference. Continuous sites, however, displayed a different pattern with lower richness compared to all other sites, especially when highly stocked. The collective line and bar charts (Figure 9) also supported the literature as the continuously grazed pastures displayed either equal or lower richness values over time compared to all other treatments. When visually comparing the rotational and continuous graphs (Figure 7), it was clear that the rotational regimes fostered higher richness throughout time and followed similar trajectories compared to the reference.

Previous research has suggested that grassland bird species richness decreases with increasing grazing intensity (Gonnet 2001, Kempema 2007), though some species may benefit from livestock presence (Vickery 1996, Hull 2002, Powell 2006, Coppedge et al. 2008, Hovick and Miller 2016). The Eastern Meadowlark was absent from 2 treatment years, the Dickcissel was absent from 8, the Grasshopper Sparrow was absent from 9, and the remaining analyzed species were present in all treatment years. All species were present in all treatments for at least 2 years. According to Kempema (2007), who studied grassland bird richness across 3 grazing intensities (low, moderate, and high), Brown-headed Cowbirds, Dickcissels, Grasshopper Sparrows, and Western Meadowlarks (*Sturnella neglecta*) were present in pastures with all 3 grazing intensities. Northern Bobwhites were only present in pastures with low or moderate grazing intensities, and Scissor-tailed Flycatchers were not studied (Kempema 2007).

Both richness and density estimates may have been irregular relative to other years due to a period of drought in 2017. Both continuous and rotational richness treatments (Figures 7 and 9) did not appear to be affected differently in 2017 compared to previous years, though 2018 and 2019 both exhibited lower richness values than previous years. This change was potentially due to a drought since the reference treatment also decreased, indicating that the shift was not a result of grazing treatment on the CGRDA. Northern Bobwhite, Dickcissel, Brown-headed Cowbird, Eastern Meadowlark, and Scissor-tailed Flycatcher density estimates appeared to be unaffected by the drought (Figures 11, 17, 20, 23, and 26). Grasshopper Sparrow density estimates were collectively highest in 2017 (Figure 14), though for reasons described below, this may have been due to a small sample size.

All species except the Brown-headed Cowbird and Eastern Meadowlark exhibited statistically significant differences of density estimates between years. For this impact assessment, this information was of little value because I was not interested in whether densities fluctuated from year to year. Annual variation such as precipitation, drought, or vegetation changes may have impacted the species; even though this was not the focus of my study, relevant insights may be gleaned. For example, the Dickcissel results—with 2015 as an anomaly with higher species density—may have been non-orthogonal, i.e. some of the independent variables were correlated (such as presence of certain vegetation across years). Previous research has documented that Dickcissels prefer to nest in various species of thistle plants such as bull thistle (*Cirsium vulgare*) or Texas thistle (*Cirsium texanum*) (Patterson and Best 1996, Dechant et al. 2002). There may have been a higher abundance of thistles in 2015 compared to other years, which was perhaps reflected in the data as higher Dickcissel presence. If this hypothesis was true, future studies should note that this species' presence is highly dependent on thistle abundance and scientists should conduct research measures accordingly. For example, this species may require an adaptive management approach more so than other species due to their selective presence (McLain and Lee 1996).

Of the representative assemblage, grazing regime did not appear to impact 5 of the 6 species (Brown-headed Cowbird, Dickcissel, Grasshopper Sparrow, Scissor-tailed Flycatcher, Northern Bobwhite). Previous research agrees with some of these findings, documenting that Brown-headed Cowbirds (Goguen and Mathews 2001), Dickcissels (Zimmerman 1997), and Northern Bobwhites (Cantu and Everett 1982, though see Lusk et al. 2001) are not particularly impacted by grazing, but rather other factors such as precipitation (Northern Bobwhite) or presence/absence of cattle (Brown-headed Cowbird and Dickcissel). Contrary to my results, other research has suggested that Scissor-tailed Flycatchers (Stroppel 2009) and Grasshopper Sparrows (Vickery 1996, Powell 2006, Coppedge et al. 2008) can be impacted by grazing regime. The Eastern Meadowlark was the only species that displayed statistically significant differences of mean density estimates between treatments.

The Eastern Meadowlark showed differences between the rotational moderate regime and all other treatments, as well as between the continuous high regime and reference (Table 24), indicating that grazing regime may have had an impact on this species' density. A clear pattern appeared to be the impact of the rotational moderate grazing regime. The line graphs (Figure 23) and box plot (Figure 24) supported this finding as the rotational moderate regime produced higher density estimates relative to all other treatment years with the slight exception of 2017, where the rotational high slightly surpassed the rotational moderate regime. This species requires a patchy grassland landscape with a mosaic of successional stages due to its ability to support different life stages and nest survival (Hull 2002, Hovick and Miller 2016). Continuously grazed pastures may not have provided this mosaic landscape for the Eastern Meadowlarks, unlike the rotationally grazed pastures. When comparing high and moderate stocking rates, it appeared that the species preferred moderately stocked pastures but could also live in highly stocked pastures (Figures 23 - 24). These findings are echoed by other research which documented that grazing may increase Eastern Meadowlark numbers (Skinner et al. 1984, Hull 2002, Powell 2006, Stroppel 2009). These results were statistically significant and I speculate that they were biologically significant as well because continuous regimes resulted in Eastern Meadowlark densities approaching zero where other treatments fostered relatively higher densities (Figure 24).

My results supported the hypothesis that Brown-headed Cowbird density increases with cattle density, although neither rotationally nor continuously grazed pastures appeared to have an impact on the species (Figures 19-21, Table 19, Table 21). Unlike other species, the pastures with high stocking rates (blue lines) almost entirely produced higher density estimates compared to the moderately stocked pastures (with the exception of the rotational pastures in 2015). This finding suggested that a higher stocked pasture, whether it be rotationally or continuously grazed, resulted in higher Brown-headed Cowbird densities, though Taylor (1986) found the opposite. As this species is named, they have evolved a symbiosis (commensalism) with cattle and spend much of their time consuming insects that cattle kick up while rummaging through pastures (Abdi 1992) and my results make sense in this context.

Due to grazing and wallowing of livestock, previous research has found that Grasshopper Sparrow numbers increased in conditions with livestock because of the resulting bare ground patches that the species uses for foraging (Vickery 1996, Powell 2006, Coppedge et al. 2008). In 2017, there were more detections of this species than any other year (Table 1), perhaps due to environmental conditions such as precipitation, or due to technicians being better or lesser trained in 2017 compared to other years. The Grasshopper Sparrow data set was very sparse relative to other analyzed species (Table 1), which resulted in a poor-informed model that rendered potentially biased results due to the low sample size (Diettrich and Kong 1995). Hence, even though my results indicated no impact of grazing regime, it is possible that there was an impact, but the sample was too small to detect the change. My results elude to the fact that if this species is of particular interest in future studies, sampling efforts should be modified to collect sufficient data for a robust model—whether it be more survey effort or more specialized species-specific surveys—to negate undesirable effects of small sample size. Alternatively, if local assemblage numbers are similar, my results could encourage future researchers to not select the Grasshopper Sparrow as a representative species for a monitoring project that analyzes density due to the potential constraint of limited data.

REFERENCES

Abdi, S. W. 1992. Survival Through Symbiosis: Investigate these interactions. The Science Teacher 59(1):22-27.

Andrade, B. O., C. Koch, I. I. Boldrini, E. Vélez-Martin, H. Hasenack, J. M. Hermann, J. Kollmann, V. D. Pillar, and G. E. Overbeck. 2015. Grassland degradation and restoration: a conceptual framework of stages and thresholds illustrated by southern Brazilian grasslands. Natureza y Conservação 13(2):95-104.

Baumgardt, J. A., M. L. Morrison, L. A. Brennan, and T. A. Campbell. 2019a. Developing Rigorous Monitoring Programs: Power and Sample Size Evaluations of a Robust Method for Monitoring Bird Assemblages. Journal of Fish and Wildlife Management 10(2):480-491.

Baumgardt, J. A., M. L. Morrison, L. A. Brennan, B. L. Pierce, and T. A. Campbell. 2019b. Development of Multispecies, Long-Term Monitoring Programs for Resource Management. Rangeland Ecology & Management 72(1):168-181.

Bilotta, G. S., R. E. Brazier, and P. M. Haygarth. 2007. The impacts of grazing animals on the quality of soils, vegetation, and surface waters in intensively managed grasslands. Advances in agronomy 94: 237-280.

Bleho, B. I., N. Koper, and C. S. Machtans. 2014. Direct effects of cattle on grassland birds in Canada. Conservation Biology 28(3):724-734.

Boulinier T., J. D. Nichols, J. R. Sauer, J. E. Hines, and K. H. Pollock. 1998. Estimating species richness: the importance of heterogeneity in species detectability. Ecology 79:1018-1028.

Brennan, L.A., and W. P. Kuvlesky. 2005. North American grassland birds: an unfolding conservation crisis?. The Journal of Wildlife Management 69(1):1-13.

Bruno, A. 2018. Monitoring Vegetation and Northern Bobwhite Densities in a Grazing Demonstration Project in South Texas. PhD Dissertation for College of Graduate Studies at Texas A&M University – Kingsville.

Buckland, S. T., D. R. Anderson, K. P. Burnham, and J. L. Laake. 1993. *Distance Sampling: Estimating Abundance of Biological Populations*. Chapman and Hall, London. 446pp.

Buckland, S. T., D. R. Anderson, K. P. Burnham, J. L. Laake, D. L. Borchers, and L. Thomas. 2001. Introduction to distance sampling: estimating abundance of biological populations. Oxford University Press, Oxford, United Kingdom.

Cantu, R. and D. D. Everett. 1982. Reproductive success and broad survival of bobwhite quail as affected by grazing practices. National Quail Symposium Proceedings 2(1):14.

Cassidy, L. R., and G. Kleppel. 2017. The effect of grazing regime on grassland bird abundance in New York State. Northeastern Naturalist 24(8):86-98.

Calder III, W.A. 1990. The scaling of sound output and territory size: Are they matched?. Ecology 71(5):1810-1816.

Christensen, N. L. 1997. Managing for heterogeneity and complexity on dynamic landscapes. The ecological basis of conservation. Springer, Boston, MA. 167-186.

Conquest, L. L. 1983. Assessing the statistical effectiveness of ecological experiments: utility of the coefficient of variation. International Journal of Environmental Studies 20(3-4):209-221.

Coppedge, B. R., S. D Fuhlendorf, W. C. Harrell, and D. M. Engle. 2008. Avian community response to vegetation and structural features in grasslands managed with fire and grazing. Biological Conservation 141(5):1196-1203.

Davis, H. T., A. M. Long, J. A. Baumgardt, T. A. Campbell, and M. L. Morrison. 2019. Factors Affecting Nest Success and Predator Assemblage of Breeding Birds in Semiarid Grasslands. Rangeland Ecology and Management 72(2):385-395.

Dechant, J. A., M. L. Sondreal, D. H. Johnson, L. D. Igl, C. M. Goldade, A. L. Zimmerman, and B. R. Euliss. 2002a. Effects of management practices on grassland birds: Dickcissel. USGS Northern Prairie Wildlife Research Center 114.

Dechant, J. A., M. L. Sondreal, D. H. Johnson, L. D. Igl, C. M. Goldade, M. P. Nenneman, and B. R. Euliss. 2002b. Effects of management practices on grassland birds: Grasshopper Sparrow. USGS Northern Prairie Wildlife Research Center.

Derner, J. D., W. K. Lauenroth, P. Stapp, and D. J. Augustine. 2009. Livestock as ecosystem engineers for grassland bird habitat in the western Great Plains of North America. Rangeland Ecology and Management 62:111-118.

Diamond, D. D., and T. E. Fulbright. 1990. Contemporary plant-communities of upland grasslands of the coastal sand plain, Texas. Southwestern Naturalist 35(4):385–392.

Diefenbach, D. R., D. W. Brauning, and J. A. Mattice. 2003. Variability in grassland bird counts related to observer differences and species detection rates. The Auk 120(4):1168-1179.

Dietterich, T. G. and E. B. Kong. 1995. Machine learning bias, statistical bias, and statistical variance of decision tree algorithms. Technical report, Department of Computer Science, Oregon State University.

Drever, M. C., K. E. Aitken, A. R. Norris, and K. Martin. 2008. Woodpeckers as reliable indicators of bird richness, forest health and harvest. Biological Conservation 141(3):624-634.

EFSA Scientific Committee, 2011. Statistical significance and biological relevance. EFSA Journal 9(9):2372.

Ethier, D. M., N. Koper, and T. D. Nudds. 2017. Spatiotemporal variation in mechanisms driving regional-scale population dynamics of a Threatened grassland bird. Ecology and Evolution 7(12):4152-4162.

Fuhlendorf, S. D., D. M. and Engle. 2001. Restoring heterogeneity on rangelands: ecosystem management based on evolutionary grazing patterns: I propose a paradigm that enhances heterogeneity instead of homogeneity to promote biological diversity and wildlife habitat on rangelands grazed by livestock. BioScience 51(8):625-632.

Fuhlendorf, S. D., W. C. Harrell, D. M. Engle, R. G. Hamilton, C. A. Davis, and D. M. Leslie. 2006. Should heterogeneity be the basis for conservation? Grassland bird response to fire and grazing. Ecological Applications 16(5):1706-1716.

Fulbright, T. E., D. D. Diamond, J. Rappole, and J. Norwine. 1990. The coastal sand plain of southern Texas. Rangelands 12(6):337–340.

Gelfand, E. W., 2002. Mice are a good model of human airway disease. American Journal of Respiratory and Critical Care Medicine 166(1):5-6.

Goguen, C.B. and Mathews, N.E., 2001. Brown-headed cowbird behavior and movements in relation to livestock grazing. Ecological Applications 11(5):1533-1544.

Gonnet, J. M., 2001. Influence of cattle grazing on population density and species richness of granivorous birds (Emberizidae) in the arid plain of the Monte, Argentina. Journal of Arid Environments 48(4):569-579.

Gould, F. W., G. O. Hoffman, and C. A. Rechenthin. 1960. Vegetational areas of Texas. Leaflet/Texas Agricultural Extension Service. no. 492.

Graham, N. A., S. K. Wilson, S. Jennings, N. V. Polunin, J. A. N. Robinson, J. P. Bijoux, and T. M. Daw. 2007. Lag effects in the impacts of mass coral bleaching on coral reef fish, fisheries, and ecosystems. Conservation Biology 21(5):1291-1300.

Grzybowski, J. A. 1982. Population structure in grassland bird communities during winter. The Condor 84(2):137-152.

Hernández, F. and F. S. Guthery. 2012. Beef, brush, and bobwhites: quail management in cattle country. Texas A&M University Press.

Hovick, T. J. and J. R. Miller. 2016. Patch-burn grazing moderates eastern meadowlark nest survival in midwestern grasslands. The American Midland Naturalist 176(1):72-80.

Hull, S. D. 2002. Effects of management practices on grassland birds: Eastern Meadowlark. Grasslands Ecosystem Initiative Northern Prairie Wildlife Research Center U.S. Geological Survey Jamestown, North Dakota 58401.

Hutto, R. L., S. M. Pletschet, and P. Hendricks. 1986. A fixed-radius point count method for nonbreeding and breeding season use. The Auk 103(3):593-602.

Jansen, R., R. M. Little, and T. M. Crowe. 1999. Implications of grazing and burning of grasslands on the sustainable use of francolins (Francolinus spp.) and on overall bird conservation in the highlands of Mpumalanga province, South Africa. Biodiversity and Conservation 8(5):587-602.

Jensen, M. N. 2001. Can cows and conservation mix? Conservationists, ranchers, and scientists attempt to preserve biodiversity and the cowboy way of life. BioScience 51(2):85-90.

Johnson, D. S., J. L. Laake, and J. M. Ver Hoef. 2010. A model-based approach for making ecological inference from distance sampling data. Biometrics 66(1):310-318.

Kempema, S. L. F. 2007. The influence of grazing systems on grassland bird density, productivity, and species richness on private rangeland in the Nebraska Sandhills. Thesis Presentation. University of Nebraska.

Knopf, F. L. 1994. Avian assemblages on altered grasslands. Studies in avian biology 15(11).

Kohl, M. T., P. R. Krausman, K. Kunkel, and D. M. Williams. 2013. Bison versus cattle: are they ecologically synonymous?. Rangeland Ecology and Management 66(6):721-731.

Larson, D. M., W. K. Dodds, M. R. Whiles, J. N. Fulgoni, and T. R. Thompson. 2016. A before-and-after assessment of patch-burn grazing and riparian fencing along headwater streams. Journal of Applied Ecology 53(5):1543-1553.

Latta, S. C., B. A. Tinoco, P. X. Astudillo, and C. H. Graham. 2011. Patterns and magnitude of temporal change in avian communities in the Ecuadorian Andes. The Condor 113(1):24-40.

Levene, H. 1960. Robust testes for equality of variances. In Contributions to Probability and Statistics (I. Olkin, ed.) 278 292. Stanford Univ. Press, Palo Alto, CA. MRO 120709

Lim, T. S. and W. Y. Loh. 1996. A comparison of tests of equality of variances. Computational Statistics and Data Analysis 22(3):287-301.

Lockwood, M. W. and B. Freeman. 2014. The TOS handbook of Texas birds (No. 47). Texas A&M University Press.

Lusk, J. J., F. S. Guthery, and S. J. DeMaso. 2001. Northern bobwhite (Colinus virginianus) abundance in relation to yearly weather and long-term climate patterns. Ecological Modelling 146(1-3):3-15.

Manley, P. N. 2006. Multiple species inventory and monitoring technical guide (Vol. 73). United States Department of Agriculture, Forest Service.

Marques, T. A., L. Thomas, S. G. Fancy, and S. T. Buckland. 2007. Improving estimates of bird density using multiple-covariate distance sampling. The Auk 124(4):1229-1243.

Massey Jr, F. J. 1951. The Kolmogorov-Smirnov test for goodness of fit. Journal of the American Statistical Association 46(253):68-78.

McLain, R. J. and R. G. Lee. 1996. Adaptive management: promises and pitfalls. Environmental Management 20(4):437-448.

McLendon, T., M. J. Trlica, K. S. Stanley, K. D. Stanley, J. Aboaf, C. R. Pappas, J. A. Swinehart, C. Hindes, and D. E. Mergen. 2013. Classification and mapping of the vegetation of the San Antonio Viejo ranch, Jim Hogg and Starr Counties, Texas. Unpublished report.

Mentis M. T., and A. W. Bailey. 1990. Changing perceptions of ®re management in savanna parks. Journal of the Grassland Society of South Africa 7: 81-85.

Morrison, M. L., W. M. Block, M. D. Strickland, B. A. Collier, and M. J. Peterson. 2008. Wildlife study design. Second edition. Springer-Verlag, New York, N.Y.

Nichols J. D. 1992. Capture-recapture models – using marked animals to study population dynamics. BioScience 42:94–102.

NOAA 2016. Climate Data Online. Accessed on 4/2/20. Available from https://www.ncdc.noaa.gov/cdo-web/

Offer, D., M. Edwards, and P. Edgar. 2003. Grazing heathland: a guide to impact assessment for insects and reptiles. English Nature Research Reports No. 497. English Nature, Peterborough.

Ostfeld, R. S., S. T. A. Pickett, M. Shachak, and G. E. Likens. 1997. Defining the scientific issues. In the Ecological Basis of Conservation (pp. 3-10). Springer, Boston, MA.

Park, I. and R. W. Schutz. 1999. " Quick and Easy" Formulae for Approximating Statistical Power in Repeated Measures ANOVA. Measurement in Physical Education and Exercise Science 3(4):249-270.

Patterson, M. P. and L. B. Best. 1996. Bird abundance and nesting success in Iowa CRP fields: the importance of vegetation structure and composition. American Midland Naturalist 135(1):153-167.

Peterjohn, B. G., and J. R. Sauer. 1999. Population status of North American grassland birds from the North American breeding bird survey. Studies in Avian Biology 19: 27-44.

Peterson, G., C. R. Allen, and C. S. Holling. 1998. Ecological resilience, biodiversity, and scale. Ecosystems 1(1):6-18.

Powell, A. F. L. A. 2006. Effects of prescribed burns and bison (Bos bison) grazing on breeding bird abundances in tallgrass prairie. The Auk 123(1):183-197.

Ratto, A., A. H. El-Shaarawi, B. J. Dutka, C. Lopez, and C. Vega. 1988. Coliphage association with coliform indicators: a case study in Peru. Toxicity Assessment 3(5):519-533.

R Core Team. 2017. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/.

Read, J. L. and A. N. Andersen. 2000. The value of ants as early warning bioindicators: responses to pulsed cattle grazing at an Australian arid zone locality. Journal of Arid Environments 45(3):231-251.

Rosenberg, K. V., P. J. Blancher, J. C. Stanton, and A. O. Panjabi. 2017. Use of North American Breeding Bird Survey data in avian conservation assessments. The Condor: Ornithological Applications 119(3):594-606.

Rosenstock, S. S., D. R. Anderson, K. M Giesen, T. Leukering, and M. F. Carter. 2002. Landbird counting techniques: current practices and an alternative. The Auk 119(1):46-53.

Rosner, B., R. J. Glynn, and M. L. T. Lee. 2006. The Wilcoxon signed rank test for paired comparisons of clustered data. Biometrics 62(1):185-192.

Sanderson, M. A., S. C. Goslee, K. J. Soder, R. H. Skinner, B. F. Tracy, and A. Deak. 2007. Plant species diversity, ecosystem function, and pasture management—a perspective. Canadian Journal of Plant Science 87(3):479-487.

Shaffer, J. A., C. M.Goldade, M. F. Dinkins, D. H. Johnson, L. D. Igl, and B. R. Euliss. 2003. Brown-headed Cowbirds in grasslands: their habitats, hosts, and response to management. USGS Northern Prairie Wildlife Research Center, p.158.

Shah, P. and P. A. Carpenter. 1995. Conceptual limitations in comprehending line graphs. Journal of Experimental Psychology: General 124(1):43-61.

Shapiro, S.S. and M. B. Wilk. 1965. An analysis of variance test for normality (complete samples). Biometrika 52(3/4):591-611.

Shine, R. and X. Bonnet. 2000. Snakes: a new ‘model organism’ in ecological research?. Trends in Ecology and Evolution 15(6):221-222.

Sierra–Corona, R., A. Davidson, E. L. Fredrickson, H. Luna-Soria, H. Suzan-Azpiri, E. Ponce-Guevara, and G. Ceballos. 2015. Black-tailed prairie dogs, cattle, and the conservation of North America’s arid grasslands. PloS one 10(3).

Skinner, R. M., T. S. Baskett, and M. D. Blendon. 1984. Bird habitat on Missouri prairies. Terrestrial Series 14. Missouri Department of Conservation, Jefferson City, Missouri. 37 pages.

Soil Survey Staff 2015. Web Soil Survey. Accessed on 4/2/20. Available from https://websoilsurvey.sc.egov.usda.gov/App/HomePage.htm

Stanley, T. R. and F. L. Knopf. 2002. Avian responses to late-season grazing in a shrub-willow floodplain. Conservation Biology 16(1):225-231.

Stanton, R. L., C. A. Morrissey, and R. G. Clark. 2018. Analysis of trends and agricultural drivers of farmland bird declines in North America: A review. Agriculture, Ecosystems & Environment 254:244-254.

Stewart-Oaten, A., W. W. Murdoch, K. R. and Parker. 1986. Environmental impact assessment: "Pseudoreplication" in time?. Ecology 67(4):929-940.

Stroppel, D. J., 2009. Evaluation of patch-burn grazing on species richness and density of grassland birds. Doctoral dissertation, University of Missouri--Columbia.

Taylor, D. M. 1986. Effects of cattle grazing on passerine birds nesting in riparian habitat. Rangeland Ecology & Management/Journal of Range Management Archives 39(3):254-258.

Terlizzi, A., L. Benedetti-Cecchi, S. Bevilacqua, S. Fraschetti, P. Guidetti, and M. J. Anderson. 2005. Multivariate and univariate asymmetrical analyses in environmental impact assessment: a case study of Mediterranean subtidal sessile assemblages. Marine Ecology Progress Series 289:27-42.

Thomas, L., S. T. Buckland, E. A. Rexstad, J. L. Laake, S. Strindberg, S. L. Hedley, J. R. B. Bishop, T. A. Marques, and K. P. Burnham. 2010. Distance software: design and analysis of distance sampling surveys for estimating population size. Journal of Applied Ecology 47:5-14.

Underwood, A. J. 1992. Beyond BACI: the detection of environmental impacts on populations in the real, but variable, world. Journal of Experimental Marine Biology and Ecology 161(2):145-178.

U.S. NABCI Committee. 2011. The State of the Birds, United States of America, 2011. [Online.] U.S. Department of the Interior, Washington, D.C. Available at www.stateofthebirds.org/2011/.

U.S. NABCI Committee. 2014. The State of the Birds, United States of America, 2014. [Online.] U.S. Department of the Interior, Washington, D.C. Available at www.stateofthebirds.org/2014/.

U.S. NABCI Committee. 2019. The State of the Birds, United States of America, 2019. [Online.] U.S. Department of the Interior, Washington, D.C. Available at www.stateofthebirds.org/2019/.

Vickery, P. D. 1996. Grasshopper Sparrow (*Ammodramus savannarum*). *In* The Birds of North America, no. 239 (A. Poole and F. Gill, Eds.). Academy of Natural Sciences, Philadelphia, and American Ornithologists' Union, Washington, D.C.

Wiens, J. A. 1997. The emerging role of patchiness in conservation biology. In 'The Ecological Basis of Conservation: Heterogeneity, Ecosystems, and Biodiversity'. (Eds STA Pickett, RS Ostfeld, M. Shachak, and GE Likens.) pp. 93-107.

Williamson, D. F., R. A. Parker, and J. S. Kendrick. 1989. The box plot: a simple visual method to interpret data. Annals of Internal Medicine 110(11):916-921.

Woolson, R. F. 2005. Wilcoxon Signed-Rank Test. Encyclopedia of Biostatistics, 2nd Edition.

Zimmerman, J. L. 1997. Avian community responses to fire, grazing, and drought in the tallgrass prairie. Pages 167–180 *in* Ecology and Conservation of Great Plains Vertebrates (F. L. Knopf and F. B. Samson, Eds.). Springer-Verlag, New York.

APPENDIX A: FIGURES

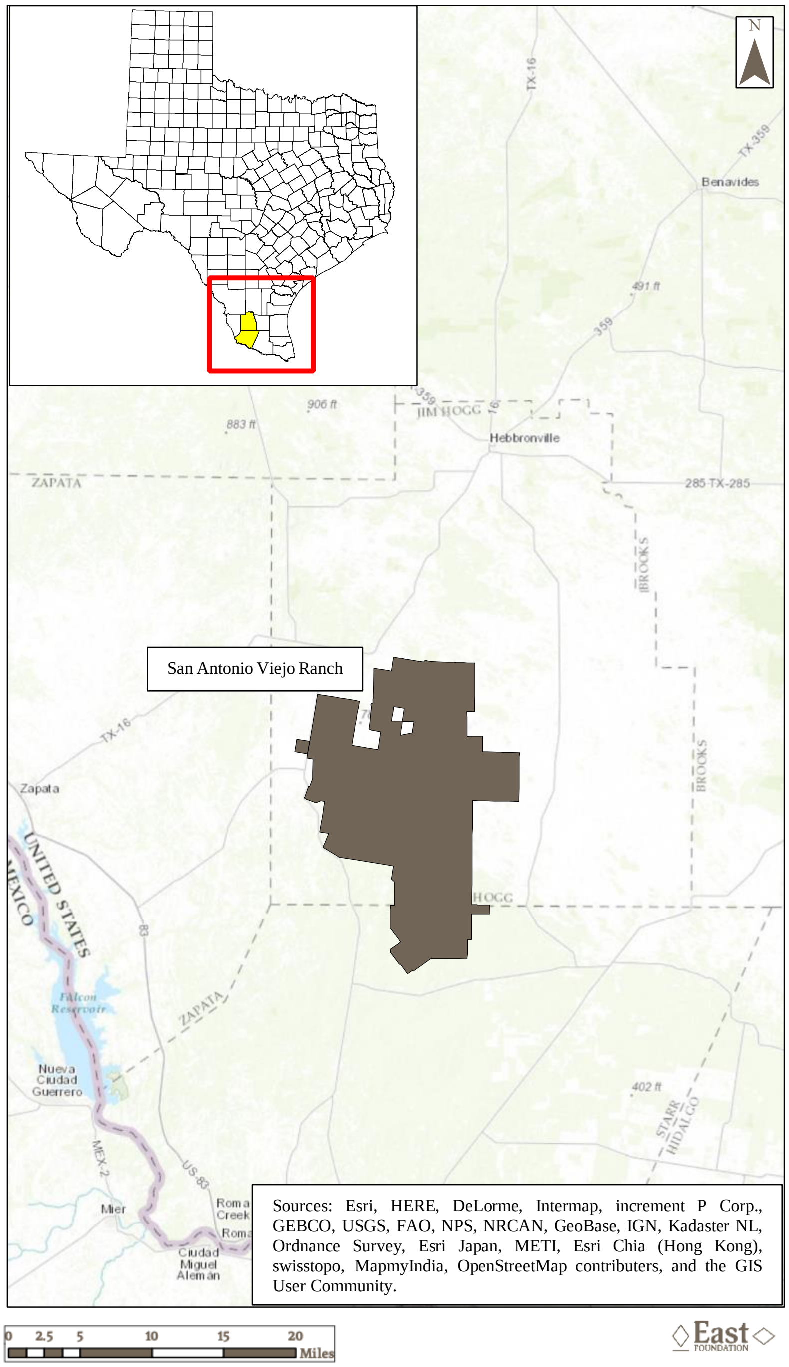


Figure 1. **San Antonio Viejo (SAV) Ranch.** This map depicts the location of the SAV ranch (604 km2; 60,496 ha) in south Texas, USA, part of Jim Hogg and Starr counties.

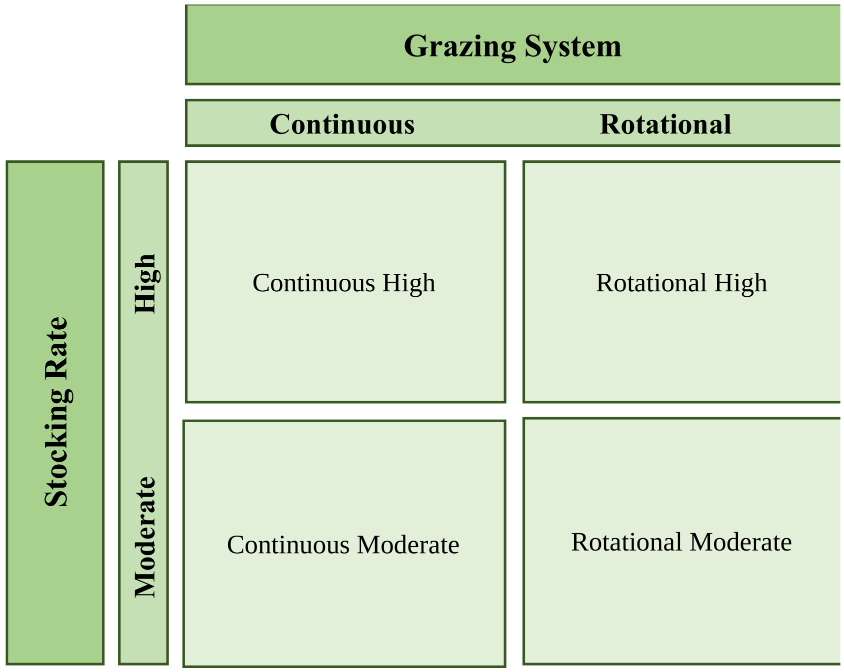


Figure 2. **Simplified Design.** This figure depicts a simplified version of my project's treatment variables: continuous and rotational grazing mixed with moderate (1 AU/ 20 ha) and high (1 AU/ 14 ha) stocking rates.

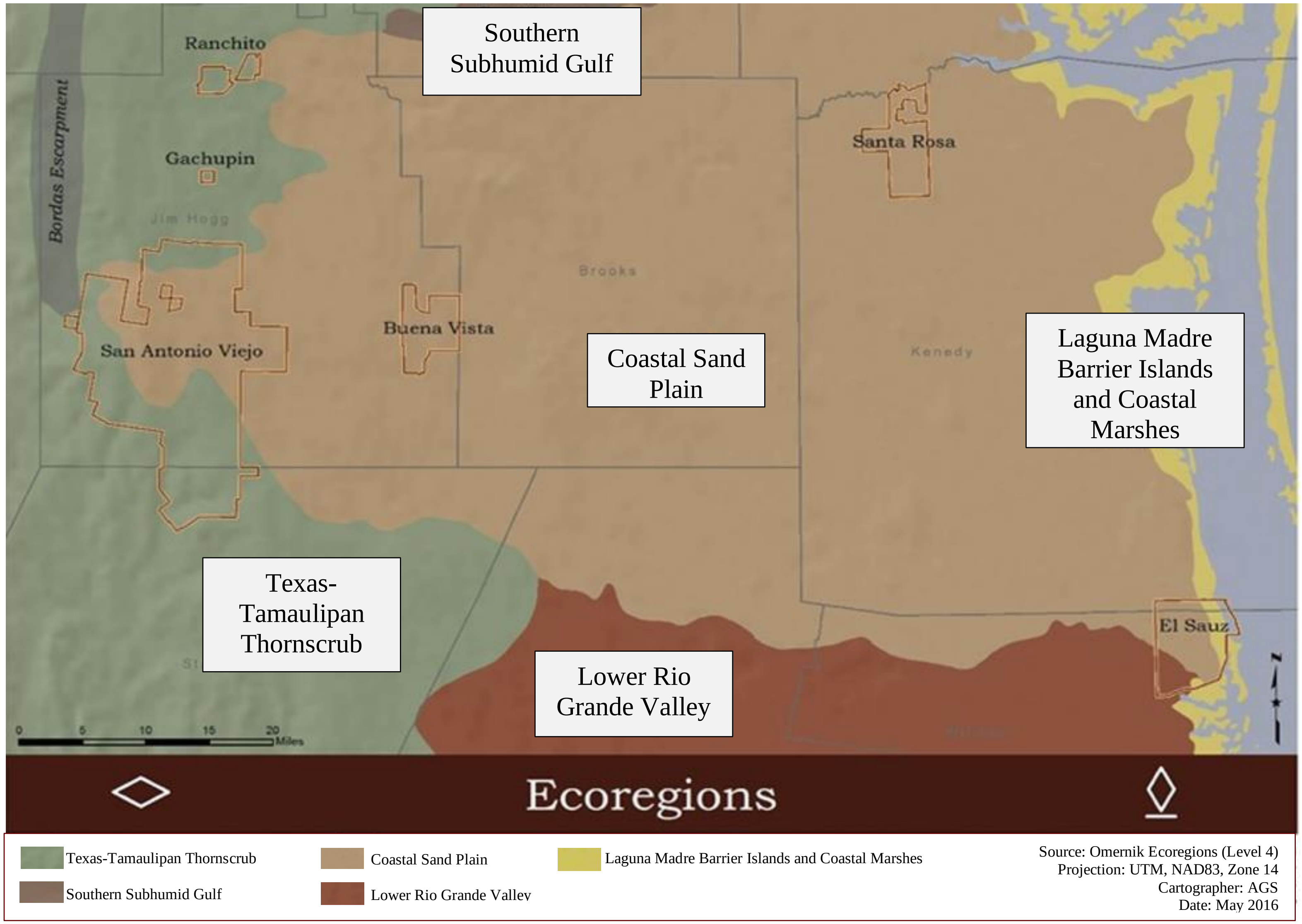


Figure 3. **Ecoregions.** This map depicts the level-4 ecoregions on the SAV ranch. Approximately half of the ranch was in the Coastal Sand Plain Ecoregion and half in the Texas-Tamaulipan Thornscrub Ecoregion.

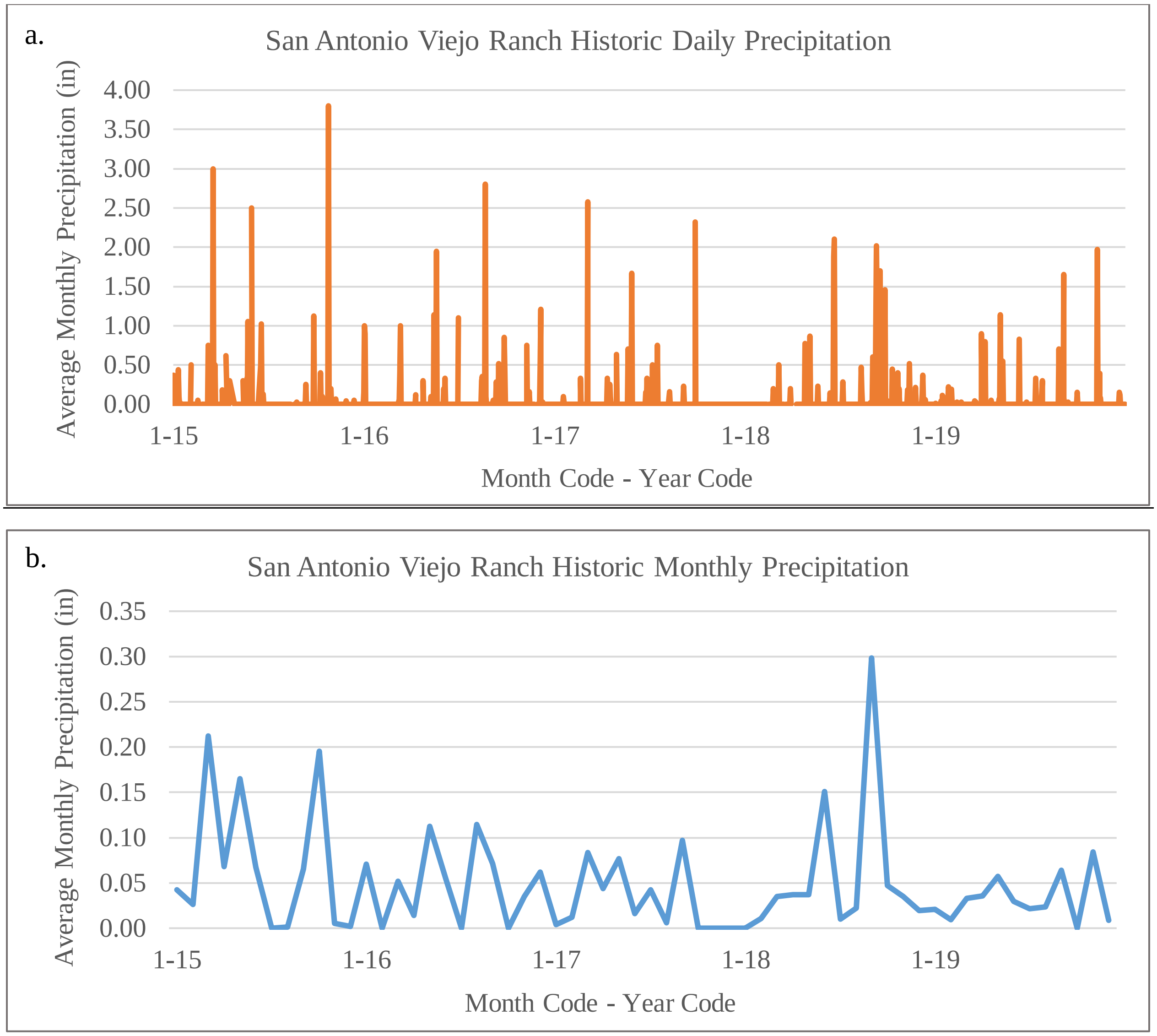


Figure 4. **Precipitation.** These figures represent the historic precipitation data for each day (a) and month (b) collected from stations "USC00412950" and "US1TXJH0010" from 1/2015 to 12/2019. Data accessed from NOAA climate data online on the 30th of March 2020. The figure on top shows the average daily precipitation and the figure on bottom shows the average monthly precipitation.

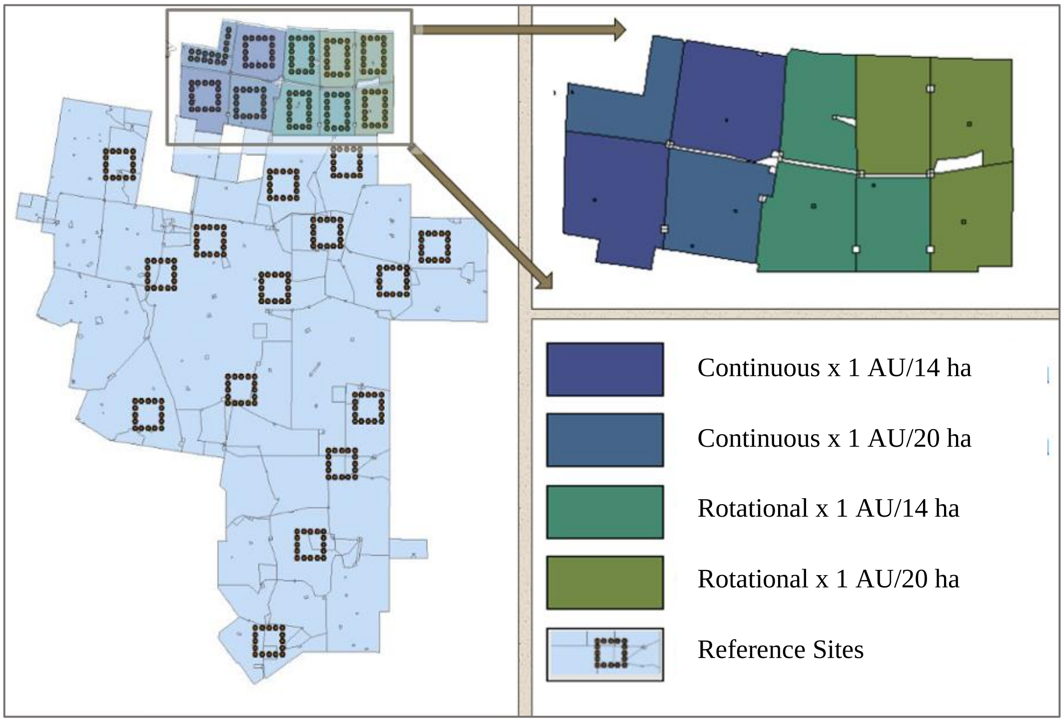


Figure 5. Study Site. This figure depicts the study boundary of the SAV ranch. Point count transects are displayed with 400 m between points from 2019, a year that represents the median number of reference transects. Treatment pastures are located on the north end of the ranch in the CGRDA pastures and the remaining area makes up the reference sites.

Figure 6. Representative Assemblage. This graphic depicts the strategically selected model grassland bird assemblage that I will use for my proposed project. Starting in the upper left and going right, there is the Dickcissel, Northern Bobwhite, and Grasshopper Sparrow. In the second row starting on the left is the Brown-headed Cowbird, Scissor-tailed Flycatcher, and Eastern Meadowlark.

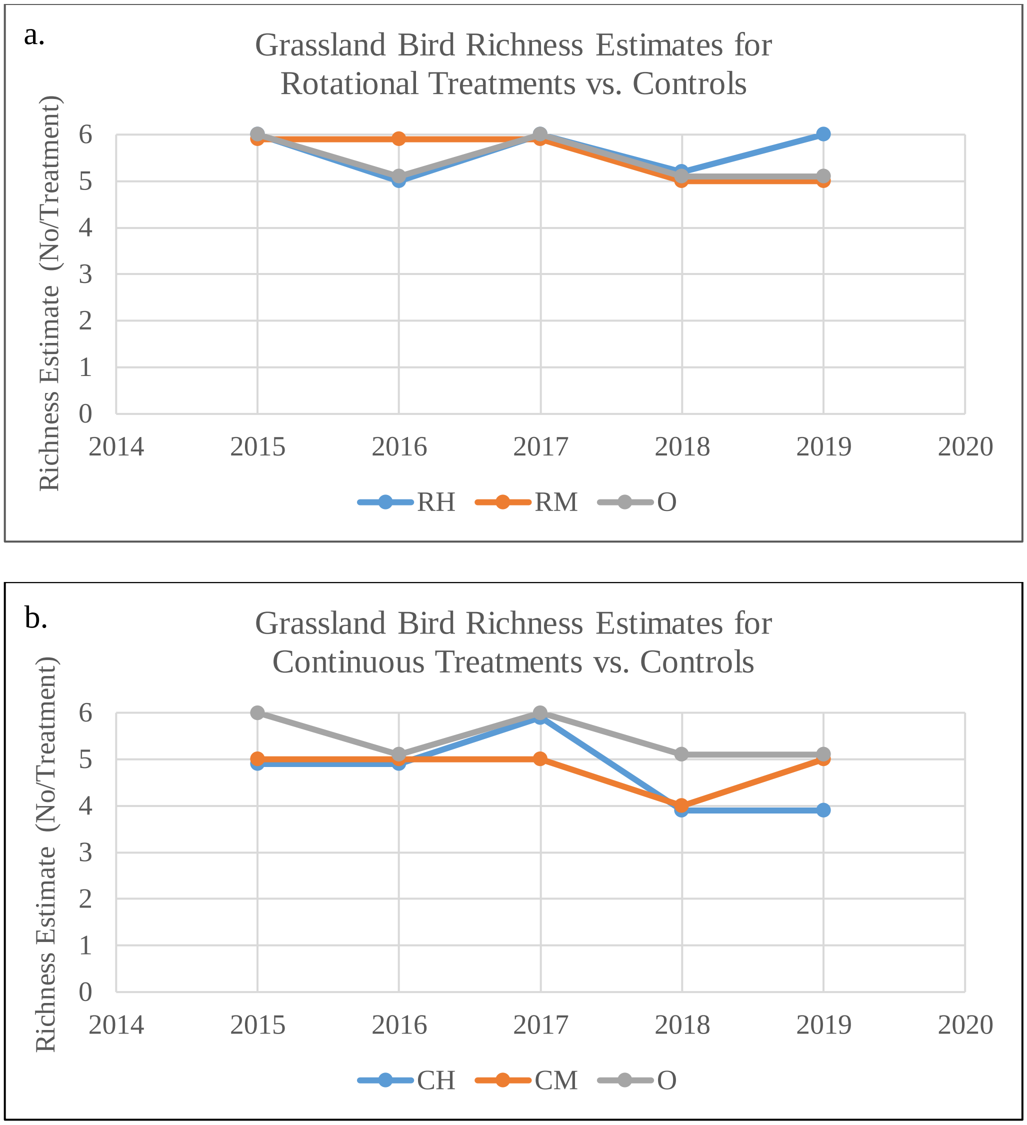


Figure 7. Grassland Bird Richness - Rotational vs. Continuous. These figures depict 2 charts representing the model grassland bird richness on rotationally grazed pastures compared to the references (a), and continuously grazed pastures compared to the references (b). These data are from point count surveys from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA (CH: continuous high; CM: continuous moderate; RH: rotational high; RM: rotational moderate; O: off the CGRDA pastures, i.e. control).

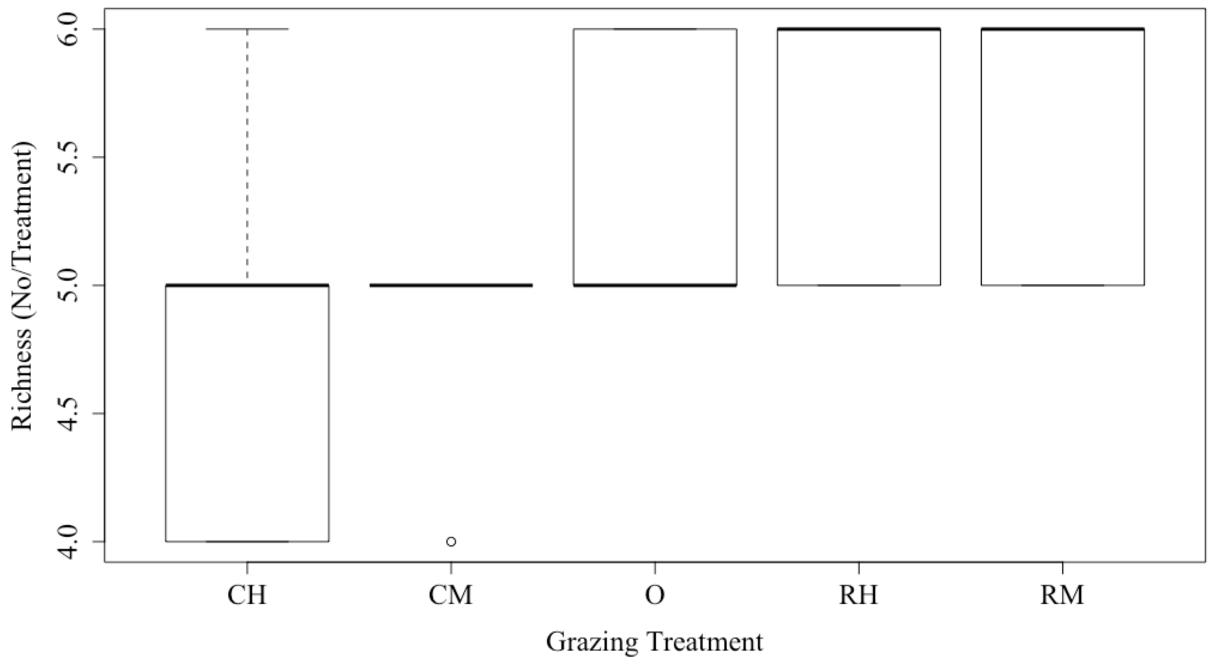


Figure 8. Model Grassland Bird Richness Box Plot. This box plot depicts the density estimates per treatment with interquartile range density estimates in boxes (25th to 75th percentile), bold horizonal lines as the median values, thin horizontal lines as the maximum and minimum values, and outliers as dots from data collected from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA (CH: continuous high; CM: continuous moderate; RH: rotational high; RM: rotational moderate; O: off the CGRDA pastures, i.e. control).

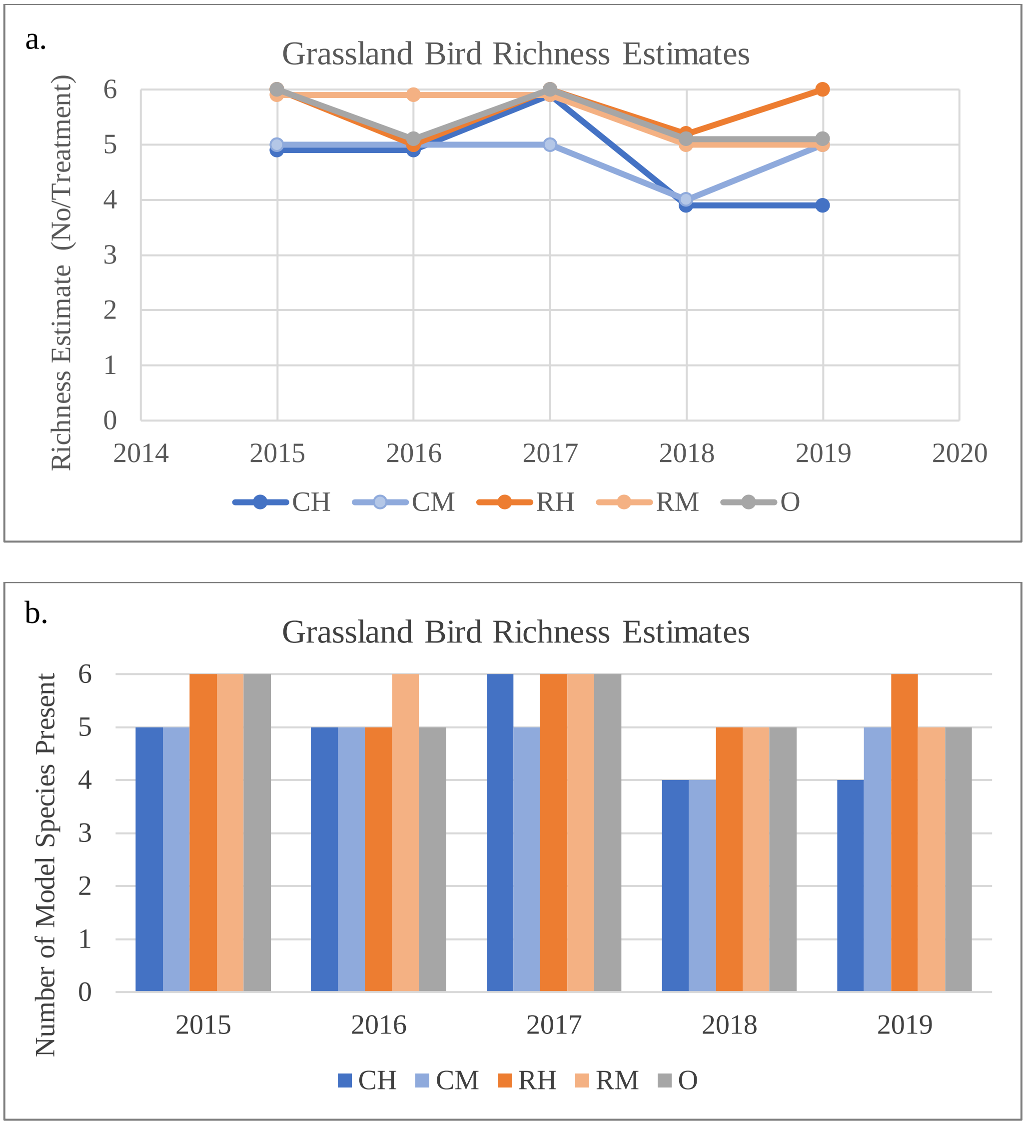


Figure 9. Grassland Bird Richness - All Treatments. These figures depict a line chart (a) and bar chart (b) for the model grassland bird richness across all treatments from point count survey data from 2015-2019 at the SAV ranch in Jim Hogg and Starr counties, Texas, USA (CH: continuous high; CM: continuous moderate; RH: rotational high; RM: rotational moderate; O: off the CGRDA pastures, i.e. control).

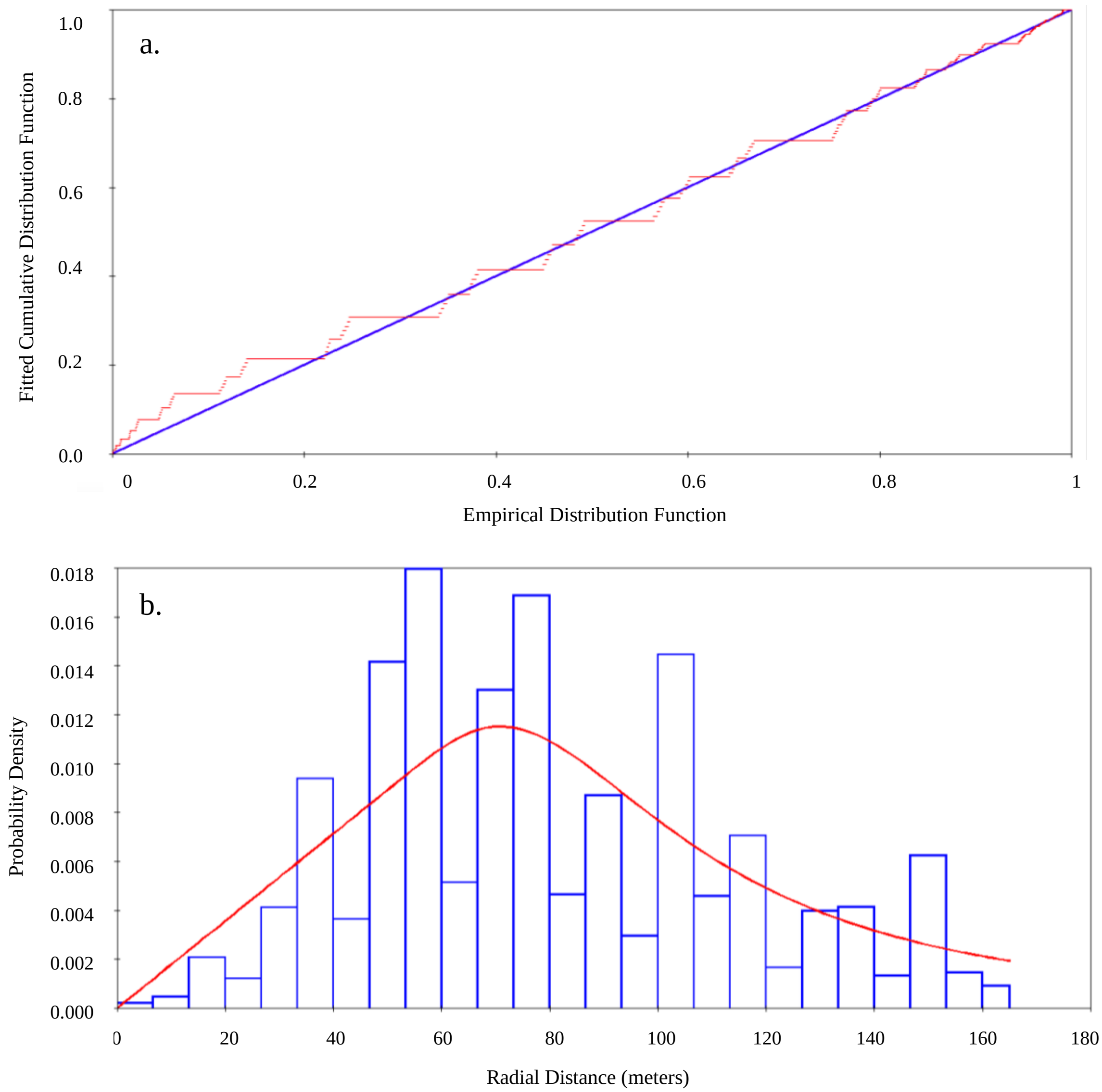


Figure 10. Northern Bobwhite Model. These charts depict data from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA. The first chart (a) depicts a Q-Q plot of the expected values in blue and the observed values in red. The second chart (b) depicts the transformed data in blue bins and the model as the red line.

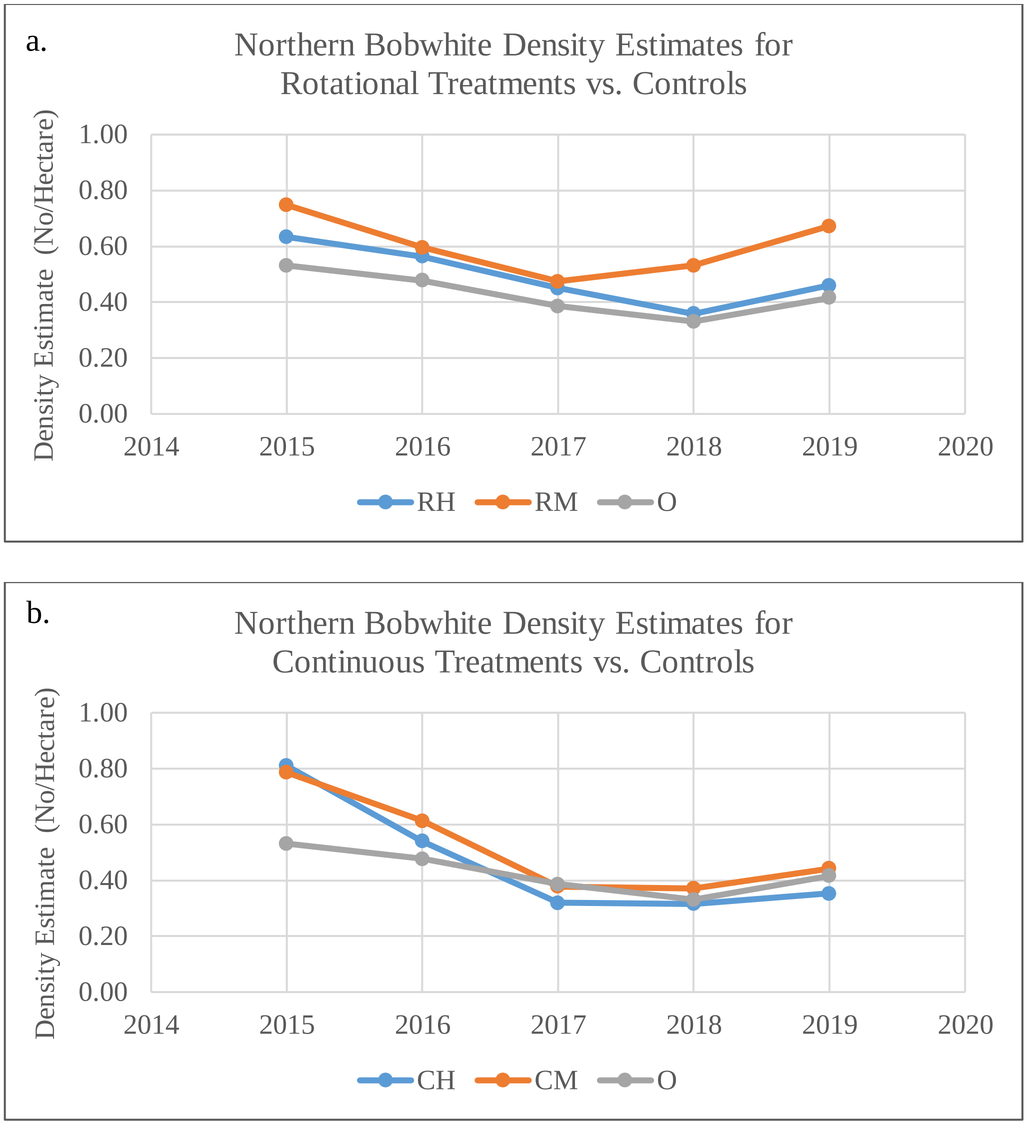


Figure 11. Northern Bobwhite - Rotational vs. Continuous. These figures depict 2 charts representing the Northern Bobwhite density on rotational treatments compared to the references (a) and continuous treatments compared to the references (b). These data are from point count surveys from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA (CH: continuous high; CM: continuous moderate; RH: rotational high; RM: rotational moderate; O: off the CGRDA pastures, i.e. control).

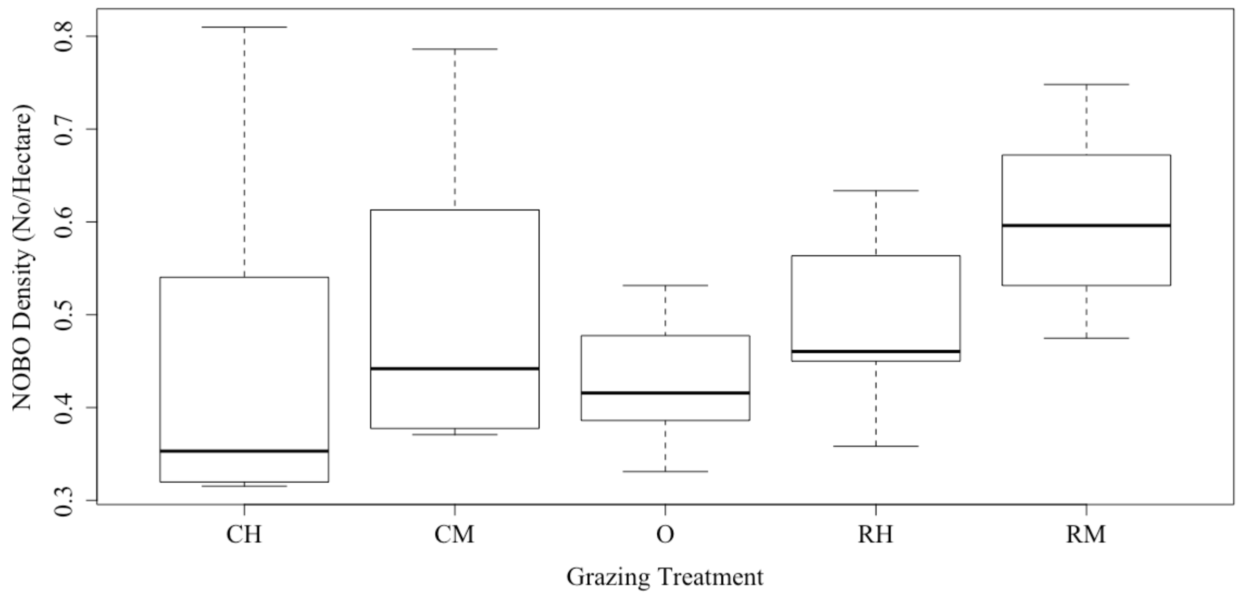


Figure 12. Northern Bobwhite Box Plot. This box plot depicts the density estimates per treatment with interquartile range density estimates in boxes (25th to 75th percentile), bold horizonal lines as the median values, thin horizontal lines as the maximum and minimum values, and outliers as dots from data collected from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA (CH: continuous high; CM: continuous moderate; RH: rotational high; RM: rotational moderate; O: off the CGRDA pastures, i.e. control).

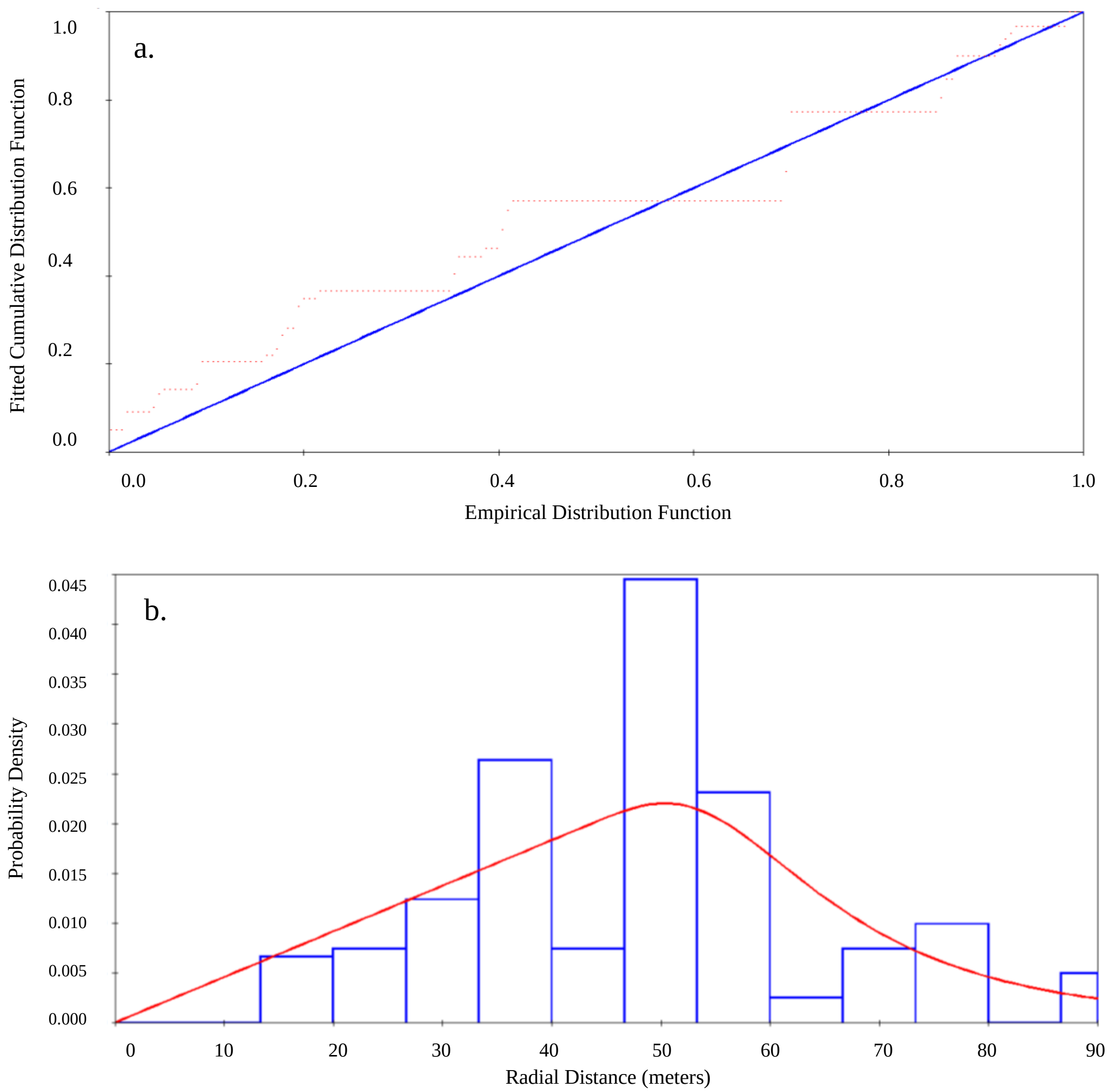
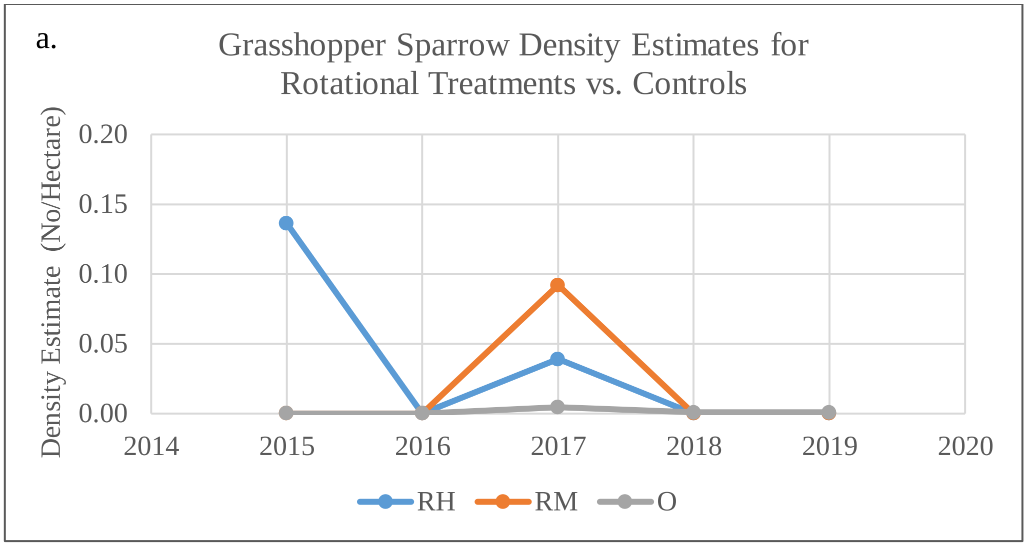


Figure 13. Grasshopper Sparrow Model. These charts depict data from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA. The first chart (a) depicts a Q-Q plot of the expected values in blue and the observed values in red. The second chart (b) depicts the transformed data in blue bins and the model as the red line.



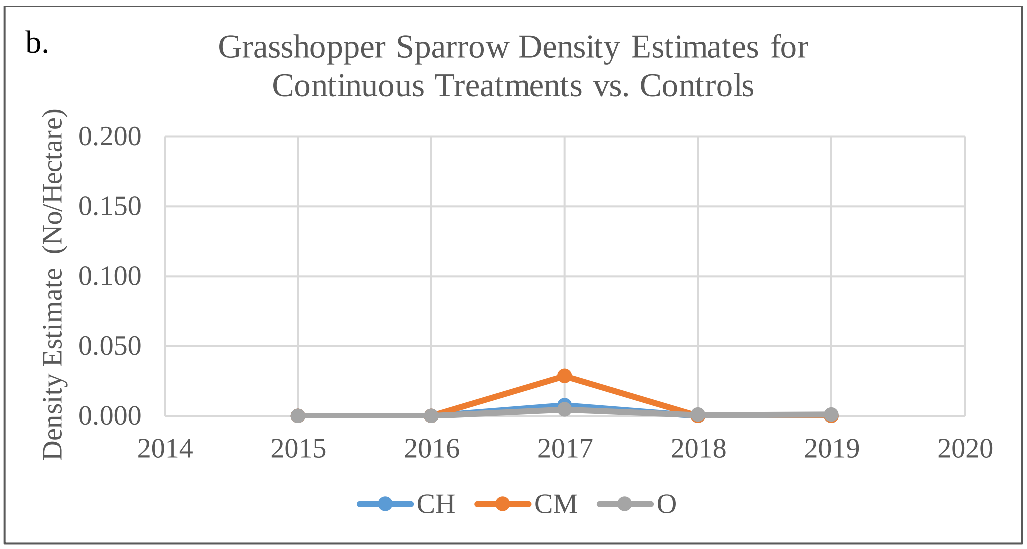


Figure 14. Grasshopper Sparrow - Rotational vs. Continuous. These figures depict 2 charts representing Grasshopper Sparrow density on rotational treatments compared to the references (a), and continuous treatments compared to the references (b). These data are from point count surveys from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA (CH: continuous high; CM: continuous moderate; RH: rotational high; RM: rotational moderate; O: off the CGRDA pastures, i.e. control).

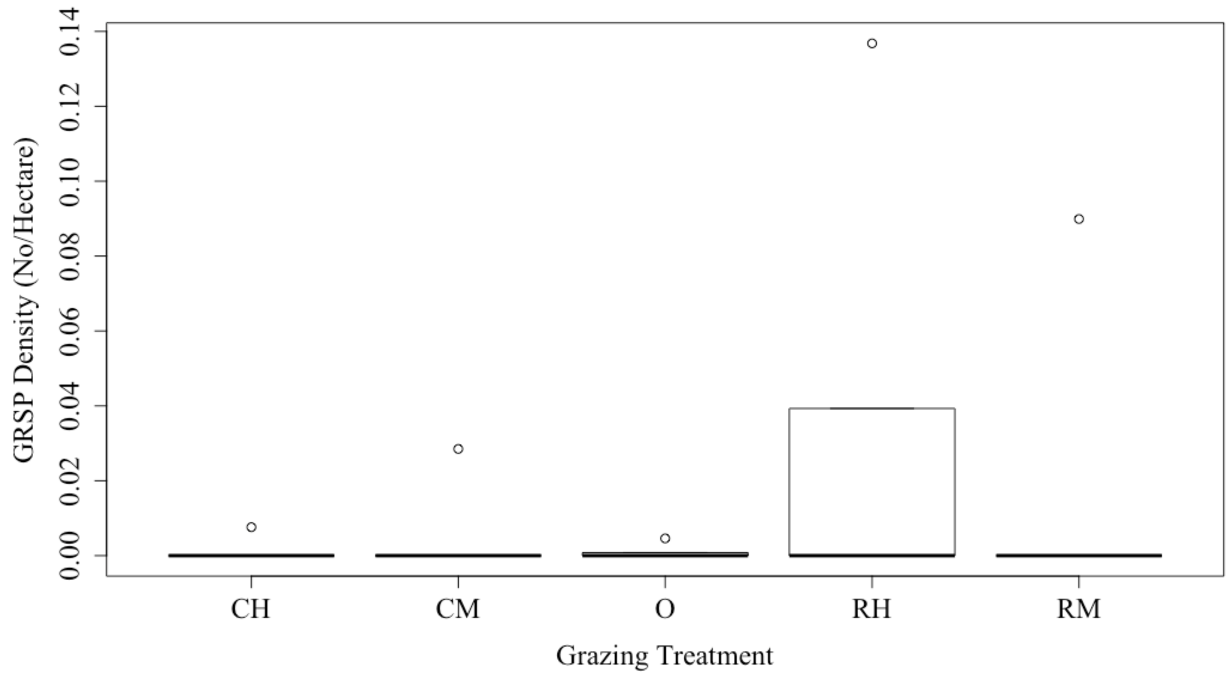


Figure 15. Grasshopper Sparrow Box Plot. This box plot depicts the density estimates per treatment with interquartile range density estimates in boxes (25th to 75th percentile), bold horizonal lines as the median values, thin horizontal lines as the maximum and minimum values, and outliers as dots from data collected from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA (CH: continuous high; CM: continuous moderate; RH: rotational high; RM: rotational moderate; O: off the CGRDA pastures, i.e. control).

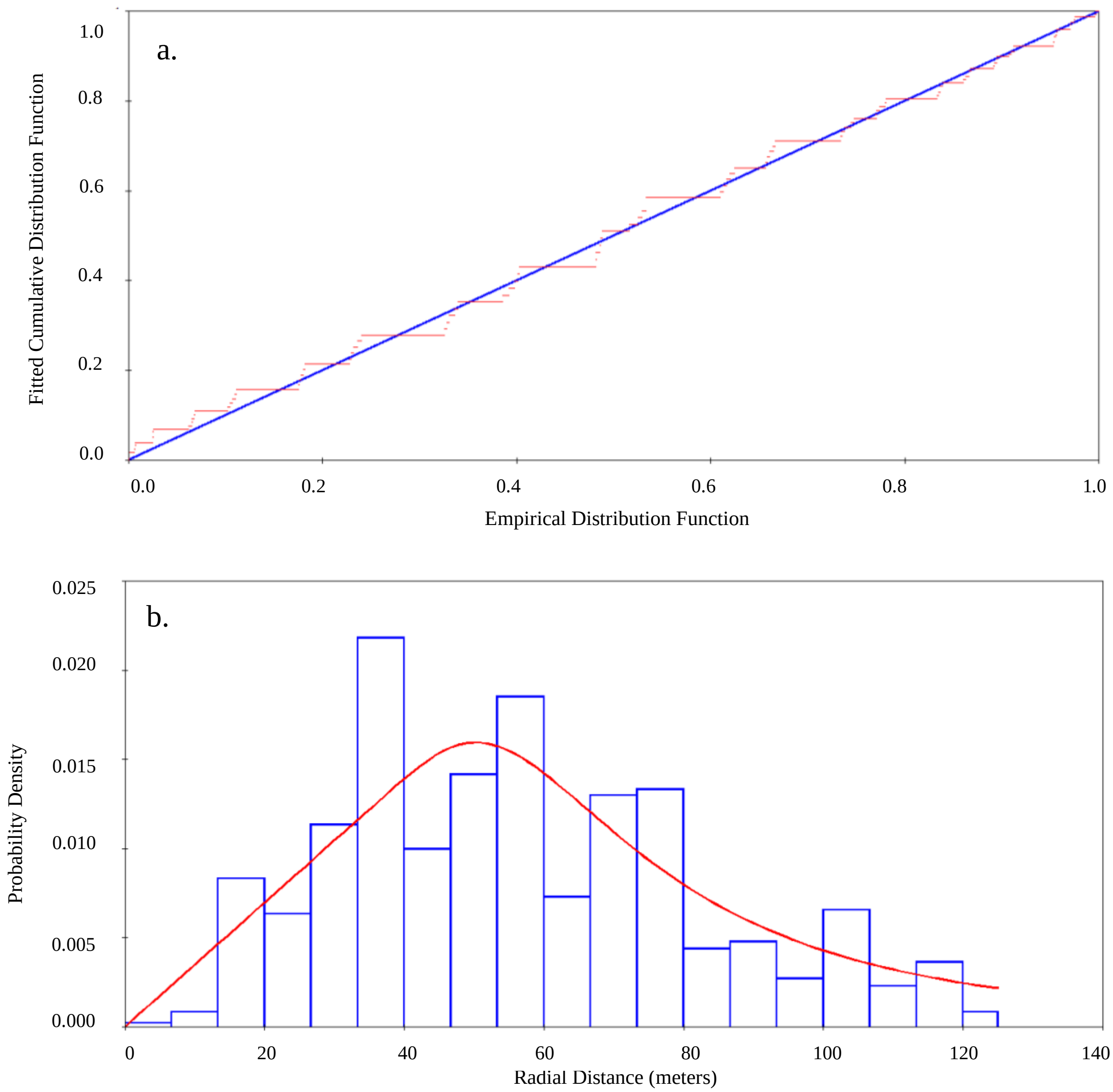
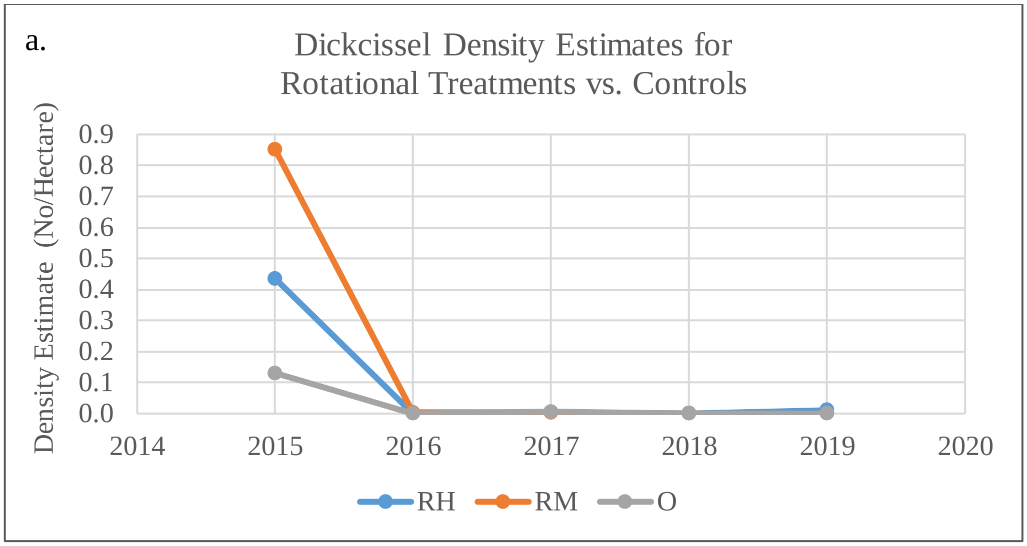


Figure 16. Dickcissel Model. These charts depict data from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA. The first chart (a) depicts a Q-Q plot of the expected values in blue and the observed values in red. The second chart (b) depicts the transformed data in blue bins and the model as the red line.



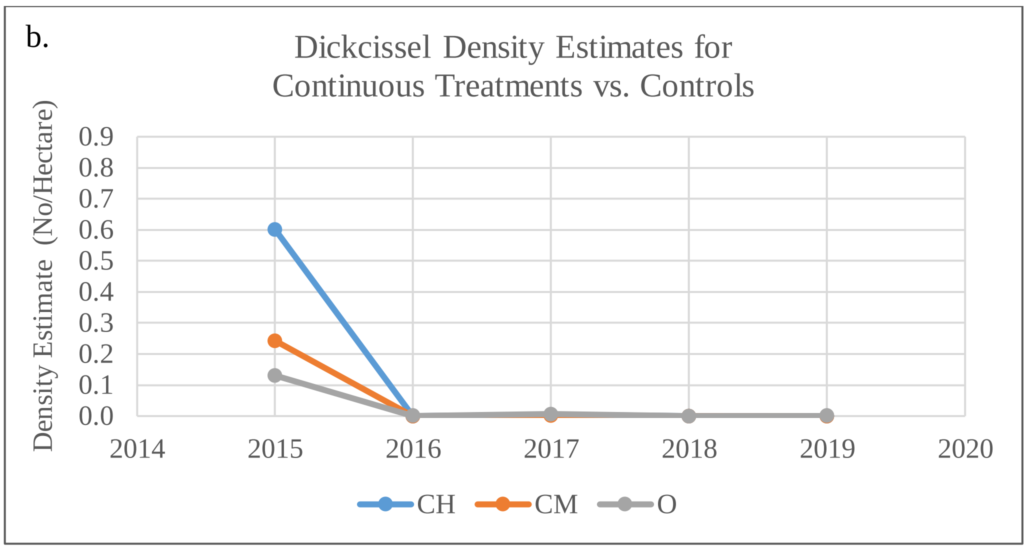


Figure 17. Dickcissel - Rotational vs. Continuous. These figures depict 2 charts representing Dickcissel density on rotational treatments compared to the references (a), and continuous treatments compared to the references (b). These data are from point count surveys from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA (CH: continuous high; CM: continuous moderate; RH: rotational high; RM: rotational moderate; O: off the CGRDA pastures, i.e. control).

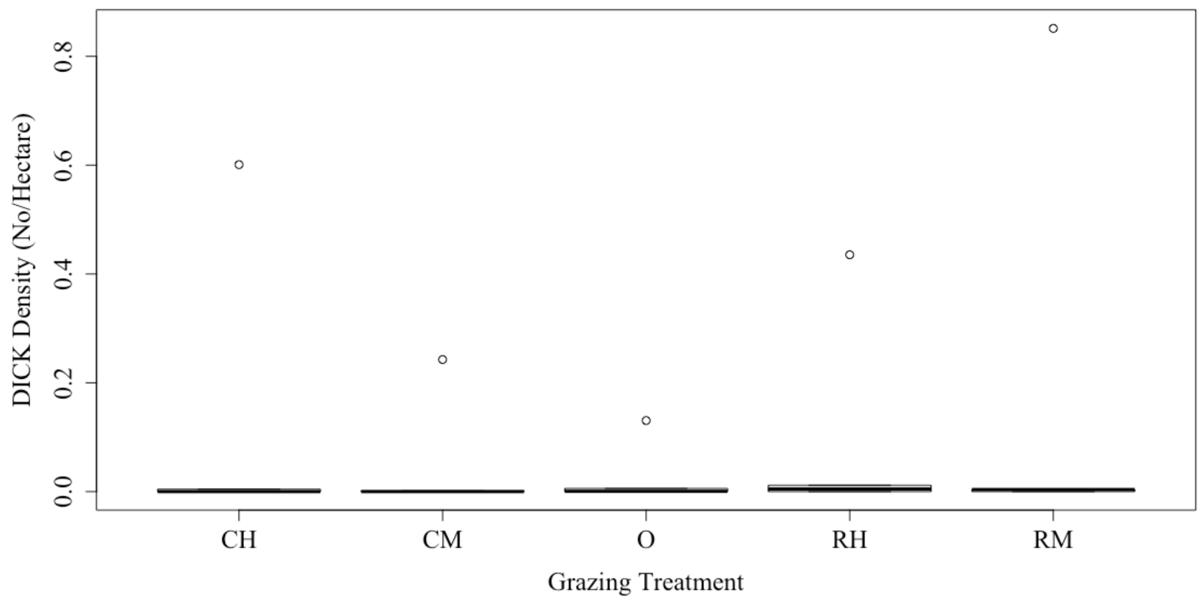


Figure 18. Dickcissel Box Plot. This box plot depicts the density estimates per treatment with interquartile range density estimates in boxes (25th to 75th percentile), bold horizonal lines as the median values, thin horizontal lines as the maximum and minimum values, and outliers as dots from data collected from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA (CH: continuous high; CM: continuous moderate; RH: rotational high; RM: rotational moderate; O: off the CGRDA pastures, i.e. control).

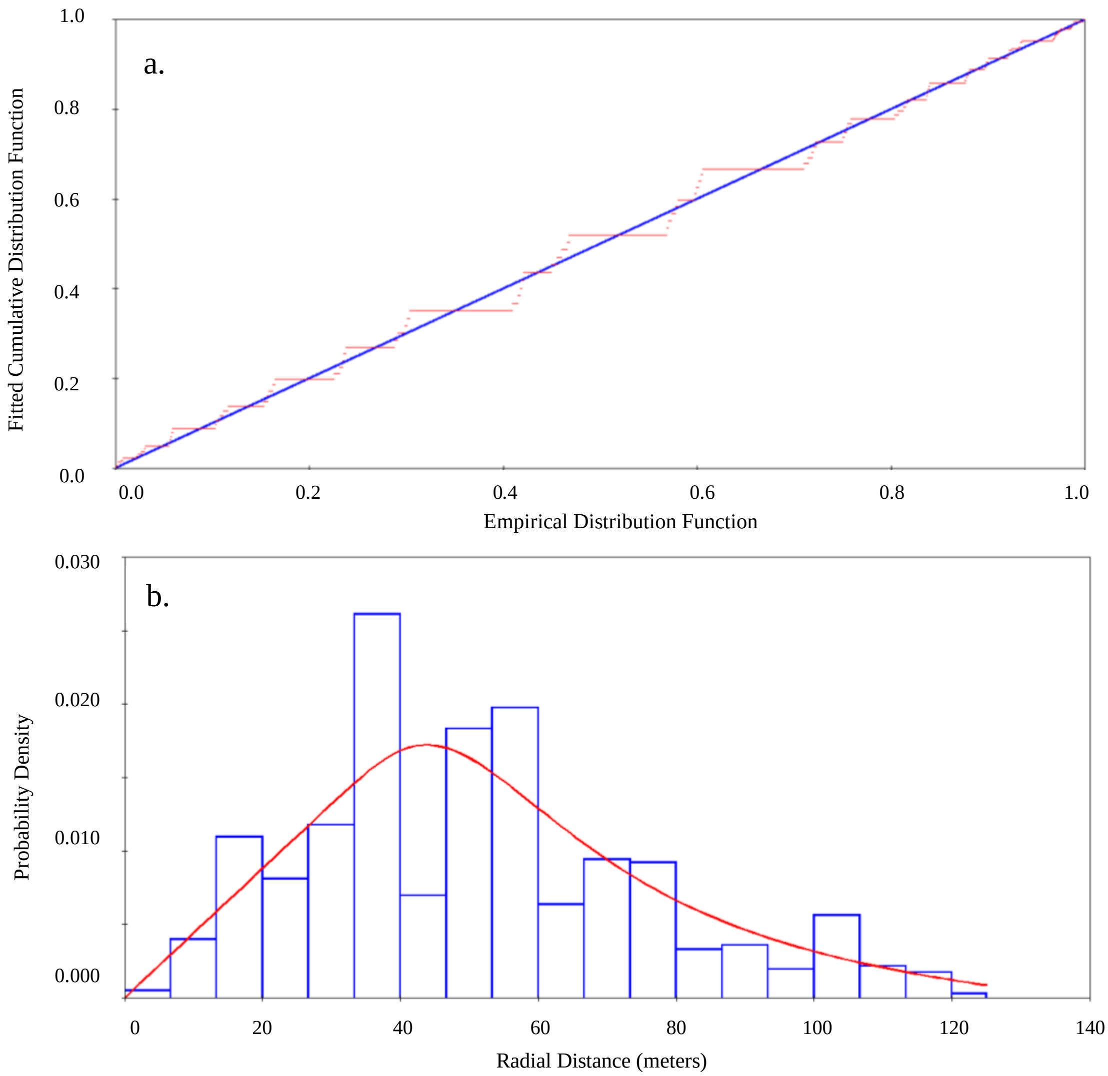


Figure 19. Brown-headed Cowbird Model. These charts depict data from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA. The first chart (a) depicts a Q-Q plot of the expected values in blue and the observed values in red. The second chart (b) depicts the transformed data in blue bins and the model as the red line.

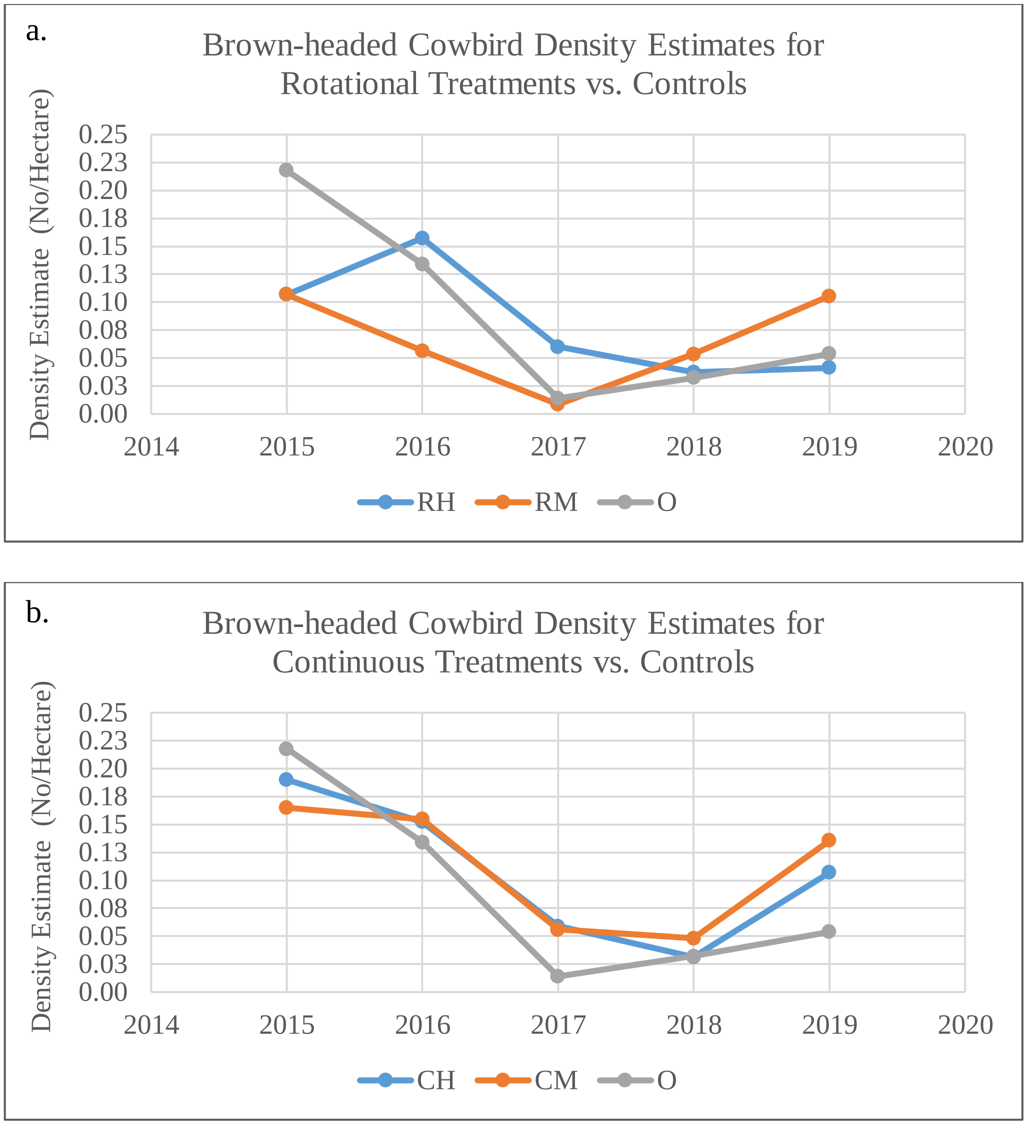


Figure 20. Brown-headed Cowbird - Rotational vs. Continuous. These figures depict 2 charts representing Brown-headed Cowbird density on rotational treatments compared to the references (a), and continuous treatments compared to the references (b). These data are from point count surveys from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA (CH: continuous high; CM: continuous moderate; RH: rotational high; RM: rotational moderate; O: off the CGRDA pastures, i.e. control).

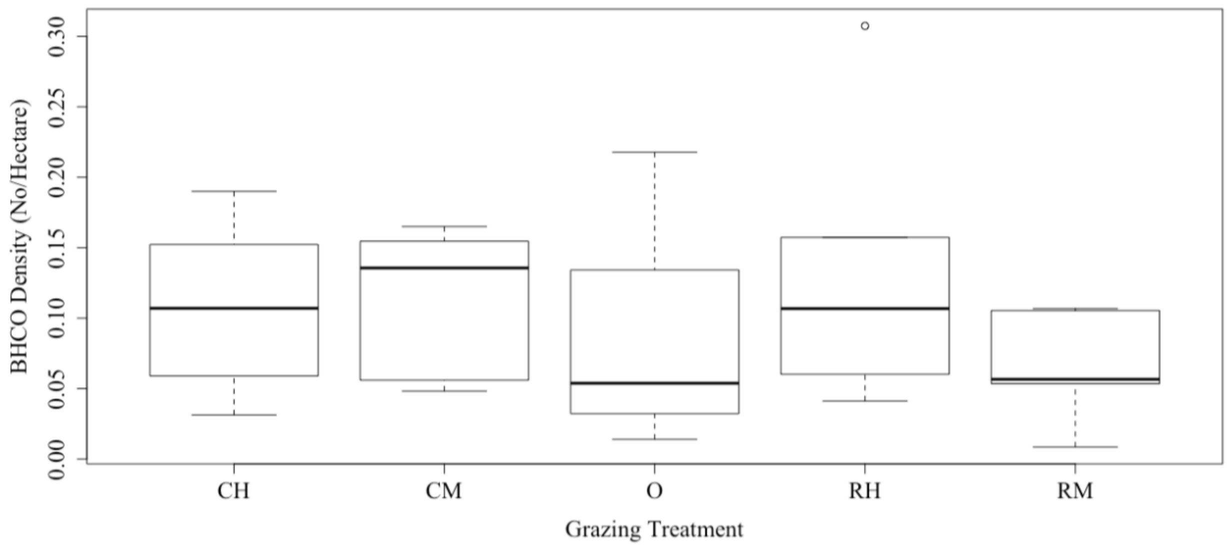


Figure 21. Brown-headed Cowbird Box Plot. This box plot depicts the density estimates per treatment with interquartile range density estimates in boxes (25th to 75th percentile), bold horizonal lines as the median values, thin horizontal lines as the maximum and minimum values, and outliers as dots from data collected from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA (CH: continuous high; CM: continuous moderate; RH: rotational high; RM: rotational moderate; O: off the CGRDA pastures, i.e. control).

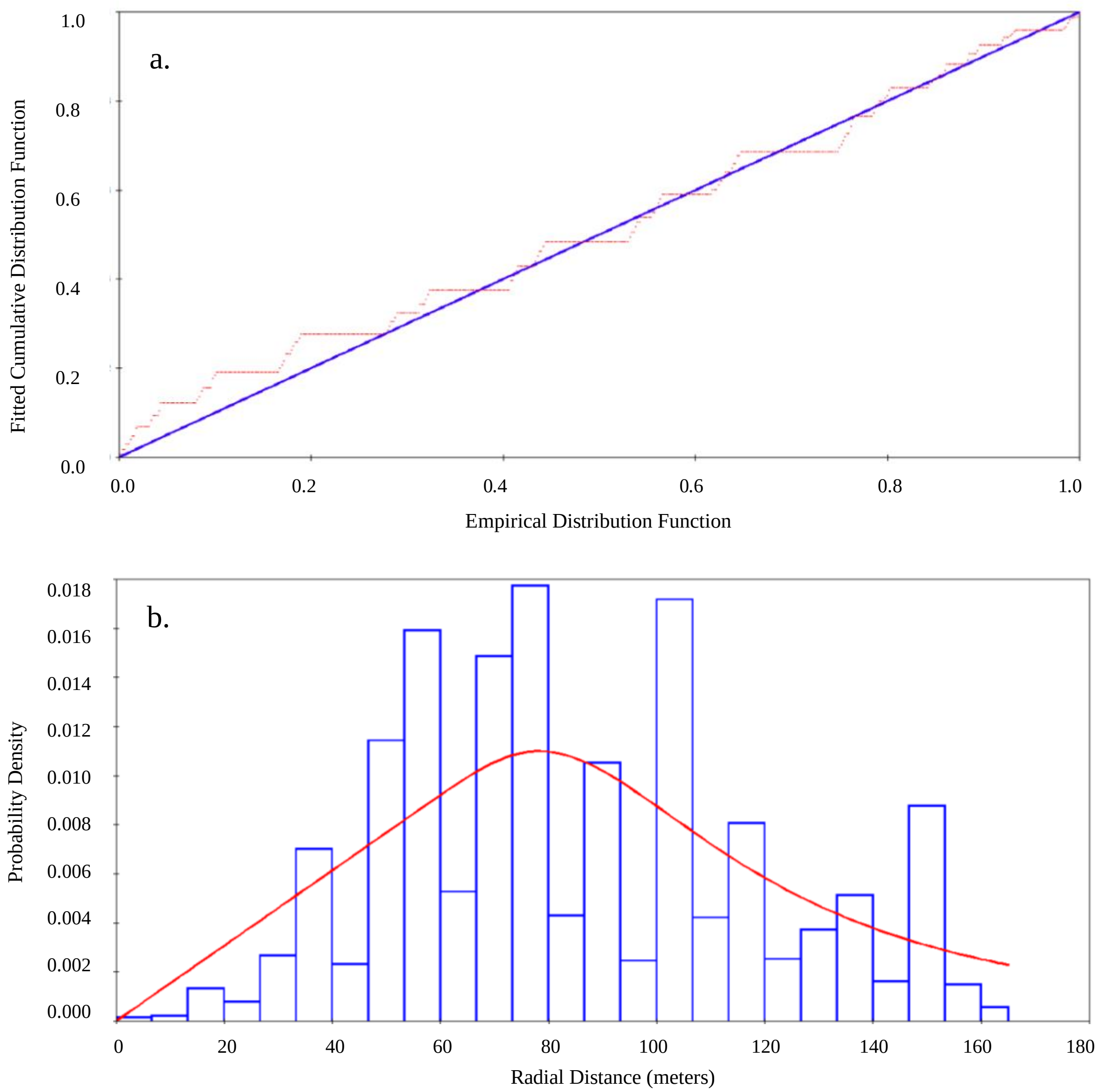


Figure 22. Eastern Meadowlark Model. These charts depict data from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA. The first chart (a) depicts a Q-Q plot of the expected values in blue and the observed values in red. The second chart (b) depicts the transformed data in blue bins and the model as the red line (CH: continuous high; CM: continuous moderate; RH: rotational high; RM: rotational moderate; O: off the CGRDA pastures, i.e. control).

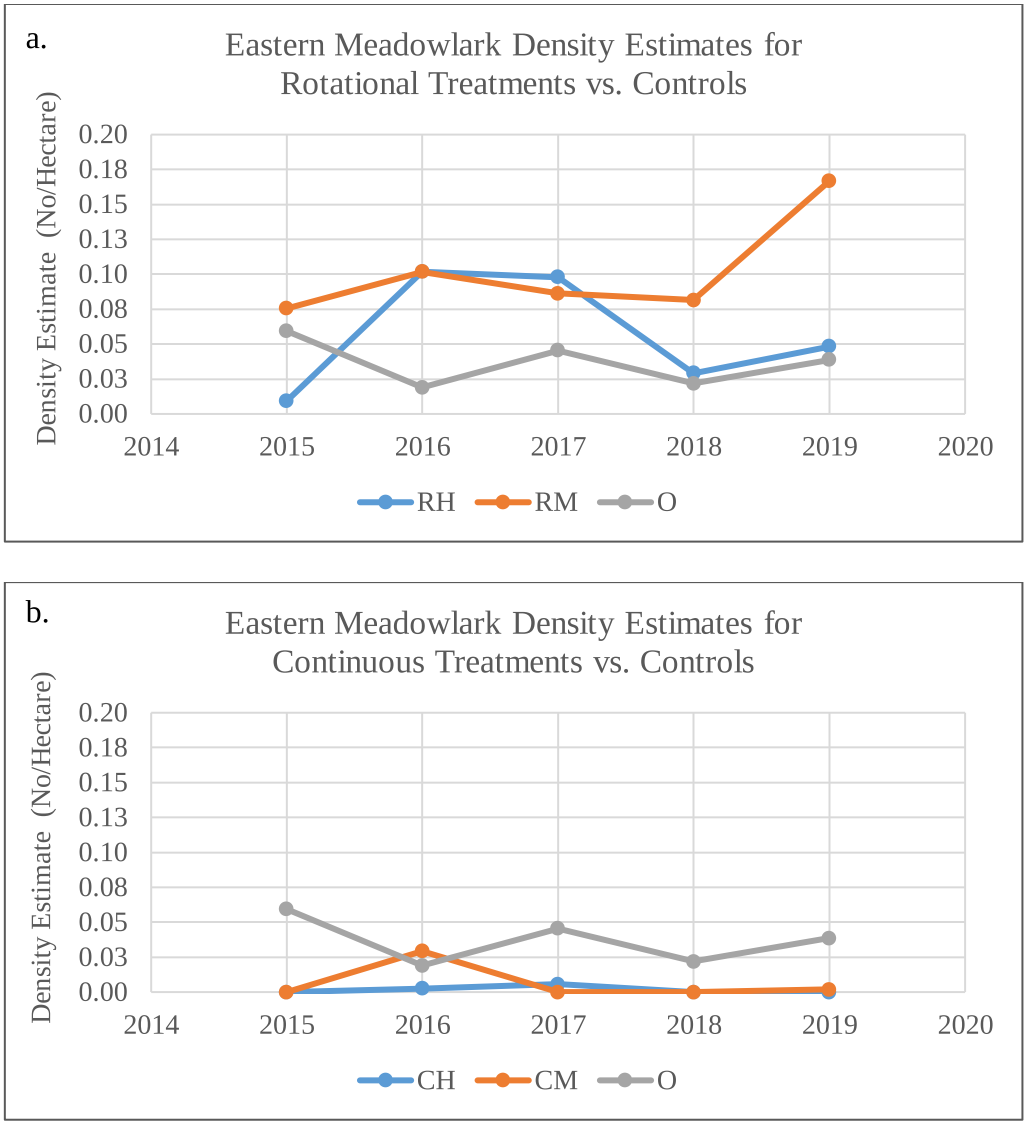


Figure 23. Eastern Meadowlark - Rotational vs. Continuous. These figures depict 2 charts representing Eastern Meadowlark density on rotational treatments compared to the references (a), and continuous treatments compared to the references (b). These data are from point count surveys from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA (CH: continuous high; CM: continuous moderate; RH: rotational high; RM: rotational moderate; O: off the CGRDA pastures, i.e. control).

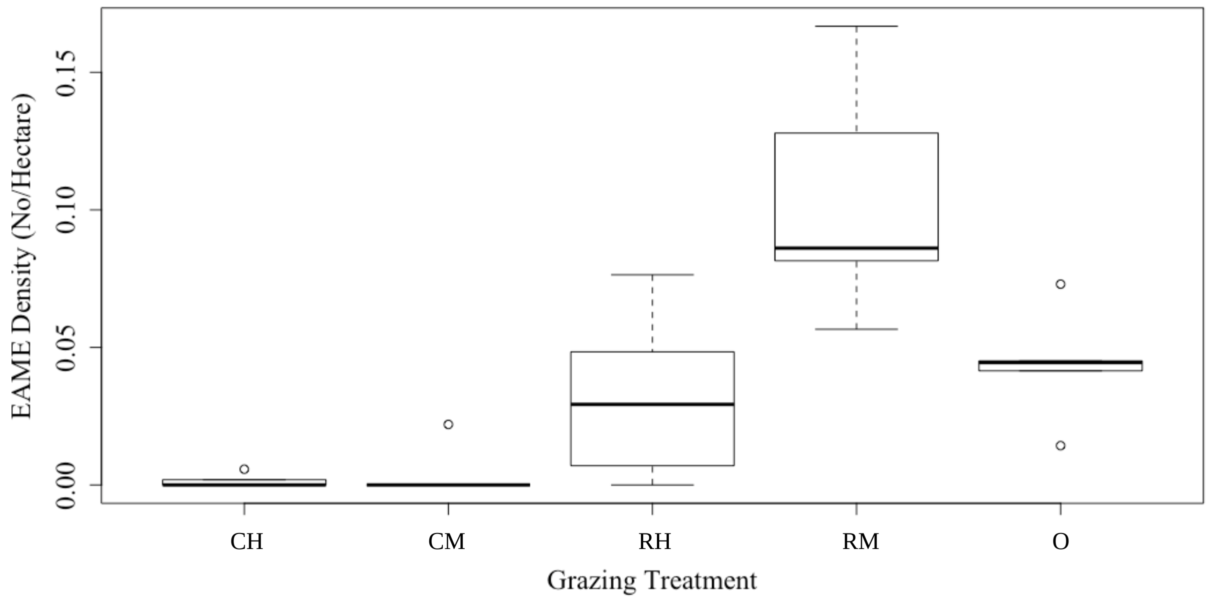


Figure 24. Eastern Meadowlark Box Plot. This box plot depicts the density estimates per treatment with interquartile range density estimates in boxes (25th to 75th percentile), bold horizonal lines as the median values, thin horizontal lines as the maximum and minimum values, and outliers as dots from data collected from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA (CH: continuous high; CM: continuous moderate; RH: rotational high; RM: rotational moderate; O: off the CGRDA pastures, i.e. control).

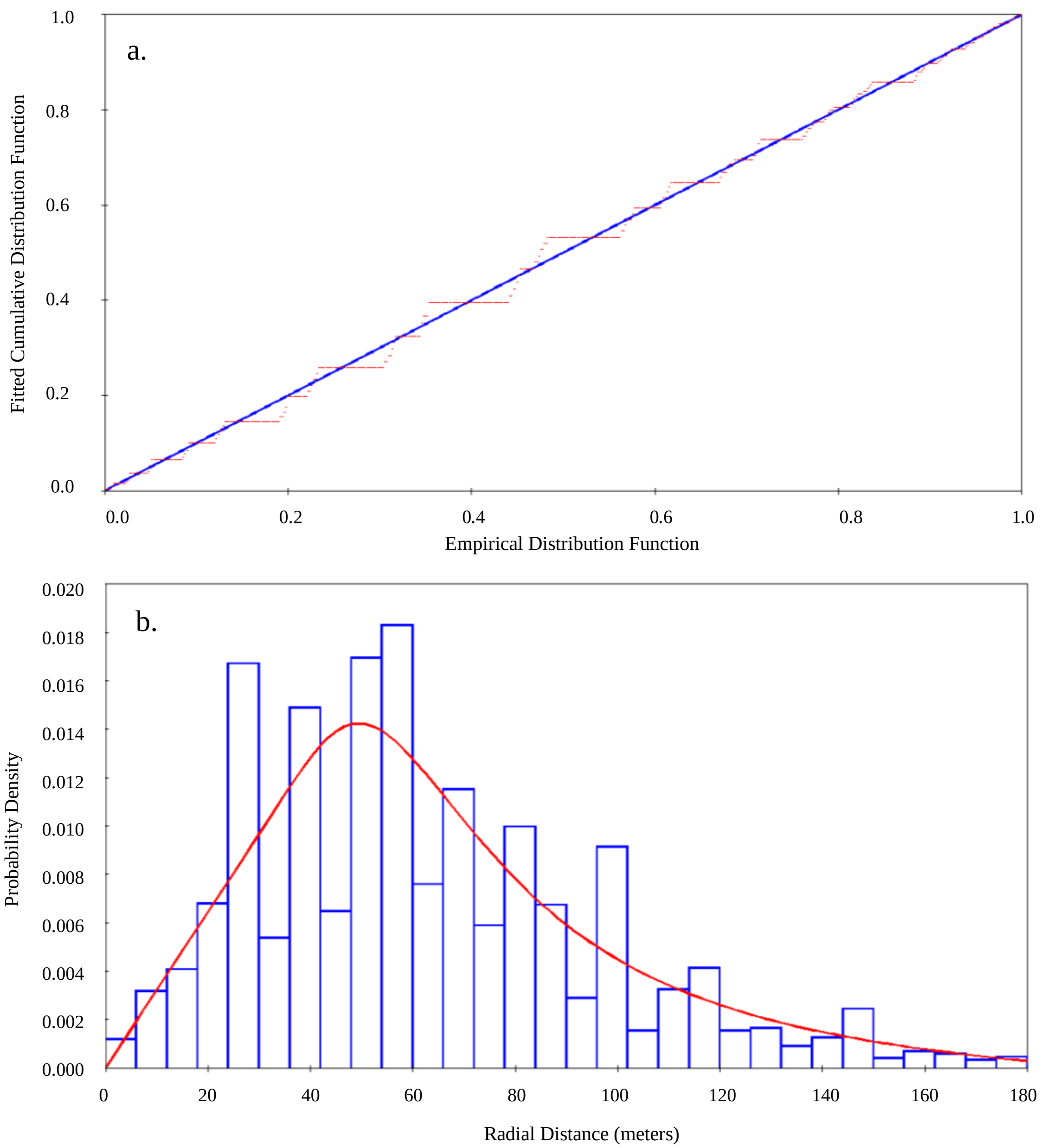


Figure 25. Scissor-tailed Flycatcher Model. These charts depict data from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA. The first chart (a) depicts a Q-Q plot of the expected values in blue and the observed values in red. The second chart (b) depicts the transformed data in blue bins and the model as the red line.

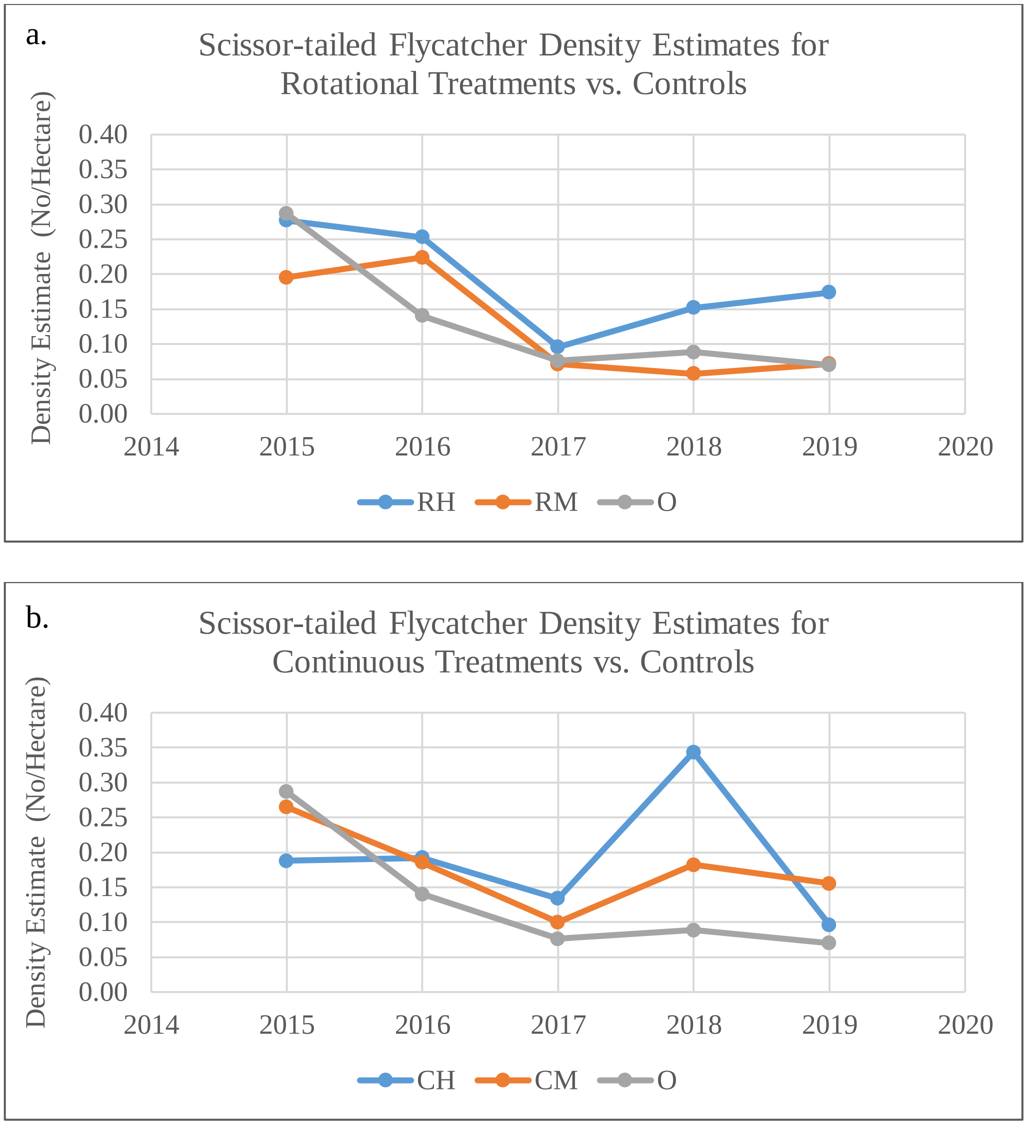


Figure 26. Scissor-tailed Flycatcher - Rotational vs. Continuous. These figures depict 2 charts representing Scissor-tailed Flycatcher density on rotational treatments compared to the references (a), and continuous treatments compared to the references (b). These data are from point count surveys from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA (CH: continuous high; CM: continuous moderate; RH: rotational high; RM: rotational moderate; O: off the CGRDA pastures, i.e. control).

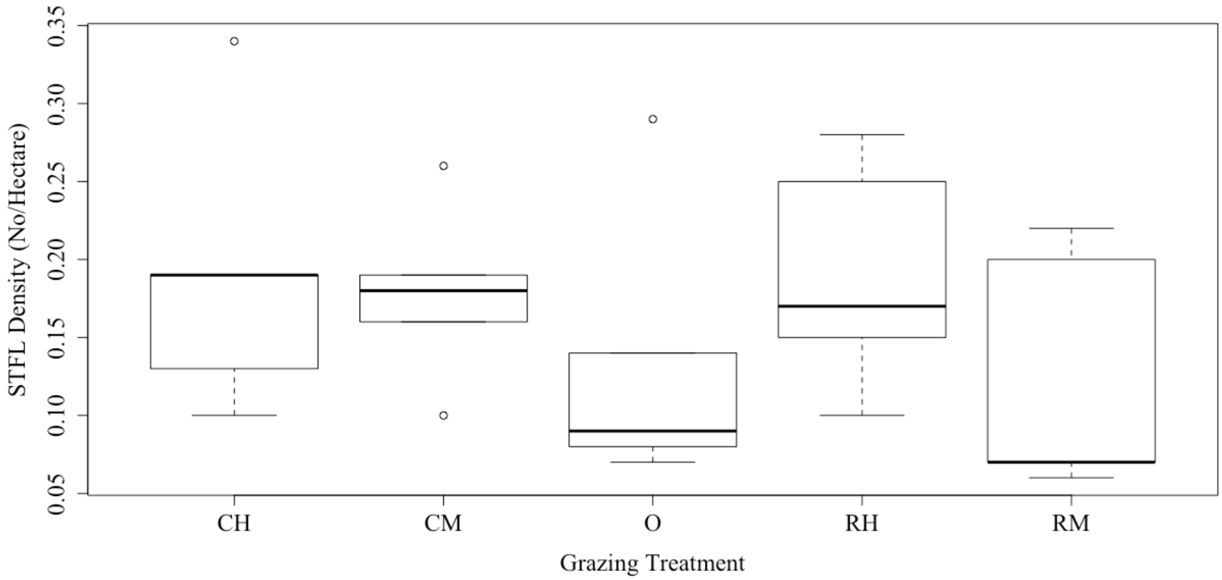


Figure 27. Scissor-tailed Flycatcher Box Plot. This box plot depicts the density estimates per treatment with interquartile range density estimates in boxes (25th to 75th percentile), bold horizonal lines as the median values, thin horizontal lines as the maximum and minimum values, and outliers as dots from data collected from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA (CH: continuous high; CM: continuous moderate; RH: rotational high; RM: rotational moderate; O: off the CGRDA pastures, i.e. control).

APPENDIX B: TABLES

Table 1. **Observations.** This table depicts the number of observations (detections) within the data for each species and each year. Data were collected from 2015-2019 on the SAV ranch in Jim Hogg and Starr Counties, Texas, USA.

	2015	**2016**	**2017**	**2018**	**2019**	**Total**
Northern Bobwhite	6104	4519	4161	2366	2756	19906
Scissor-tailed Flycatcher	1049	720	448	431	322	2970
Dickcissel	1481	3	23	0	5	1512
Grasshopper Sparrow	54	0	129	1	1	185
Brown-headed Cowbird	699	398	145	119	146	1507
Eastern Meadowlark	572	486	623	209	342	2232

Table 2. **Changes.** This table displays the point count survey changes that were made each year from 2015 to 2019 on the SAV ranch.

	2015	**2016**	**2017**	**2018**	**2019**
# Observers per Point	2	2	2	1	1
# Points per Transect	12	12	16	16	16
# Transects in Treatment	10	10	10	10	10
# Transects in Reference	20	15	5	15	15
# Points in Treatment	120	120	160	160	160
# Points in Reference	240	180	80	240	240
# Total Points	360	300	240	400	400
# Visits	4	4	6	4	4
Survey Effort	8	8	12	4	4

Table 3. Richness Pairwise Comparison. This table depicts *P*-values of transformed richness values of the model grassland bird assemblage from a Wilcoxon signed-rank test. There was a statistically significant difference between the continuous high treatment and reference, continuous moderate treatment and rotational moderate treatment, as well as the continuous moderate treatment and reference (CH: continuous high; CM: continuous moderate; RH: rotational high; RM: rotational moderate; O: off the CGRDA pastures, i.e. control).

	CH	CM	RH	RM
CM	1.000			
RH	0.083	0.083		
RM	0.102	**0.046**	0.317	
O	**0.046**	**0.046**	0.317	1.000

Table 4. **Northern Bobwhite Detection Function Models.** This table depicts detection function models that I used to assess the Northern Bobwhite data set and their resulting ΔAIC values and goodness-of-fit test results from Kolmogorov-Smirnov (K-S), Cramer VonMises (CvM unif), and Cramer VonMises (CvM cos) tests.

Detection Function	**ΔAIC**	**K-S**	**CvM unif**	**CvM cos**
CDS Stratified	0.00	0.000-0.288	0.001-0.150	0.001-0.075
MCDS Stratified	86.50	0.000-0.9337	0.001-0.950	0.001-0.850
CDS Global	1974.50	0.000	0.001	0.001
MCDS Global	1974.91	0.000	0.001	0.001

Table 5. Northern Bobwhite Density. Spatial and temporal ID (species 4 letter code, 2 digit year code, 1 digit treatment code [1:CH, 2:CM, 3:RH, 4:RM, 5:O]), number of points (k), number of detections (n), detection probability ($\hat{p}$), density ± standard error (D±SE, birds/ha), coefficient of variation of density estimate (%CV($\hat{D}$)), 95% confidence intervals of density estimate (95% CI [$\hat{D}$]), and test results from 3 goodness-of-fit (GOF) tests: Kolmogorov–Smirnov results (K-S), Cramer VonMises (CvM unif), and Cramer VonMises (CvM cos) from point count survey data from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA.

				$\hat{D}$					
ID	**k**	**n**	**$\hat{p}$**	**D (±SE)**	**%CV($\hat{D}$)**	**95% CI**	**GOF K-S**	**GOF CvM unif**	**GOF CvM cos**
NOBO151	24	547	0.41	0.810 ± 0.051	6.24	(0.098 - 1.731)	0.000	0.001	0.001
NOBO152	24	531	0.41	0.786 ± 0.055	7.01	(0.106 - 1.694)	0.000	0.001	0.001
NOBO153	36	642	0.41	0.634 ± 0.044	6.93	(0.083 - 1.363)	0.000	0.001	0.001
NOBO154	36	758	0.41	0.748 ± 0.035	4.65	(0.067 - 1.570)	0.000	0.001	0.001
NOBO155	240	3590	0.41	0.531 ± 0.021	3.99	(0.040 - 1.106)	0.000	0.001	0.001
NOBO161	24	365	0.41	0.540 ± 0.040	7.48	(0.077 - 1.171)	0.000	0.001	0.001
NOBO162	24	414	0.41	0.613 ± 0.044	7.12	(0.084 - 1.323)	0.000	0.001	0.001
NOBO163	36	571	0.41	0.564 ± 0.030	5.35	(0.058 - 1.192)	0.000	0.001	0.001
NOBO164	36	604	0.41	0.596 ± 0.030	5.00	(0.057 - 1.256)	0.000	0.001	0.001
NOBO165	180	2419	0.41	0.477 ± 0.018	3.70	(0.034 - 0.991)	0.000	0.001	0.001
NOBO171	32	432	0.41	0.320 ± 0.019	6.00	(0.037 - 0.681)	0.000	0.001	0.001
NOBO172	32	510	0.41	0.377 ± 0.034	9.14	(0.064 - 0.832)	0.000	0.001	0.001
NOBO173	48	912	0.41	0.450 ± 0.026	5.75	(0.049 - 0.955)	0.000	0.001	0.001
NOBO174	48	962	0.41	0.475 ± 0.027	5.69	(0.051 - 1.007)	0.000	0.001	0.001
NOBO175	80	1304	0.41	0.386 ± 0.019	4.91	(0.036 - 0.812)	0.000	0.001	0.001
NOBO181	32	142	0.41	0.315 ± 0.042	13.34	(0.075 - 0.729)	0.000	0.001	0.001
NOBO182	32	167	0.41	0.371 ± 0.037	9.99	(0.068 - 0.825)	0.000	0.001	0.001
NOBO183	48	242	0.41	0.358 ± 0.039	10.86	(0.070 - 0.804)	0.000	0.001	0.001
NOBO184	48	359	0.41	0.531 ± 0.046	8.65	(0.085 - 1.164)	0.000	0.001	0.001
NOBO185	240	1118	0.41	0.331 ± 0.018	5.43	(0.034 - 0.699)	0.000	0.001	0.001
NOBO191	32	159	0.41	0.353 ± 0.025	7.20	(0.048 - 0.762)	0.000	0.001	0.001
NOBO192	32	199	0.41	0.442 ± 0.033	7.55	(0.063 - 0.957)	0.000	0.001	0.001
NOBO193	48	311	0.41	0.460 ± 0.032	6.95	(0.060 - 0.990)	0.000	0.001	0.001
NOBO194	48	454	0.41	0.672 ± 0.040	5.88	(0.075 - 1.428)	0.000	0.001	0.001
NOBO195	240	1404	0.41	0.416 ± 0.018	4.39	(0.034 - 0.869)	0.000	0.001	0.001

Table 6. Northern Bobwhite Treatment ANOVA. This table depicts results of a single factor ANOVA test comparing treatments of the Northern Bobwhite density data that was collected from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA.

SUMMARY

Groups	*Count*	*Sum*	*Average*	*Variance*
CH	5	2.338	0.468	0.045
CM	5	2.589	0.518	0.032
RH	5	2.466	0.493	0.011
RM	5	3.022	0.604	0.012
O	5	2.142	0.428	0.006

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Between Groups	0.0872	4	0.0218	1.0226	0.4196	2.8661
Within Groups	0.4263	20	0.0213			
Total	0.5135	24				

Table 7. Northern Bobwhite Years ANOVA. This table depicts results of a single factor ANOVA test comparing years of the Northern Bobwhite density data that was collected from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA.

SUMMARY

Groups	*Count*	*Sum*	*Average*	*Variance*
2015	5	3.509	0.702	0.014
2016	5	2.790	0.558	0.003
2017	5	2.008	0.402	0.004
2018	5	1.907	0.381	0.008
2019	5	2.343	0.469	0.015

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Between Groups	0.3440	4	0.0860	10.1491	0.0001	2.8661
Within Groups	0.1695	20	0.0085			
Total	0.5135	24				

Table 8. Northern Bobwhite ANOVA Table. This table depicts an analysis of variance table, including degrees of freedom (DF), sum of squares (Sum Sq), mean squares (MS), F value, and *P*-value from density estimates of the Northern Bobwhite from data collected from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA.

	DF	Sum Sq	MS	F Value	Pr(>F)
years	1	0.2067	0.2067	19.2960	0.0005
treatments	4	0.0872	0.0218	2.0340	0.1410
years:treatments	4	0.0588	0.0147	1.3730	0.2900
residuals	15	0.1607	0.0107		

Table 9. **Grasshopper Sparrow Detection Function Models.** This table depicts the detection function models that I used to assess the Grasshopper Sparrow data set and their resulting ΔAIC values and goodness-of-fit test results from Kolmogorov-Smirnov (K-S), Cramer VonMises (CvM unif), and Cramer VonMises (CvM cos) tests.

Detection Function	**ΔAIC**	**K-S**	**CvM unif**	**CvM cos**
MCDS Stratified	0.00	0.000-0.599	0.000-0.750	0.000-0.650
CDS Stratified	0.09	0.000-0.527	0.000-0.650	0.000-0.550
MCDS Global	13.38	0.000	0.005	0.005
CDS Global	15.45	0.000	0.005	0.005

Table 10. Grasshopper Sparrow Density. Spatial and temporal ID (species 4 letter code, 2 digit year code, 1 digit treatment code [1:CH, 2:CM, 3:RH, 4:RM, 5:O]), number of points (k), number of detections (n), detection probability ($\hat{p}$), density ± standard error (D±SE, birds/ha), coefficient of variation of density estimate (%CV($\hat{D}$)), 95% confidence intervals of density estimate (95% CI [$\hat{D}$]), and test results from 3 goodness-of-fit (GOF) tests: Kolmogorov–Smirnov results (K-S), Cramer VonMises (CvM unif), and Cramer VonMises (CvM cos) from point count survey data from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA.

				$\hat{D}$					
ID	**k**	**n**	**$\hat{p}$**	**D (±SE)**	**%CV($\hat{D}$)**	**95% CI**	**GOF K-S**	**GOF CvM unif**	**GOF CvM cos**
GRSP151	24	0	0.54	0.000	0.00	(0.000 - 0.000)	0.000	0.005	0.005
GRSP152	24	0	0.54	0.000	0.00	(0.000 - 0.000)	0.000	0.005	0.005
GRSP153	36	54	0.54	0.137 ± 0.107	78.44	(0.103 - 0.694)	0.000	0.005	0.005
GRSP154	36	0	0.54	0.000	0.00	(0.000 - 0.000)	0.000	0.005	0.005
GRSP155	240	0	0.54	0.000	0.00	(0.000 - 0.000)	0.000	0.005	0.005
GRSP161	24	0	0.54	0.000	0.00	(0.000 - 0.000)	0.000	0.005	0.005
GRSP162	24	0	0.54	0.000	0.00	(0.000 - 0.000)	0.000	0.005	0.005
GRSP163	36	0	0.54	0.000	0.00	(0.000 - 0.000)	0.000	0.005	0.005
GRSP164	36	0	0.54	0.000	0.00	(0.000 - 0.000)	0.000	0.005	0.005
GRSP165	180	0	0.54	0.000	0.00	(0.000 - 0.000)	0.000	0.005	0.005
GRSP171	32	4	0.54	0.008 ± 0.006	78.59	(0.006 - 0.039)	0.000	0.005	0.005
GRSP172	32	15	0.54	0.028 ± 0.011	38.91	(0.015 - 0.090)	0.000	0.005	0.005
GRSP173	48	31	0.54	0.039 ± 0.011	27.16	(0.016 - 0.106)	0.000	0.005	0.005
GRSP174	48	73	0.54	0.090 ± 0.018	19.58	(0.029 - 0.222)	0.000	0.005	0.005
GRSP175	80	6	0.54	0.005 ± 0.003	62.12	(0.003 - 0.019)	0.000	0.005	0.005
GRSP181	32	0	0.54	0.000	0.00	(0.000 - 0.000)	0.000	0.005	0.005
GRSP182	32	0	0.54	0.000	0.00	(0.000 - 0.000)	0.000	0.005	0.005
GRSP183	48	0	0.54	0.000	0.00	(0.000 - 0.000)	0.000	0.005	0.005
GRSP184	48	0	0.54	0.000	0.00	(0.000 - 0.000)	0.000	0.005	0.005
GRSP185	240	1	0.54	0.001 ± 0.001	100.23	(0.001 - 0.005)	0.000	0.005	0.005
GRSP191	32	0	0.54	0.000	0.00	(0.000 - 0.000)	0.000	0.005	0.005
GRSP192	32	0	0.54	0.000	0.00	(0.000 - 0.000)	0.000	0.005	0.005
GRSP193	48	0	0.54	0.000	0.00	(0.000 - 0.000)	0.000	0.005	0.005
GRSP194	48	0	0.54	0.000	0.00	(0.000 - 0.000)	0.000	0.005	0.005
GRSP195	240	0	0.54	0.000	0.00	(0.000 - 0.000)	0.000	0.005	0.005

Table 11. Grasshopper Sparrow Treatment Pairwise Comparison. This table depicts the *P*-values for the density estimates of the model species from a Wilcoxon signed-rank test from data collected from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA. There are no statistically significant differences in the data between any of the treatment sites (CH: continuous high; CM: continuous moderate; RH: rotational high; RM: rotational moderate; O: off the CGRDA pastures, i.e. control).

	CH	CM	RH	RM
CM	0.32			
RH	0.65	0.65		
RM	0.18	0.18	0.29	
O	0.32	0.32	0.65	0.65

Table 12. Grasshopper Sparrow Year Pairwise Comparison. This table depicts the *P*-values for the density estimates of the model species from a Wilcoxon signed-rank test from data collected from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA. There are statistically significant differences in the mean density estimates between 2016 and 2017, 2017 and 2018, and 2017 and 2019.

	2015	2016	2017	2018
2016	0.317			
2017	0.500	**0.043**		
2018	0.655	0.317	**0.043**	
2019	0.317		**0.043**	0.317

Table 13. **Dickcissel Detection Function Models.** This table depicts the detection function models that I used to assess the Dickcissel data set and their resulting ΔAIC values and goodness-of-fit test results from a Kolmogorov-Smirnov (K-S), Cramer VonMises (CvM unif), and Cramer VonMises (CvM cos) test.

Detection Function	**ΔAIC**	**K-S**	**CvM unif**	**CvM cos**
CDS Stratified	0.00	0.099-0.969	0.250-0.850	0.125-0.850
MCDS Stratified	3.19	0.005-0.969	0.075-0.850	0.075-0.850
CDS Global	89.01	0.000	0.025	0.050
MCDS Global	97.23	0.001	0.025	0.050

Table 14. Dickcissel Density. Spatial and temporal ID (species 4 letter code, 2 digit year code, 1 digit treatment code [1:CH, 2:CM, 3:RH, 4:RM, 5:O]), number of points (k), number of detections (n), detection probability ($\hat{p}$), density ± standard error (D±SE, birds/ha), coefficient of variation of density estimate (%CV($\hat{D}$)), 95% confidence intervals of density estimate (95% CI [$\hat{D}$]), and test results from 3 goodness-of-fit (GOF) tests: Kolmogorov–Smirnov results (K-S), Cramer VonMises (CvM unif), and Cramer VonMises (CvM cos) from point count survey data from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA.

				$\hat{D}$					
ID	**k**	**n**	**$\hat{p}$**	**D (±SE)**	**%CV($\hat{D}$)**	**95% CI**	**GOF K-S**	**GOF CvM unif**	**GOF CvM cos**
DICK151	24	208	0.37	0.601 ± 0.155	25.79	(0.245 - 1.616)	0.000	0.025	0.050
DICK152	24	84	0.37	0.243 ± 0.062	25.36	(0.098 - 0.649)	0.000	0.025	0.050
DICK153	36	226	0.37	0.435 ± 0.077	17.70	(0.130 - 1.057)	0.000	0.025	0.050
DICK154	36	442	0.37	0.851 ± 0.120	14.08	(0.210 - 1.982)	0.000	0.025	0.050
DICK155	240	452	0.37	0.131 ± 0.024	18.47	(0.040 - 0.318)	0.000	0.025	0.050
DICK161	24	0	0.37	0.000			0.000	0.025	0.050
DICK162	24	0	0.37	0.000			0.000	0.025	0.050
DICK163	36	0	0.37	0.000			0.000	0.025	0.050
DICK164	36	2	0.37	0.004 ± 0.003	69.77	(0.003 - 0.018)	0.000	0.025	0.050
DICK165	180	1	0.37	0.000 ± 0.000	100.05	(0.000 - 0.002)	0.000	0.025	0.050
DICK171	32	3	0.37	0.004 ± 0.003	73.64	(0.003 - 0.021)	0.000	0.025	0.050
DICK172	32	1	0.37	0.001 ± 0.001	100.05	(0.001 - 0.009)	0.000	0.025	0.050
DICK173	48	5	0.37	0.005 ± 0.003	65.51	(0.003 - 0.021)	0.000	0.025	0.050
DICK174	48	4	0.37	0.004 ± 0.002	48.48	(0.002 - 0.014)	0.000	0.025	0.050
DICK175	80	10	0.37	0.006 ± 0.002	36.08	(0.003 - 0.017)	0.000	0.025	0.050
DICK181	32	0	0.37	0.000			0.000	0.025	0.050
DICK182	32	0	0.37	0.000			0.000	0.025	0.050
DICK183	48	0	0.37	0.000			0.000	0.025	0.050
DICK184	48	0	0.37	0.000			0.000	0.025	0.050
DICK185	240	0	0.37	0.000			0.000	0.025	0.050
DICK191	32	0	0.37	0.000			0.000	0.025	0.050
DICK192	32	0	0.37	0.000			0.000	0.025	0.050
DICK193	48	4	0.37	0.012 ± 0.006	48.48	(0.007 - 0.041)	0.000	0.025	0.050
DICK194	48	0	0.37	0.000			0.000	0.025	0.050
DICK195	240	1	0.37	0.001 ± 0.001	100.05	(0.000 - 0.004)	0.000	0.025	0.050

Table 15. Dickcissel Treatment Pairwise Comparison. This table depicts the *P*-values for the density estimates of the model species from a Wilcoxon signed-rank test from data collected from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA. There are no statistically significant differences in the data between any of the treatment sites (CH: continuous high; CM: continuous moderate; RH: rotational high; RM: rotational moderate; O: off the CGRDA pastures, i.e. control).

	CH	CM	RH	RM
CM	0.18			
RH	0.72	0.72		
RM	1.00	0.11	0.47	
O	0.29	0.11	0.47	0.72

Table 16. Dickcissel Year Pairwise Comparison. This table depicts the *P*-values for the density estimates of the model species from a Wilcoxon signed-rank test from data collected from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA. There were statistically significant differences in the mean density estimates between 2015 and all other years, as well as between 2017 and 2018.

	2015	2016	2017	2018
2016	**0.043**			
2017	**0.043**	0.068		
2018	**0.043**	0.180	**0.043**	
2019	**0.043**	0.593	0.500	0.180

Table 17. **Brown-headed Cowbird Detection Function Models.** This table depicts the detection function models that I used to assess the Brown-headed Cowbird data set and their resulting ΔAIC values and goodness-of-fit test results from Kolmogorov-Smirnov (K-S), Cramer VonMises (CvM unif), and Cramer VonMises (CvM cos) tests.

Detection Function	**ΔAIC**	**K-S**	**CvM unif**	**CvM cos**
CDS Stratified	0.00	0.001-0.998	0.007-0.950	0.003-0.950
MCDS Stratified	6.98	0.001-0.667	0.001-0.550	0.001-0.650
CDS Global	38.26	0.000	0.025	0.025
MCDS Global	40.76	0.000	0.025	0.025

Table 18. Brown-headed Cowbird Density. Spatial and temporal ID (species 4 letter code, 2 digit year code, 1 digit treatment code [1:CH, 2:CM, 3:RH, 4:RM, 5:O]), number of points (k), number of detections (n), detection probability ($\hat{p}$), density ± standard error (D±SE, birds/ha), coefficient of variation of density estimate (%CV($\hat{D}$)), 95% confidence intervals of density estimate (95% CI [$\hat{D}$]), and test results from 3 goodness-of-fit (GOF) tests: Kolmogorov–Smirnov results (K-S), Cramer VonMises (CvM unif), and Cramer VonMises (CvM cos) from point count survey data from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA.

				$\hat{D}$					
ID	**k**	**n**	**$\hat{p}$**	**D (±SE)**	**%CV($\hat{D}$)**	**95% CI**	**GOF K-S**	**GOF CvM (unif)**	**GOF CvM (cos)**
BHCO151	24	59	0.33	0.190 ± 0.062	32.40	(0.089 - 0.547)	0.508	0.550	0.450
BHCO152	24	47	0.30	0.165 ± 0.031	19.00	(0.053 - 0.407)	0.654	0.650	0.550
BHCO153	36	47	0.40	0.107 ± 0.022	20.68	(0.036 - 0.268)	0.454	0.450	0.350
BHCO154	36	60	0.39	0.107 ± 0.029	27.46	(0.044 - 0.289)	0.208	0.450	0.350
BHCO155	240	471	0.23	0.218 ± 0.025	11.54	(0.044 - 0.491)	0.069	0.250	0.250
BHCO161	24	35	0.24	0.152 ± 0.061	39.84	(0.082 - 0.481)	0.720	0.650	0.650
BHCO162	24	28	0.19	0.155 ± 0.059	38.17	(0.081 - 0.480)	0.441	0.550	0.450
BHCO163	36	59	0.27	0.157 ± 0.040	25.15	(0.061 - 0.415)	0.307	0.550	0.450
BHCO164	36	26	0.32	0.057 ± 0.018	30.97	(0.026 - 0.160)	0.644	0.550	0.450
BHCO165	180	232	0.24	0.134 ± 0.019	14.05	(0.032 - 0.311)	0.138	0.250	0.250
BHCO171	32	34	0.31	0.059 ± 0.020	33.56	(0.028 - 0.173)	0.278	0.350	0.350
BHCO172	32	32	0.30	0.056 ± 0.022	38.64	(0.030 - 0.175)	0.001	0.007	0.003
BHCO173	48	47	0.28	0.060 ± 0.015	24.55	(0.023 - 0.158)	0.343	0.450	0.450
BHCO174	48	8	0.33	0.009 ± 0.009	103.83	(0.007 - 0.068)	0.011	0.017	0.075
BHCO175	80	23	0.35	0.014 ± 0.005	32.29	(0.007 - 0.040)	0.023	0.075	0.075
BHCO181	32	13	0.66	0.031 ± 0.013	42.98	(0.018 - 0.103)	0.545	0.650	0.550
BHCO182	32	9	0.30	0.048 ± 0.028	58.84	(0.033 - 0.199)	0.510	0.650	0.650
BHCO183	48	37	0.13	0.307 ± 0.270	87.91	(0.240 - 1.719)	0.816	0.850	0.750
BHCO184	48	15	0.30	0.054 ± 0.023	43.40	(0.030 - 0.176)	0.533	0.650	0.550
BHCO185	240	39	0.26	0.032 ± 0.008	25.07	(0.012 - 0.085)	0.597	0.550	0.550
BHCO191	32	20	0.30	0.107 ± 0.032	29.67	(0.047 - 0.299)	0.747	0.750	0.650
BHCO192	32	16	0.19	0.136 ± 0.048	35.12	(0.067 - 0.405)	0.959	0.950	0.950
BHCO193	48	15	0.39	0.041 ± 0.020	48.83	(0.025 - 0.147)	0.742	0.850	0.750
BHCO194	48	21	0.21	0.105 ± 0.047	44.99	(0.061 - 0.357)	0.998	0.950	0.950
BHCO195	240	72	0.27	0.054 ± 0.010	18.07	(0.016 - 0.130)	0.894	0.950	0.950

Table 19. Brown-headed Cowbird Treatment ANOVA. This table depicts results of a single factor ANOVA test comparing treatments of the Brown-headed Cowbird data from data collected from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA. There were no significant differences.

SUMMARY

Groups	*Count*	*Sum*	*Average*	*Variance*
CH	5	0.540	0.108	0.004
CM	5	0.559	0.112	0.003
RH	5	0.673	0.135	0.011
RM	5	0.331	0.066	0.002
O	5	0.452	0.090	0.007

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Between Groups	0.0131	4	0.0033	0.5922	0.6723	2.8661
Within Groups	0.1102	20	0.0055			
Total	0.1233	24				

Table 20. Brown-headed Cowbird Years ANOVA. This table depicts results of a single factor ANOVA test comparing years of the Brown-headed Cowbird data from data collected from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA. There were no significant differences.

SUMMARY

Groups	*Count*	*Sum*	*Average*	*Variance*
2015	5	0.786	0.157	0.002
2016	5	0.655	0.131	0.002
2017	5	0.198	0.040	0.001
2018	5	0.473	0.095	0.014
2019	5	0.443	0.089	0.002

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Between Groups	0.0402	4	0.0100	2.4163	0.0826	2.8661
Within Groups	0.0831	20	0.0042			
Total	0.1233	24				

Table 21. Brown-headed Cowbird ANOVA Table. This table depicts an analysis of variance table, including degrees of freedom (DF), sum of squares (Sum Sq), mean squares (MS), F value, and *P*-value [Pr(>F)] from density estimates of the Brown-headed Cowbird from data collected from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA.

	DF	Sum Sq	MS	F Value	Pr(>F)
years	1	0.0015	0.0015	1.6240	0.2220
treatments	4	0.0042	0.0011	1.1500	0.3710
years:treatments	4	0.0060	0.0015	1.6350	0.2170
residuals	15	0.0137	0.0009		

Table 22. **Eastern Meadowlark Detection Function Models.** This table depicts the detection function models that I used to assess the Eastern Meadowlark data set and their resulting ΔAIC values and goodness-of-fit test results from Kolmogorov-Smirnov (K-S), Cramer VonMises (CvM unif), and Cramer VonMises (CvM cos) tests.

Detection Function	**ΔAIC**	**K-S**	**CvM unif**	**CvM cos**
CDS Stratified	0.00	0.001-0.942	0.075-0.850	0.125-0.850
MCDS Stratified	220.22	0.001-0.829	0.001-0.850	0.001-0.750
CDS Global	255.50	0.000	0.001	0.001
MCDS Global	358.56	0.000	0.001	0.001

Table 23. Eastern Meadowlark Density. Spatial and temporal ID (species 4 letter code, 2 digit year code, 1 digit treatment code [1:CH, 2:CM, 3:RH, 4:RM, 5:O]), number of points (k), number of detections (n), detection probability ($\hat{p}$), density ± standard error (D±SE, birds/ha), coefficient of variation of density estimate (%CV($\hat{D}$)), 95% confidence intervals of density estimate (95% CI [$\hat{D}$]), and test results from 3 goodness-of-fit (GOF) tests: Kolmogorov–Smirnov results (K-S), Cramer VonMises (CvM unif), and Cramer VonMises (CvM cos) from point count survey data from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA.

				$\hat{D}$					
ID	**k**	**n**	**$\hat{p}$**	**D (±SE)**	**%CV($\hat{D}$)**	**95% CI**	**GOF K-S**	**GOF CvM unif**	**GOF CvM cos**
EAME151	24	0	0.48	0.000	0.00	(0.000 - 0.000)	0.000	0.001	0.001
EAME152	24	0	0.48	0.000	0.00	(0.000 - 0.000)	0.000	0.001	0.001
EAME153	36	11	0.48	0.009 ± 0.004	42.97	(0.005 - 0.031)	0.000	0.001	0.001
EAME154	36	89	0.48	0.076 ± 0.021	27.21	(0.032 - 0.205)	0.000	0.001	0.001
EAME155	240	466	0.48	0.059 ± 0.010	17.32	(0.017 - 0.143)	0.000	0.001	0.001
EAME161	24	2	0.48	0.003 ± 0.002	69.21	(0.002 - 0.012)	0.000	0.001	0.001
EAME162	24	23	0.48	0.029 ± 0.015	49.72	(0.018 - 0.107)	0.000	0.001	0.001
EAME163	36	120	0.48	0.102 ± 0.021	20.35	(0.034 - 0.255)	0.000	0.001	0.001
EAME164	36	201	0.48	0.102 ± 0.021	20.35	(0.034 - 0.255)	0.000	0.001	0.001
EAME165	180	112	0.48	0.019 ± 0.005	25.21	(0.007 - 0.050)	0.000	0.001	0.001
EAME171	32	9	0.48	0.006 ± 0.003	53.58	(0.004 - 0.022)	0.000	0.001	0.001
EAME172	32	0	0.48	0.000	0.00	(0.000 - 0.000)	0.000	0.001	0.001
EAME173	48	231	0.48	0.098 ± 0.018	18.68	(0.030 - 0.240)	0.000	0.001	0.001
EAME174	48	203	0.48	0.086 ± 0.018	20.55	(0.029 - 0.216)	0.000	0.001	0.001
EAME175	80	177	0.48	0.045 ± 0.013	28.20	(0.019 - 0.124)	0.000	0.001	0.001
EAME181	32	0	0.48	0.000	0.00	(0.000 - 0.000)	0.000	0.001	0.001
EAME182	32	0	0.48	0.000	0.00	(0.000 - 0.000)	0.000	0.001	0.001
EAME183	48	23	0.48	0.029 ± 0.009	29.91	(0.013 - 0.082)	0.000	0.001	0.001
EAME184	48	64	0.48	0.081 ± 0.037	45.04	(0.047 - 0.275)	0.000	0.001	0.001
EAME185	240	86	0.48	0.022 ± 0.006	27.33	(0.009 - 0.059)	0.000	0.001	0.001
EAME191	32	0	0.48	0.000	0.00	(0.000 - 0.000)	0.000	0.001	0.001
EAME192	32	1	0.48	0.002 ± 0.002	100.03	(0.002 - 0.012)	0.000	0.001	0.001
EAME193	48	38	0.48	0.048 ± 0.012	25.35	(0.019 - 0.128)	0.000	0.001	0.001
EAME194	48	131	0.48	0.167 ± 0.034	20.37	(0.056 - 0.417)	0.000	0.001	0.001
EAME195	240	152	0.48	0.039 ± 0.008	20.89	(0.013 - 0.097)	0.000	0.001	0.001

Table 24. Eastern Meadowlark Treatment Pairwise Comparison. This table depicts the *P*-values for the density estimates of the model species from a Wilcoxon signed-rank test from data collected from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA. There were statistically significant differences in the means of the data between rotational moderate and all other treatments, as well as between continuous high and the control (CH: continuous high; CM: continuous moderate; RH: rotational high; RM: rotational moderate; O: off the CGRDA pastures, i.e. control).

	CH	CM	RH	RM
CM	0.655			
RH	0.080	0.068		
RM	**0.043**	**0.043**	**0.043**	
O	**0.043**	0.080	0.686	**0.043**

Table 25. Eastern Meadowlark Year Pairwise Comparison. This table depicts the *P*-values for the density estimates of the model species from a Wilcoxon signed-rank test from data collected from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA. There are no statistically significant differences in the mean density estimates between any years.

	2015	2016	2017	2018
2016	0.22			
2017	0.47	0.35		
2018	0.11	0.50	0.47	
2019	0.29	0.89	0.47	0.59

Table 26. **Scissor-tailed Flycatcher Detection Function Models.** This table depicts the detection function models that I used to assess the Scissor-tailed Flycatcher data set and their resulting ΔAIC values and goodness-of-fit test results from Kolmogorov-Smirnov (K-S), Cramer VonMises (CvM unif), and Cramer VonMises (CvM cos) tests.

Detection Function	**ΔAIC**	**K-S**	**CvM unif**	**CvM cos**
CDS Stratified	0.00	0.047-0.991	0.175-0.950	0.175-0.950
MCDS Stratified	5.23	0.004-0.978	0.075-0.950	0.075-0.950
CDS Global	174.48	0.000	0.010	0.005
MCDS Global	186.50	0.000	0.005	0.001

Table 27. Scissor-tailed Flycatcher Density. Spatial and temporal ID (species 4 letter code, 2 digit year code, 1 digit treatment code [1:CH, 2:CM, 3:RH, 4:RM, 5:O]), number of points (k), number of detections (n), detection probability ($\hat{p}$), density ± standard error (D±SE, birds/ha), coefficient of variation of density estimate (%CV($\hat{D}$)), 95% confidence intervals of density estimate (95% CI [$\hat{D}$]), and test results from 3 goodness-of-fit (GOF) tests: Kolmogorov–Smirnov results (K-S), Cramer VonMises (CvM unif), and Cramer VonMises (CvM cos) from point count survey data from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA.

				$\hat{D}$					
ID	**k**	**n**	**$\hat{p}$**	**D (±SE)**	**%CV($\hat{D}$)**	**95% CI**	**GOF K-S**	**GOF CvM unif**	**GOF CvM cos**
STFL151	24	45	0.12	63.100 ± 0.188	0.06	(0.092 - 0.554)	0.484	0.550	0.450
STFL152	24	75	0.14	68.500 ± 0.265	0.07	(0.110 - 0.717)	0.716	0.650	0.650
STFL153	36	119	0.15	69.000 ± 0.277	0.06	(0.094 - 0.695)	0.318	0.650	0.650
STFL154	36	78	0.14	66.500 ± 0.195	0.04	(0.063 - 0.483)	0.119	0.350	0.350
STFL155	240	731	0.13	65.000 ± 0.287	0.03	(0.049 - 0.632)	0.047	0.175	0.175
STFL161	24	48	0.13	64.000 ± 0.192	0.07	(0.100 - 0.594)	0.654	0.850	0.750
STFL162	24	61	0.17	73.800 ± 0.186	0.06	(0.087 - 0.534)	0.153	0.350	0.350
STFL163	36	110	0.15	69.000 ± 0.253	0.05	(0.087 - 0.640)	0.782	0.750	0.750
STFL164	36	114	0.17	75.000 ± 0.224	0.05	(0.073 - 0.557)	0.058	0.250	0.175
STFL165	180	379	0.18	77.300 ± 0.140	0.02	(0.034 - 0.325)	0.048	0.250	0.175
STFL171	32	73	0.14	67.000 ± 0.134	0.04	(0.061 - 0.379)	0.483	0.450	0.450
STFL172	32	63	0.16	72.200 ± 0.100	0.03	(0.046 - 0.287)	0.311	0.350	0.350
STFL173	48	100	0.18	75.900 ± 0.096	0.02	(0.030 - 0.236)	0.141	0.250	0.250
STFL174	48	76	0.18	76.000 ± 0.071	0.01	(0.023 - 0.176)	0.217	0.350	0.350
STFL175	80	134	0.18	76.000 ± 0.076	0.01	(0.024 - 0.188)	0.131	0.350	0.350
STFL181	32	46	0.10	57.700 ± 0.344	0.11	(0.157 - 0.975)	0.644	0.750	0.650
STFL182	32	43	0.18	76.000 ± 0.182	0.06	(0.088 - 0.536)	0.978	0.950	0.950
STFL183	48	63	0.21	83.000 ± 0.152	0.04	(0.057 - 0.394)	0.341	0.450	0.450
STFL184	48	34	0.20	99.200 ± 0.057	0.02	(0.032 - 0.185)	0.555	0.650	0.550
STFL185	240	235	0.27	93.000 ± 0.089	0.01	(0.022 - 0.206)	0.742	0.750	0.750
STFL191	32	20	0.16	71.900 ± 0.096	0.04	(0.051 - 0.303)	0.936	0.850	0.850
STFL192	32	38	0.19	77.900 ± 0.156	0.05	(0.076 - 0.458)	0.988	0.950	0.950
STFL193	48	65	0.19	78.000 ± 0.174	0.04	(0.066 - 0.454)	0.991	0.950	0.950
STFL194	48	44	0.31	100.800 ± 0.072	0.02	(0.027 - 0.186)	0.839	0.950	0.850
STFL195	240	149	0.22	83.000 ± 0.070	0.01	(0.018 - 0.163)	0.730	0.750	0.750

Table 28. Scissor-tailed Flycatcher Treatment ANOVA. This table depicts results of a single factor ANOVA test comparing treatments of the Scissor-tailed Flycatcher data from data collected from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA. There were no significant differences.

SUMMARY

Groups	*Count*	*Sum*	*Average*	*Variance*
CH	5	0.954	0.191	0.009
CM	5	0.888	0.178	0.004
RH	5	0.952	0.190	0.006
RM	5	0.620	0.124	0.006
O	5	0.662	0.132	0.008

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Between Groups	0.0210	4	0.0052	0.8084	0.5344	2.8661
Within Groups	0.1298	20	0.0065			
Total	0.1508	24				

Table 29. Scissor-tailed Flycatcher Years ANOVA. This table depicts results of a single factor ANOVA test comparing years of the Scissor-tailed Flycatcher data from data collected from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA.

SUMMARY

Groups	*Count*	*Sum*	*Average*	*Variance*
2015	5	1.211	0.242	0.002
2016	5	0.995	0.199	0.002
2017	5	0.477	0.095	0.001
2018	5	0.823	0.165	0.012
2019	5	0.568	0.114	0.002

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Between Groups	0.0730	4	0.0183	4.6962	0.0078	2.8661
Within Groups	0.0777	20	0.0039			
Total	0.1508	24				

Table 30. Scissor-tailed Flycatcher ANOVA Table. This table depicts an analysis of variance table, including degrees of freedom (DF), sum of squares (Sum Sq), mean squares (MS), F value, and *P*-value [Pr(>F)] from density estimates of the Brown-headed Cowbird from data collected from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA.

	DF	Sum Sq	MS	F Value	Pr(>F)
years	1	0.0432	0.0432	9.1140	0.0086
treatments	4	0.0202	0.0051	1.0660	0.4072
years:treatments	4	0.0132	0.0033	0.6940	0.6074
residuals	15	0.0711	0.0047		

Table 31. Significant Differences. This table shows where statistically significant differences occurred within the density estimate data. All species except the Brown-headed Cowbird and Eastern Meadowlark had significant differences between years, and the Eastern Meadowlark was the only species to exhibit statistically significant differences between treatments. Data was collected from 2015-2019 at the SAV ranch in Jim Hogg and Starr Counties, Texas, USA.

	Between Years	**Between Treatments**
Brown-headed Cowbird		
Dickcissel	X	
Eastern Meadowlark		X
Grasshopper Sparrow	X	
Northern Bobwhite	X	
Scissor-tailed Flycatcher	X	