



Biology of Tropical Birds

# Territoriality, breeding philopatry, and nest site selection of Chestnut Seedeater (*Sporophila cinnamomea*) in grasslands of southern Brazil

## Territorialidad, filopatría reproductiva, y selección del sitio de nidificación de *Sporophila cinnamomea* en pastizales del sur de Brasil

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**ABSTRACT.** We provide the first insights on the territorial behavior of the Chestnut Seedeater (*Sporophila cinnamomea*), a vulnerable, migratory species inhabiting the Pampas grasslands of South America. Our study aims to describe and estimate territory size, calculate rates of philopatry and breeding dispersal, describe the main aspects of the species' territorial behavior, and characterize nest sites. Over four breeding seasons (2018–2022), we monitored 70 adult individuals across two grassland areas of southern Brazil. We applied the Kernel Density Estimation method to estimate the territory size and core areas and analyzed the composition and floristic structure of nest sites. The average size of breeding territories was  $1.91 \pm 0.59$  ha, and the core areas were  $0.23 \pm 0.05$  ha. Remarkably, the average size of territories of philopatric males did not significantly vary in consecutive breeding seasons. The breeding philopatry rate was 51.5%. Moreover, we observed no significant differences in dispersion distance among consecutive breeding seasons, suggesting stable breeding site fidelity. Analysis of nest sites identified 14 plant species exhibiting higher abundance, with dense and tall vegetation prevalent. These findings underscore the importance of understanding population dynamics and the composition and structure of vegetation within Chestnut Seedeater nest sites as initial steps toward developing effective management and conservation strategies for their habitats.

**RESUMEN.** Proporcionamos las primeras ideas sobre el comportamiento territorial de *Sporophila cinnamomea*, una especie vulnerable y migratoria que habita los pastizales pampeanos de Sudamérica. Nuestro estudio tiene el objetivo de describir y estimar el tamaño del territorio, calcular las tasas de filopatría y dispersión reproductiva, describir los principales aspectos del comportamiento territorial, y caracterizar los sitios de nidificación de la especie. Durante cuatro temporadas reproductivas (2018–2022), monitoreamos 70 individuos adultos en dos áreas de pastizales del sur de Brasil. Aplicamos el método de Estimación de Densidad de Kernel para estimar el tamaño de los territorios y las áreas centrales, y analizamos la composición y estructura florística de los sitios de nidificación. El tamaño promedio de los territorios reproductivos fue de  $1,91 \pm 0,59$  ha, y las áreas centrales fueron de  $0,23 \pm 0,05$  ha. Notablemente, el tamaño promedio de los territorios de los machos filopátricos no varió significativamente entre temporadas reproductivas consecutivas. La tasa de filopatría reproductiva fue del 51,5%. Además, no observamos diferencias significativas en la distancia de dispersión entre temporadas reproductivas consecutivas, lo que sugiere una fidelidad estable al sitio de reproducción. Mediante el análisis de los sitios de nidificación se identificaron 14 especies de plantas que presentaron la mayor abundancia, siendo prevalente la vegetación densa y alta. Estos hallazgos remarcan la importancia de comprender la dinámica poblacional y la composición y estructura de la vegetación dentro de los sitios de nidificación de *Sporophila cinnamomea* como pasos iniciales hacia el desarrollo de estrategias efectivas de manejo y conservación para sus hábitats.

**Key Words:** grassland bird; Neotropical birds; nest site selection; Pampas; philopatry; songbirds; territorial behavior

### INTRODUCTION

Territoriality is a prevalent behavioral phenomenon in animals, driven by competition for essential resources such as food, copulation opportunities, shelter, and hierarchical dominance (Stutchbury and Morton 2001, Alcock 2016). This behavior is particularly prominent in bird species, including passerines, which establish and defend territories (i.e., exclusive area utilized by a pair within their home range to activities related to nesting) during the breeding season (Bibby et al. 1992, Powell 2000, Winkler 2016). Territorial behavior is often characterized by inter- and intraspecific agonistic encounters, which peak at the onset of the breeding season when territories are established (Stutchbury and

Morton 2001). These interactions involve various communicative mechanisms such as song, call notes, displays, and physical contact (Powell 2000, Byers and Kroodsma 2016).

Territory selection in birds is an adaptive behavior closely linked to vegetation structure and composition within the habitat, playing a critical role in their life history, including fecundity, survival, and nest success (Azpiroz and Blake 2016). Breeding territories serve multiple functions, including providing access to important reproductive resources such as a stable food supply and an optimal microclimate. In addition, they reduce predation risk through nest concealment within vegetation, while also offering thermoregulatory advantages during incubation (Marini et al.

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2010, Azpiroz and Blake 2016, Winkler 2016). Male birds often defend territories to attract mates, deter extra-pair copulation, and provide resources for offspring (Stutchbury and Morton, 2001).

Breeding philopatry (i.e. the tendency of returning to the same breeding area each year) is intricately linked to individual dispersal patterns, with these movements significantly influencing population dynamics, gene flow, and genetic diversity (Fajardo et al. 2009, Sia et al. 2020). In many bird species, females tend to disperse more for reproduction, while breeding site fidelity is male-biased (Dingle 1996, Beier et al. 2017, Browne et al. 2021). For instance, some South American grasslands migratory bird species, such as Black-bellied Seedeater (*Sporophila melanogaster*), Tropeiro Seedeater (*Sporophila beltoni*), Lined Seedeaters (*Sporophila lineola*), and Ibera Seedeater (*Sporophila iberensis*) exhibit a remarkable behavioral pattern of philopatry, returning to the same area or nearby in successive years for breeding (Winkler et al. 2016, Repenning and Fontana 2019a, Browne et al. 2021, Martins et al. 2021). This behavioral trait underscores the importance of comprehending breeding philopatry to elucidate population dynamics across different species.

Given the ecological significance of territoriality and philopatry, a comprehensive understanding of these behaviors in the Neotropical region is imperative, particularly for globally threatened bird species with South American temperate-tropical migration patterns (Jahn et al., 2020). In this study, we aimed to address knowledge gaps regarding the Chestnut Seedeater (*Sporophila cinnamomea*), a threatened species of South America. Considering the rapid degradation of grassland habitats in southern South America (Azpiroz et al. 2012), acquiring fundamental information about Chestnut Seedeater biology, including behavior and territoriality, is critical for developing effective conservation strategies for this species and its congeners. Studies focusing on threatened species' habitats are necessary to establish priority conservation areas and action plans (Di Giacomo et al. 2010). Specifically, our objectives were to (1) describe and estimate territory size, (2) calculate rates of breeding philopatry and breeding dispersal, (3) investigate whether the territory size of philopatric breeders varied between breeding seasons (2018–2019, 2019–2020), (4) assess dispersal rates among four breeding seasons (2018–2022), and (5) describe the main aspects of the species' territorial behavior. In addition, (6) we provided insights into nest site selection behavior and characterized nest sites used by Chestnut Seedeaters, comparing vegetation characteristics between nest sites and control sites.

We assumed that Chestnut Seedeaters would select habitats characterized by dense and tall vegetation for nest building, consistent with observations in other species within the genus. These habitats typically feature vegetation ranging from 80 to 100 cm in height, as observed in related species such as the Black-bellied Seedeater, Tropeiro Seedeater, and Rusty-collared Seedeater (*Sporophila collaris*; Rovedder 2011, Repenning 2012, Rosoni et al. 2019a). Furthermore, we anticipated that vegetation composition and structure would significantly influence the selection and establishment of breeding territories by Chestnut Seedeaters.

## METHODS

### Study species

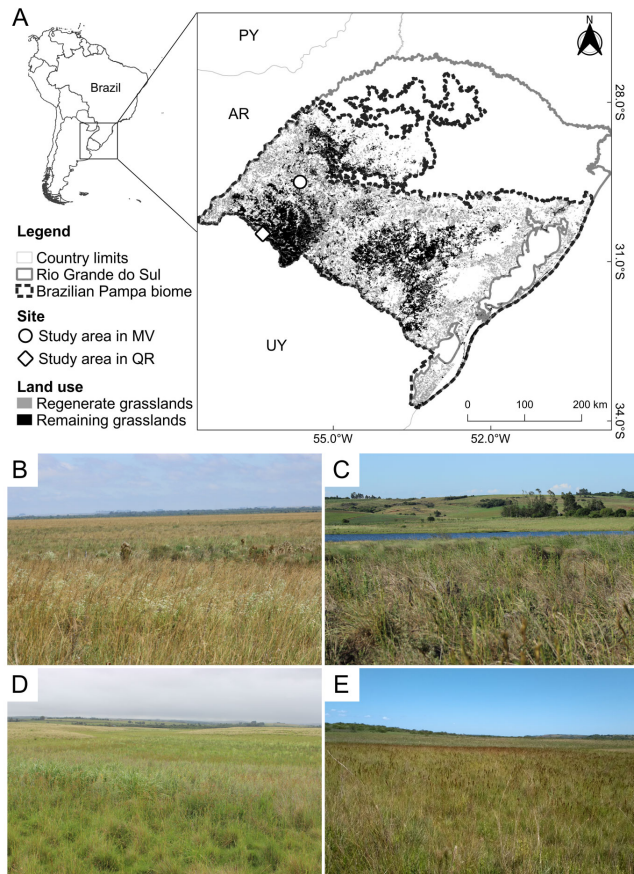
The Chestnut Seedeater belongs to the “southern capuchino seedeaters,” a monophyletic group comprising 10 species characterized by smaller body sizes compared to their congeners (Ridgely and Tudor 1989, Estalles et al. 2022). This species breeds in the temperate grasslands of northeast Argentina, western and southeast Uruguay, southeast Paraguay, and southern Brazil during the austral spring and summer (BirdLife International 2024, Rosoni et al. 2023), and after breeding, migrates northward to the tropics of central Brazil (Silva 1999, Rosoni 2022). The species occur sympatrically with other capuchinos species at breeding sites in the South American grasslands (Ridgely and Tudor 1989, Silva 1999, Campagna et al. 2015, Estalles et al. 2022). The Chestnut Seedeater predominantly inhabits tall grassland adjacent to moist swales, moist shrubby marsh, or wet savanna (Jaramillo 2020).

The Chestnut Seedeater exhibits sexual dichromatism, with males predominantly chestnut-colored with a grey cap, wings, and tail, while females are uniformly dull brownish (Repenning and Fontana 2019b, BirdLife International 2024). Furthermore, only males possess the ability to produce the pure, long, and very distinctive song (Repenning and Fontana 2019b). Currently classified as globally Vulnerable, the Chestnut Seedeater population is estimated to consist of between 2500 and 9999 mature individuals in the wild (BirdLife International 2024). This species faces a rapid population decline primarily because of the intense selective capture of males for the illegal trade, driven by their striking plumage and prized song. This practice causes a significant population imbalance among reproductive individuals. In addition, extensive habitat fragmentation and loss exacerbate the decline. Rapid afforestation with *Eucalyptus* and *Pinus* spp. affects even wet valley bottoms, despite subsequent poor tree growth. Pesticides and other chemicals, carried by drainage and runoff, directly contaminate marshes. Furthermore, mechanized agriculture, invasive grasses, and annual burning pose significant threats to their wintering, breeding, and migratory habitats (BirdLife International 2024).

### Study area

We carried out fieldwork in two municipalities in the central-western region of the state of Rio Grande do Sul, in the Pampas grasslands of southern Brazil (Fig. 1A). We monitored Chestnut Seedeater populations in the municipality of Manoel Viana on cooperative properties in the Santa Maria do Ibicuí Settlement (29°29'51.85"S; 55°38'18.35"W; 73 m a.s.l.; 647 ha) and on private farms (30°29'06.95"S; 56°20'27.32"W; 103 m a.s.l.; 679 ha) in the municipality of Quaraí. Both sites are within the most extensive and threatened grassland ecosystem in the Neotropics, the grasslands of southeastern South America (Azpiroz et al. 2012) and include a mosaic of vegetation types dominated by grasslands on dry soils and marshes (Fig. 1B-E). The vegetation is dominated by Poaceae, Apiaceae, and Onagraceae (Repenning and Fontana 2008, Rosoni et al. 2021). The regional climate is humid mesothermal (Cfa in Köppen-Geiger's classification) with hot summers, precipitation distributed throughout the year, and an average annual temperature of 22 °C (Alvares et al. 2013).

**Fig. 1.** (A) Geographic location of the study areas in Rio Grande do Sul, in the Pampas grasslands of southern Brazil. White circle: study area in Manoel Viana, MV; white diamond: study area in the municipality of Quaraí, QR. Neighboring countries: Paraguay (PY), Argentina (AR), and Uruguay (UY). Grassland landscapes in the study areas in Manoel Viana (B–C) and Quaraí (D–E).



### Breeding territories and philopatry

We define breeding territories as areas defended by males through song, call notes, displays, and physical contact (Powell 2000). Over four breeding seasons, from October to March in 2018–2019 and 2019–2020, and from November to January in 2020–2021 and 2021–2022, we captured males within their breeding territories using mist nets attracting them to the nets using playback. Playback was only played once to minimize disturbance. In comparison, females were opportunistically captured because they did not respond to playback. We determined the sex and age of the individuals based on plumage, cloacal protuberance, and song. We captured and banded individuals with a combination of a Brazilian National Conservation Center (CEMAVE) metal band and a unique combination of colored plastic bands to make birds individually identifiable.

We searched for nests of banded birds by systematically walking in places where males were singing and defending territories, or where adults were observed carrying nest materials or food for

the nestlings (Winter et al. 2003). We marked the geographical coordinates of each breeding territory using the Android application AlpineQuest Off-Road Explorer version 2.3.2. To do this, we recorded the coordinates of each nest site, the centroid of the estimated territory, or the locations where individuals were captured using mist nets. We used these coordinates as reference points to locate the territories during consecutive breeding seasons. Because we did not find any nests to assign specific geographic coordinates, we relied on the coordinates of the estimated territory centroid and the capture locations of individuals. We monitored individuals that returned to the breeding area during successive breeding seasons (2019–2022). Only males that established their territories within the study areas were considered as philopatric breeders (Sia et al. 2020). We calculated breeding philopatry based on the annual return rate of banded birds from one year to the next, categorized by sex (males and females; Pasinelli et al. 2007, Repenning 2012). Breeding dispersal (measured in meters) was determined as the distance between the reference point in year “*t*” and the initial location of the individual singing and defending the territory in year “*t* + 1” (adapted from Fajardo et al. 2009).

### Determining and estimating territory sizes

We collected data from each banded male using binoculars for two hours in the morning and two hours in the afternoon, maintaining a distance of 20–30 m from focal individuals (Franz 2012, Repenning and Fontana 2016). These surveys were carried out during the early (November) and middle (December/January) stages of the breeding season because this is when males typically exhibit more behaviors related to defending their territories (Rosoni et al. 2019a). We determined the boundaries of breeding territories by closely monitoring agonistic social behavior and male territoriality (Alcock 2016), e.g., agonistic singing, chasing other individuals, territorial singing, and physical combat (Bibby et al. 1992). Using AlpineQuest Off-Road Explorer, we recorded specific points corresponding to perches frequently used by males for singing or defending their territory. To minimize observational bias, we recorded a new point each time the male left the perch spontaneously, thereby avoiding any observer influence on its behavior (Martins et al. 2021). Moreover, if the same perch was used repeatedly, we re-recorded it. Finally, we considered a minimum of 25–30 locations per animal adequate for the estimation analyses (Seaman et al. 1999, Martins et al. 2021).

We estimated the size of breeding territories using the non-parametric Kernel Density Estimation (KDE) method, based on 95th density isopleth (KDE 95th; Worton 1989, Seaman et al. 1999, Powell 2000, Barg et al. 2005, Calenge 2006). This method was computed with a contour level of 95% and using the reference bandwidth, as it is suitable for situations where the locations form a single tight cluster (Worton 1989, Martins et al. 2021). In addition, we adapted the rule-based ad hoc bandwidth approach suggested by Brack (2013), named the population smoothing approach (hpop). To obtain the hpop, we initially calculated the bandwidth for each individual using the reference bandwidth (href, Seaman et al. 1999). Subsequently, we determined the median value for all individuals from href ad hoc bandwidth (hpop = 15.92) and applied this value to the model for all individuals sampled. This hpop approach was chosen because it provided the most accurate contour value fitting to the distribution of locations, thereby preventing the generation of overestimated contour. Our territory size estimation was carried

out using the “adehabitatHR,” “maptools,” and “sp” packages in R software version 3.6.3 (Calenge 2006, Bivand et al. 2013, Bivand and Lewin-Koh 2020, R Core Team 2020). We calculated the area using the kernel.area function from the “adehabitatHR” package. The images of territories were edited in QGIS software version 3.14.15-Pi (QGIS Development Team 2022).

We assumed that core use areas (regions within territories intensively used by the birds) can often be associated with nest sites (Martins et al. 2021). To define the core use areas for each bird, we followed the methodology proposed by Harris et al. (1990) with modifications from Barg et al. (2005). Initially, we calculated the density of isolines from breeding territories at intervals of 10% between 10 and 90%. Subsequently, we determined the limit of the core area based on the lowest inflection point value between two isoline values (Harris et al. 1990). Finally, we calculated the inflection point using the formula proposed by Wal and Rodgers (2012), which involves identifying the isoline value that best fits the curve of the exponential regression equation through the least-squares fit. This method was chosen over the traditional 50% technique for identifying core use areas because it allows for data selection independent of sample size, considering the absence of a general area-sample size relationship among males. Finally, the core use areas were characterized in terms of percent of use (percent of total locations within the core area) and intensity (ratio between the percent of use and percent territory; Freitas et al. 2019).

#### Selection and characterization of the nest site

In the characterization of nest sites, the following variables were considered: environment type, vegetation cover, vegetation height, and plant density, composition, and abundance. At the end of the breeding seasons (March) in 2018–2019 and 2019–2020, we collected qualitative and quantitative information regarding vegetation composition, abundance, and floristic structure (Rosoni et al. 2019a) for 45 breeding territories. Our vegetation sampling commenced immediately after the inactivity of the last active nest in each breeding season in order not to disturb breeding birds. We assumed that plant species at the end of breeding season (summer) were mostly the same as at the beginning (spring), as winter grass species (non-breeding) are replaced by summer grass species in the spring (Repenning and Fontana 2016). We conducted our vegetation sampling using two 2 x 2 m plots, referred to as “nest” (n = 45) and “control” (n = 45) plots in each territory (Bibby et al. 1992). The nest plots were centered on the plant supporting the nest. To determine control plots, we created a grid mesh of the same size as the plots within each territory, assigning each grid a number that was later drawn by lot (adapted from Bibby et al. 1992). This random selection process ensured that each territory had a corresponding control plot. The control plots were employed to assess the accuracy of variables in the microhabitats concerning the selection of “used” versus “available” nest sites (Davis 2005), with the nest plot excluded from the random selection process. For territories with more than one nest located in the same territory, we randomly selected one nest plot and discarded the other nest plots for that pair.

We categorized the environment type where each marked individual’s nest was located and defined them (Cordeiro and Hasenack 2009, Areta and Repenning 2011) as follows:

1. Dry grasslands: areas with water deficits situated in high, flat areas amid well-drained regions. They are characterized mainly by the presence of grasses from genera *Stipa*, *Briza*, *Schizachyrium*, *Paspalum*, and *Axonopus*, along with *Eupatorium* shrubs, which are commonly found in dry soils.
2. Lowland marshes: areas subjected to seasonal flooding during the breeding season and are typically river meadows or flooded grasslands dominated by grasses, sedges, or rushes.
3. Wet grasslands: this is a transitional area between dry grassland and lowland marsh, and its humidity can vary based on precipitation levels. Wet grasslands are areas that are situated at a higher elevation compared to lowland marshes and are characterized by the presence of species like *Eryngium pandanifolium*, *Eleocharis* sp., *Andropogon lateralis*, and sparse *Bacharis* sp. shrubs.

We collected data on vegetation cover, individual height of the most abundant plant species, plant density, composition, and abundance at each nesting and control plots (Bibby et al. 1992). Grass species were meticulously sampled to avoid overestimating the number of individuals. Specifically, for grasses, we regarded the vegetative part of the plants as the individual, distinguishing, in certain instances, between the number of clumps (vegetative) and inflorescences (reproductive), as we regarded them as structurally distinct portions of the plants (Repenning 2012). First, we visually estimated the vegetation cover in three strata categories: (1) low, characterized by small grass and herbs ( $\leq 30$  cm in height); (2) medium, characterized by the presence of herbs, shrubs, and grasses with inflorescences ( $> 30$  cm and  $\leq 80$  cm in height); and (3) high, characterized by the presence of shrubs, clumps of grass with tall inflorescences ( $> 80$  cm in height; Rosoni et al. 2019a). For this estimate, zero indicates absence and 100% means full coverage for a given stratum (adapted from Jones and Robertson 2001). Then, we measured the height of each individual of the most common plant considering seven classes: 0–30 cm, 31–60 cm, 61–90 cm, 91–120 cm, 121–150 cm, 151–180 cm, and above 180 cm. The modal value of heights represented the plot’s height.

We estimated plant density using a 1 m<sup>2</sup> board divided into 16 quadrants, placed vertically on one side of the plot close to the ground. From the opposite side, kneeling at a distance of approximately 2 m, we identified vegetation and recorded the number of quadrants obstructed by vegetation, with values ranging from 0 (no obstruction) to 16 (total obstruction; Bibby et al. 1992). To assess composition and abundance in the sample plots, we calculated the relative abundance of individuals by plant species. Species with relative abundances of individuals lower than 1% were excluded from the analysis, resulting in 57 plant species for the analysis of abundance and composition.

#### Statistical analysis

We opted for non-parametric statistics because our data on territory size, core area, and breeding dispersal distance did not adhere to the assumptions of normality and homoscedasticity of variances. We used the Wilcoxon-Mann-Whitney U-test for dependent samples (Fowler and Cohen 1995) to compare breeding territory and core area sizes for philopatric males between breeding seasons. We applied the Kruskal-Wallis H-test to assess whether there were differences in breeding dispersal

distances among all breeding seasons (Fowler and Cohen 1995). We used the Wilcoxon-Mann-Whitney U-test for independent samples to compare breeding dispersal distances between breeding seasons in a pairwise manner (Fowler and Cohen 1995).

To analyze the relationship between vegetation structure and composition of nest and control plots, we utilized non-metric multidimensional scaling (NMDS) with the Gower index (Palacio et al. 2020). First, we conducted NMDS using the metaMDS function of the “vegan” package in R (Oksanen et al. 2020). Subsequently, we tested the significance of the variables using the Multivariate Permutation Analysis of Variance (PERMANOVA; Anderson and Walsh 2013) with 999 permutations, employing the adonis2 function of the “vegan” package in R (Oksanen et al. 2020). We then adjusted the environmental variables on the NMDS plot and determined their statistical significance, testing it with 999 permutations using the envfit function of the “vegan” package. To assess the percentage contribution of each plant species to between-group dissimilarity, we utilized the SIMPER function of the “vegan” package (Oksanen et al. 2020). We present the results with mean values  $\pm$  standard deviation and a significance level of  $\alpha = 0.05$  (Fowler and Cohen 1995).

## RESULTS

### Breeding territory and philopatry

In total, we banded 70 individuals (61 males and nine females) throughout the breeding seasons. We estimated the sizes of 39 breeding territories (see Table S1) for 30 different males, based on 1771 location points, averaging  $45.41 \pm 6.21$  points per territory (range 25–53 locations). The average size of breeding territories was  $1.91 \pm 0.59$  ha (range 0.97–3.70 ha). Most breeding territories ( $n = 31$ ) had one core area, and eight territories had two core use areas (Table S1). The average size of the core use areas was  $0.23 \pm 0.05$  ha (range 0.10–0.33 ha, Table S1). The core use areas varied in the total proportion of the breeding territory they occupied ( $13.37 \pm 5.84$ , range 3.14–27.49%), the proportion of use ( $47.14 \pm 11.90$ , range 18.75–68.09%), and the intensity of use ( $3.95 \pm 1.18$ , range 2.04–7.50%, Table S1). In 69% of the monitored territories, we identified contact areas between neighboring territories, resulting in small clusters comprising 2 to 8 contact zones. This clustering behavior is particularly notable in areas surrounded by crops and pastures with high cattle stocking (Fig. S1).

We measured the territories of nine philopatric males during two breeding seasons. The average size of breeding territories was  $1.99 \pm 0.43$  ha in the 2018–2019 season (range 1.39–2.82 ha) and  $1.90 \pm 0.74$  ha in the 2019–2020 season (range 1.18–3.70 ha). Four males showed an average increase of  $0.77 \pm 0.70$  ha in their breeding territories from the 2018–2019 season to the 2019–2020 season, while five males exhibited an average reduction of  $0.79 \pm 0.42$  ha in their territories during the same period. Despite these variations in territory size, there were no significant differences in territory size between the breeding seasons ( $W = 27$ ,  $p = 0.652$ ), nor in the size of the core use areas ( $W = 11$ ,  $p = 0.673$ ). We observed overlapping breeding territories in consecutive breeding seasons for eight males, with an average overlap area of  $0.55 \pm 0.24\%$  (range 0.19–0.93%).

Of the 61 males banded, 49% returned in the subsequent breeding season after banding, decreasing to 25% in the second and 16% in the third year. Of the 9 females banded, 11% returned in one season, 22% in two breeding seasons, and 11% in three breeding seasons (2019–2022). The average overall breeding philopatry rate over three breeding seasons (2019–2022), considering one sighting per individual, was 51.5% (birds banded in the first year: 89%; banded in the second year: 32%; banded in the third year: 32%, Table S2). Individuals (males and females) dispersed an average of  $164.7 \pm 264.2$  m from their territories between consecutive breeding seasons (Table 1). We did not find significant differences in dispersal among the three breeding seasons ( $H = 2.168$ ,  $p = 0.338$ ), nor in pairwise comparisons between 2019–2020 with 2020–2021 ( $W = 322$ ,  $p = 0.213$ ), 2020–2021 with 2021–2022 ( $W = 94$ ,  $p = 1$ ), and 2019–2020 with 2021–2022 ( $W = 209$ ,  $p = 0.281$ ).

**Table 1.** Descriptive statistics for philopatric individuals of Chestnut Seedeater (*Sporophila cinnamomea*) in breeding dispersal monitored during the breeding seasons 2018–2022, in the grasslands of Rio Grande do Sul, in the Pampas grasslands of southern Brazil. Distances are in meters.

Breeding season	Average	Median	SD	Minimum	Maximum	N
One year	160.42	57.27	301.81	2.64	1556.98	31
Two years	186.42	146.40	192.24	10.68	661.01	17
Three years	195.18	64.46	307.75	22.27	1068.96	11

On one occasion, we identified a female dispersing among territories. In 2018, a female (C79908) successfully raised a nestling with a male (C79909). In the following breeding season, she dispersed 197.04 m and mated with the male from a neighboring territory (C79910), remaining paired with him in the subsequent breeding seasons (2019–2020, 2020–2021, and 2021–2022). The previous male (C79909) remained in the same territory in the following breeding seasons and paired with other female or females (unbanded). In another observation, we noted a territory change within the same breeding season (2019–2020). After successfully nesting with a male (C79903), a female (C111783) moved 1110.7 m and paired with another male (C111768). Males C79905 and C79919 changed territories between breeding seasons, dispersing 661.01 and 96.05 m, respectively.

### Territorial behavior

Males engaged in at least one interaction involving agonistic social behavior with another individual while defending their breeding territories in 26 territories, accounting for 67% of the observed territories. On average,  $1.64 \pm 1.74$  interactions per male were recorded, ranging from 0 to 7 interactions, with a total of 104 hours observed. Approximately 81% of these interactions occurred with conspecific males ( $n = 21$  males), while approximately 12% involved congener males ( $n = 2$  Marsh Seedeaters [*Sporophila palustris*];  $n = 1$  Rusty-collared Seedeater), and the remaining 8% involved conspecific females ( $n = 2$  females). Agonistic interactions with intruder males were more frequent when females displayed receptive behavior for copulation or during nest building in established territories. During these phases, we observed ( $n = 4$ ) that territorial males

sometimes concealed females among the vegetation within the territory. Nevertheless, six attempts of extra-pair copulation by males who entered the territories were registered during the study. The defense of territories by males tended to decrease as the breeding season progressed. Toward the end of the breeding season, typically from late January onward (as mentioned in Rosoni 2022), males showed fewer agonistic displays within their territories. During this period, mixed intraspecific flocks consisting of juveniles, females, and other males were frequently observed in territories, and the boundaries between territories and feeding areas became less distinct and harder to observe.

### Selection and characterization of the nest site

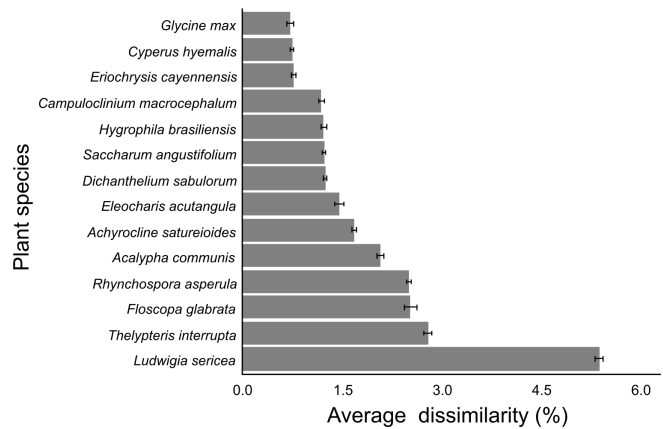
In six different occasions, males suggested potential nest sites to the females. These specific behaviors included wing shaking to expose their white wing speculum and adopting a brooding-like posture among the branches of supporting plants. Subsequently, females chose these suggested sites for nesting.

Most nests (51%,  $n = 23$ ) were in lowland marsh environments, followed by 38% ( $n = 17$ ) in wet grassland and 11% ( $n = 5$ ) in dry grassland. In our study area, lowland marsh environments primarily consisted of the lower stratum presence of small grasses with vegetative parts or small clumps of *Luziola peruviana* (Poaceae), *Rhynchospora asperula*, *Cyperus hyemalis* (Cyperaceae), and shrubs such as *Ludwigia sericea* (Onagraceae). The middle stratum was characterized by medium-sized herbs, including *Floscopa glabrata* (Commelinaceae), *L. sericea*, *Thelypteris interrupta* (Thelypteridaceae), and medium-sized inflorescences of *Rhynchospora asperula* and *Eleocharis acutangula* (Cyperaceae). In the upper stratum, larger shrub species such as *L. sericea*, *Senecio brasiliensis* (Asteraceae), and long inflorescences of *Eriochrysis cayennensis* (Poaceae), *L. peruviana*, and *Eryngium pandanifolium* (Apiaceae) predominated. In wet grassland environments, the lower stratum was dominated by small herbs and shrubs of *Acalypha communis* (Euphorbiaceae), *Achyrocline satureioides*, *Cirsium laevigatum* (Asteraceae), *L. sericea*, and medium-sized inflorescences of *Andropogon virgatus* (Poaceae). In the upper stratum, larger shrub species such as *L. sericea*, *C. laevigata*, and long inflorescences of *Schizachyrium microstachyum*, *Sorghastrum setosum*, and *Eragrostis airoides* (Poaceae) were prevalent. Finally, in dry grassland environments, two strata predominated. The lower stratum in these environments consisted of small herbs like *Centella asiatica* (Apiaceae), small grasses with vegetative and inflorescences parts *Steinchisma hians*, *S. microstachyum*, *Andropogon lateralis* (Poaceae), and *Hatschbachiella tweedieana* (Asteraceae). The middle stratum was characterized by medium-sized herbs, including *Richardia humistrata* (Rubiaceae), *Ambrosia tenuifolia* (Asteraceae), and medium-sized inflorescences of *S. microstachyum* and *Melinis repens* (Poaceae).

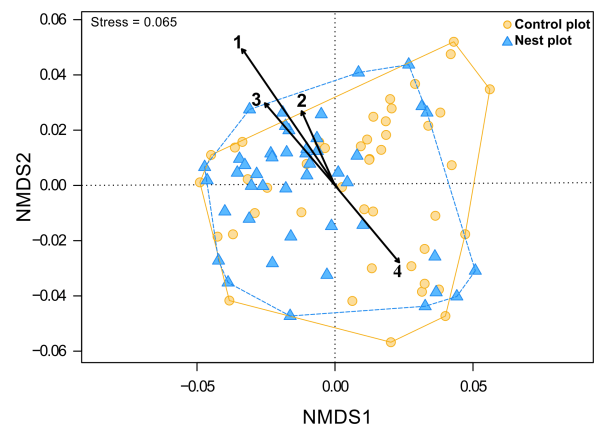
We found differences in the abundance and composition of plant species between the nest and control plots (PERMANOVA:  $F = 2.05$ ,  $r^2 = 0.02$ ,  $p = 0.01$ ), with 14 plant species being more abundant (25.52%) in the nesting plots (Fig. 2). Furthermore, differences were identified between the nest and control plots across the three environments used as nest sites (PERMANOVA:  $F = 4.18$ ,  $r^2 = 0.04$ ,  $p = 0.04$ ). We noted a slight clustering of nest plots in relation to predictors: plant density (envfit:  $r^2 = 0.51$ ,  $p = 0.001$ ), high stratum (envfit:  $r^2 = 0.21$ ,  $p = 0.001$ ), and height of

vegetation (envfit:  $r^2 = 0.12$ ,  $p = 0.002$ ). This suggests that the vegetation in the nesting plots tended to be relatively tall and dense (Fig. 3). In contrast, the control plots showed a subtle trend of association with the lower stratum of the vegetation (envfit:  $r^2 = 0.18$ ,  $p = 0.001$ , Fig. 3). Although the differences were subtle (~15% dissimilarity, Table 2), the vegetation within the breeding territories presented a slightly different structure compared to nesting and control plots.

**Fig. 2.** The abundance of plant species most significantly contributing to the dissimilarity between nest and control plots in breeding territories of Chestnut Seedeater (*Sporophila cinnamomea*) during the two breeding seasons 2018–2020 in the grasslands of Rio Grande do Sul, in the Pampas grasslands of southern Brazil. The standard deviation of the average contribution percentage is represented by bars on the graph.



**Fig. 3.** Non-metric multidimensional scaling (NMDS) between nest and control plots of nest sites of Chestnut Seedeater (*Sporophila cinnamomea*) in breeding territories during two breeding seasons (2018–2020) in the grasslands of Rio Grande do Sul, in the Pampas grasslands of southern Brazil. Results are based on plant species composition and abundance (Gower dissimilarity index). Environmental variables best fit: 1–plant density; 2–plant height. Vegetation cover: 3–high stratum and 4–low stratum.



**Table 2.** Variables used in non-metric multidimensional scaling (NMDS) fitted for nest and control plots in nest sites of Chestnut Seedeater (*Sporophila cinnamomea*) grasslands of Rio Grande do Sul, in the Pampas grasslands of southern Brazil. Asterisks denote levels of significance.

Variables	df	R <sup>2</sup>	F	P
Environments	1	0.01	0.916	0.541
Plant density	1	0.043	4.078	0.001*
Height	1	0.01	0.985	0.467
Lower stratum	1	0.019	1.858	0.044*
Middle stratum	1	0.021	1.963	0.032*
High stratum	1	0.024	2.243	0.019*

## DISCUSSION

Investigating the territorial behavior of the Chestnut Seedeater, we observed that the species establishes and maintains breeding territories throughout the breeding season. Approximately 50% of individuals return to the study area for multiple breeding seasons, indicating a notable degree of site fidelity. Particularly, male Chestnut Seedeaters exhibit a high level of breeding philopatry. Furthermore, our study revealed that birds selected nest sites characterized by dense and tall vegetation. These findings align with observation of other Neotropical passerine species inhabiting grasslands habitats (e.g., Di Giacomo et al. 2010, Azpiroz and Blake 2016).

### Breeding territory and philopatry

Compared to other migrant seedeaters, such as the Black-bellied Seedeater, Tropeiro Seedeater, and Lined Seedeater, the Chestnut Seedeater possesses relatively large breeding territories: ranging from 0.93 to 3.70 ha versus 0.27 to 1.60 ha (Rovedder 2011, Repenning 2012, Martins et al. 2021). Several factors may influence core area size, territory size and clustering, including food availability, competitor density, and the availability of suitable nest sites (Hill and Lein 1989). The core use areas observed in Chestnut Seedeaters were consistent in size among individuals and across breeding seasons. Similar stability in core area sizes over time has been reported in studies on Cerulean Warblers (*Setophaga cerulea*, Parulidae) in Ontario, Canada (Jones and Robertson 2001). Interestingly, the selection of core use areas in these warblers was influenced by the diameter of trees within the territories (Jones and Robertson 2001). On the other hand, territory clustering in species like Black-capped Chickadees (*Poecile atricapillus*, Paridae) and Mountain Chickadees (*P. gambeli*) in Canada was associated more with the availability of suitable nest sites than with competition for food (Hill and Lein 1989). Therefore, it is likely the clustering of Chestnut Seedeater territories and core area, with slight variations in size, can be attributed to a complex interplay of resources, including available seeds, grass, nest materials, and nest substrates. Although such clustering may seem common in *Sporophila* seedeaters (Rovedder 2011, Repenning 2012, Rosoni et al. 2019a), we suggest that these factors contribute to the stabilization of core use areas and, consequently, to territory clustering in habitats that potentially offer higher-quality food resources and greater materials for nest building.

The rate of breeding philopatry in the Chestnut Seedeater falls within the range observed in other South American temperate-tropical migrant passerines (Bravo et al. 2017, Jahn et al. 2020, Sia et al. 2020). This outcome confirms that the Chestnut Seedeater demonstrates a strong degree of fidelity to its breeding area, particularly among males. Our data align with those for the Tropeiro Seedeater, which exhibited an average philopatry rate of 60% over a three-year study (Repenning 2012). A similar pattern was observed in Ibera Seedeater males in Argentina, with an initial philopatry rate of 47%, subsequently decreasing to 28% and 6% in the third year (Browne et al. 2021). In Tropeiro Seedeaters and Ibera Seedeaters, philopatry rates for females remained below 30% (Repenning 2012, Browne et al. 2021), but not lower as found here. Two non-mutually exclusive factors may explain the lower philopatry rates observed in females: (1) females tend to disperse more widely than males (Fajardo et al. 2009, Winkler et al. 2016, Sia et al. 2020); or (2) female seedeaters are often less conspicuous than males, making them harder to detect (Fajardo et al. 2009, Repenning and Fontana 2019b). However, it is worth noting that our female dispersal rates should be interpreted cautiously because of the relatively small sample size of banded females, which might lead to an underestimation of their philopatry. Therefore, understanding females' dispersal behavior between and during breeding is a challenge that can be answered with long-term monitoring studies in the future.

### Territorial behavior

Territorial males of the Chestnut Seedeater intensified territorial behavior when females were engaged in the nest-building phase and were most receptive to copulation. This behavior is justifiable because it serves to deter extra-pair copulation attempts (Stutchbury and Morton 2001). Similar behavior has been observed in other seedeater species during the nest-building phase, including the Black-bellied Seedeater, Tawny-Bellied Seedeater (*Sporophila hypoxantha*), Tropeiro Seedeater, and Lined Seedeater (Rovedder 2011, Franz 2012, Repenning 2012, Martins et al. 2021). In line with Martins et al. (2021), we propose that males defend breeding territories primarily to prevent extra-pair copulation, rather than solely because of food availability as primarily suggested (Stutchbury and Morton 2001, Rovedder 2011).

Breeding dispersal over short distances, with an average of 165 m, emphasizes the Chestnut Seedeater's fidelity to its breeding area. These results contrast with the much longer dispersal distances recorded in male Tropeiro Seedeaters (average of 734 m) and females (average of 2064 m; Repenning 2012). In general, in birds, breeding dispersals tend to involve shorter-distance movements (Winkler et al. 2016), with females typically undertaking more frequent and wider dispersal compared to males (Newton 2008). Our findings align with this broader pattern, highlighting that Chestnut Seedeater dispersal is biased toward females, with males being more philopatric. This suggests that females play a crucial role in reducing inbreeding (Newton 2008). Once again, the result should be used with caution because of the small size of the female sample.

Male Chestnut Seedeaters appear to suggest potential nest sites to females, akin to a behavior observed in other seedeaters (Rovedder 2011, Repenning and Fontana 2019a, Ferreira et al.

2022). Ferreira et al. (2022) propose that the male's display of potential nest sites to females may be a widespread phenomenon among seedeaters. This behavior has also been documented in various bird species, including weavers, herons, and wrens (Winkler 2016). Typically, such behavior is linked to species that defend territories and build nests that require substantial energy because these nests are constructed rapidly, have a short incubation period, and experience swift nestling development (Rovedder 2011, Franz 2012, Repenning 2012, Rosoni et al. 2019b). Furthermore, the selection of a nesting site appears to involve "negotiation" between the male and female, as they must both agree on the location and participate in nest construction (Winkler 2016) or defense, in the case of Chestnut Seedeaters (this study).

#### **Selection and characterization of the nest site**

The selection of specific territory locations is evident from the statistically significant differences in plant density and vegetation height between the nest plots and control plots. Similar distinctions in plant species composition between nest and control plots were observed in the Black-bellied Seedeater, indicating that the choice of nest sites among southern capuchinos is not randomly chosen (Rovedder 2011). In addition, fourteen plant species exhibited higher abundance in nest sites, notably including *Glycine max*, an exotic species that dominated one breeding territory, encompassing more than 60% of the soybean crop area within its boundaries. A similar study on the Rusty-collared Seedeater in the same region as our study did not find differences in plant species composition between the plots. However, in the sampled plots, the presence of *Manihot esculenta*, an exotic species, was identified as one of the most abundant species (Rosoni et al. 2019a). Nevertheless, the preference for nest sites with dense and tall grassland vegetation (ranging from 60 to 100 cm) appears to be a common trait among grassland seedeaters (Rovedder 2011, Repenning 2012, Rosoni et al. 2019a).

Similarly, North American bird species also demonstrate similar patterns in nest site selection. For instance, in West Virginia, USA, Bobolinks (*Dolichonyx oryzivorus*, Icteridae) had higher nest success rates in sites with dense vegetation, while Savannah Sparrows (*Passerculus sandwichensis*, Passerellidae) and Red-winged Blackbirds (*Agelaius phoeniceus*, Icteridae) showed the highest success rates in sites with tall vegetation (Warren and Anderson 2005). The breeding and nest success of grassland birds appear strongly linked to the plant structure and composition at their nest sites (Azpiroz and Blake 2016). Our findings regarding nest site selection indicate that the Chestnut Seedeater is not only a species obligate to grasslands but also highly reliant on high-density, well-preserved grasslands for breeding and feeding (Bencke 2009, Azpiroz et al. 2012, Rosoni et al. 2021, 2023).

#### **Concluding remarks and conservation recommendations**

In the study area, lowland marshes account for 7% of the total area, followed by wet grasslands at 15%, with dry grasslands predominating at 78%. Interestingly, most Chestnut Seedeater territories were located in remote lowland marsh environments that are difficult to access. Within these marshes, at least five core use areas associated with nest site selection were identified, characterized by dense emergent vegetation. Such areas may serve as breeding refuges by providing high-quality, non-productive sites within the agricultural matrix, and thus exhibit greater

resilience to cattle access (Pucheta et al. 2024). Nest building primarily occurs in the lowland marsh environments. These habitat preferences align with those observed in other members of the southern capuchinos group, including the Black-bellied Seedeater, Ibera Seedeater, Marsh Seedeater, and Pearly-bellied Seedeater (*Sporophila pileata*; Rovedder 2011, Vizentin-Bugoni et al. 2013, Freitas et al. 2018, Browne et al. 2021). In contrast, habitat preferences in other seedeater species, such as Tropeiro Seedeater and Rusty-collared Seedeater, span wet and dry grasslands associated with water bodies (Repenning 2012, Rosoni et al. 2019a). The preference for wet environments may be an inherent or learned trait among southern capuchinos, given their frequent coexistence in the grasslands of southeastern South America (Azpiroz et al. 2012, Campagna et al. 2015). Notably, a study involving the Lesser Grass-Finch (*Emberizoides ypiranganus*, Thraupidae) in southern Brazil, conducted in areas with controlled fires vs. areas burned without controlled fires, revealed adaptive behavior in the establishment and size of breeding territories. Intriguingly, areas subject to controlled fires exhibited a higher proportion of breeding territories in the lowland marsh environment because these areas remained unaffected by the fires while maintaining the appropriate vegetation structure for shelter and nest sites (Chiarani and Fontana 2015). This emphasizes the critical significance of comprehending the precise location, composition, and intensity of core use areas. Such insights hold the potential to guide tailored management and conservation strategies, a matter of utmost importance (Wal and Rodgers 2012), especially for threatened species like the Chestnut Seedeater.

Our findings suggest that conservation efforts should encompass not only the mitigation of disturbances, such as cattle grazing at inappropriate densities but also the coordination and implementation of sustainable practices, or the restoration of areas undergoing conversion or with low vegetation coverage, particularly in lowland marsh environments within our study area. Preserving such areas as Permanent Protection Areas in accordance with the law and restoring lowland marsh or wet grasslands areas could potentially facilitate the expansion and establishment of new breeding territories for the Chestnut Seedeater. Furthermore, these restored habitats may serve as vital stopover sites for individuals migrating to other regions between migratory periods.

This study represents the first investigation into the territorial behavior and nest site selection of the Chestnut Seedeater. By addressing these aspects, our data contribute significantly to filling critical gaps identified in the Brazilian National Action Plan for the Conservation of South Brazilian Grassland Birds. This plan outlines conservation strategies for 27 endangered grassland bird species, including seven *Sporophila* seedeaters (ICMbio 2021). In addition, our findings on breeding philopatry, breeding dispersal, and the size of core use areas and breeding territories provide essential groundwork for future research on the population dynamics of the Chestnut Seedeater and its congeners. Understanding the composition and plant structure of these species' breeding territories and nest sites is paramount for developing more effective native grassland management strategies, particularly for the conservation of grassland birds in private land and Brazilian Conservation Units (Protected Areas).



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#### Author Contributions:

JRRR, CSF, and CJC conceptualized the study. JRRR curated the data. JRRR performed the analysis. JRRR, CSF, and CJC performed funding acquisition. JRRR performed investigation. JRRR, CSF, and CJC elaborate the methodology. JRRR, CSF, and CJC managed the project. CSF and CJC supervised the project. JRRR wrote the first original draft paper. JRRR, CSF, and CJC revised the manuscript. All authors read and approved the final manuscript.

#### Acknowledgments:

We are grateful to all the Santa Maria do Ibicuí Settlement people, especially J. Paiva and V. Schwendler, for their hospitality; we thank the owners I. Wagner, C. S. Fontana, A. Quinteiro, B. Pujol, M. Brasil (F. Outeiro, in memoriam), and H. Guterres for allowing us to access their properties; J.P.G. Just, C.A. Aguiar, A.V. Santana, and J. Lopes for the company and help during the fieldwork; M. Vieira (UFRGS) who kindly helped to identify the plants. This research is part of a project supported by the Rufford Foundation, United Kingdom (ID 27044-1), through Igré - Associação Sócio-Ambientalista. This study was supported by grants to J.R.R.R. and C.J.C. who received doctoral and postdoctoral fellowships, respectively, from the Coordenação de Aperfeiçoamento de Pessoal de Nível Superior – CAPES (88882.439380/2019-01 and 88882.316294/2019-01), and to C.S.F., who received a research grant from the Conselho Nacional de Desenvolvimento Científico e Tecnológico – CNPq (310608/2019-8 and 308700/2022-8). This study was carried out under licenses granted by the Instituto Chico Mendes de Conservação da Natureza – ICMBio (SISBIO 63556-1) and by the Centro Nacional de Pesquisa e Conservação de Aves Silvestres – CEMAVE (433211).

#### Data Availability:

All data are included in the manuscript.

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## Appendix 1

### Territoriality, breeding philopatry, and nest site selection of Chestnut Seedeater (*Sporophila cinnamomea*) in grasslands of southern Brazil

This document includes:

**Table S1.** Breeding territories of Chestnut Seedeater (*Sporophila cinnamomea*) estimated with KDE 95th density isopleth.

**Table S2.** Raw banding data of Chestnut Seedeater (*Sporophila cinnamomea*).

**Fig. S1.** Kernel contour of the estimated breeding territory of 15 adult male Chestnut Seedeaters (*Sporophila cinnamomea*).

**Table S1.** Breeding territories of Chestnut Seedeater (*Sporophila cinnamomea*) estimated with Kernel Density Estimator 95th density isopleth (KDE 95th). The number of locations recorded in the territory. Percentages used to estimate core area sizes for 30 males monitored during the breeding seasons 2018–2019 and 2019–2020 in the grasslands of Rio Grande do Sul, in the Pampas grasslands of southern Brazil.

Bird ID	Locations	Territory area (ha)	Core use area estimate (%)	Core use area (ha)	No. of cores	Territory (%) <sup>§</sup>	Use (%) <sup>#</sup>	Intensity <sup>††</sup>
C79902 <sup>†</sup>	51	2.25	0.3	0.2	1	9	41.2	4.6
C79903 <sup>†</sup>	50	1.53	0.4	0.29	1	19	50	2.6
C79904 <sup>†</sup>	53	2.16	0.2	0.17	1	7.7	30.2	3.9
C79905 <sup>†</sup>	46	2.22	0.2	0.17	1	7.4	34.8	4.7
C79906 <sup>†</sup>	50	0.97	0.5	0.25	1	25.3	68	2.7
C79907 <sup>†</sup>	50	1.57	0.4	0.26	1	16.7	50	3
C79909 <sup>†</sup>	50	1.94	0.3	0.22	1	11.5	42	3.7
C79910 <sup>†</sup>	45	1.68	0.4	0.26	1	15.6	51.1	3.3
C79911 <sup>†</sup>	48	1.67	0.3	0.23	1	13.9	37.5	2.7
C79913 <sup>†</sup>	43	1.55	0.4	0.33	1	21.5	60.5	2.8
C79914 <sup>†</sup>	50	1.51	0.4	0.3	1	20.1	52	2.6
C79915 <sup>†</sup>	52	1.6	0.4	0.25	1	15.8	57.7	3.7

**Table S1.** Continuation.

C79916 <sup>†</sup>	50	1.89	0.3	0.22	1	11.8	44	3.7
C79917 <sup>†</sup>	45	2.18	0.2	0.17	2	7.9	35.6	4.5
C79918 <sup>†</sup>	50	2.31	0.3	0.25	1	10.8	38	3.5
C79919 <sup>†</sup>	46	2.82	0.2	0.19	2	6.8	30.4	4.5
C79922 <sup>†</sup>	44	1.63	0.4	0.26	1	16.1	47.7	3
C79923 <sup>†</sup>	46	1.64	0.3	0.24	1	14.5	41.3	2.9
C79929 <sup>†</sup>	47	1.07	0.5	0.24	1	22.6	68.1	3
C79930 <sup>†</sup>	47	1.39	0.4	0.22	1	16.1	53.2	3.3
C79947 <sup>†</sup>	50	2.51	0.2	0.14	1	5.5	32	5.8
C111768 <sup>‡</sup>	43	1.38	0.4	0.2	1	14.2	62.8	4.4
C111770 <sup>‡</sup>	32	1.89	0.3	0.23	1	11.9	43.8	3.7
C111771 <sup>‡</sup>	34	2.57	0.2	0.21	2	8.1	41.2	5.1
C111772 <sup>‡</sup>	51	2.92	0.2	0.21	2	7.2	41.2	5.8
C111774 <sup>‡</sup>	25	1.78	0.3	0.24	1	13.4	56	4.2
C111777 <sup>‡</sup>	48	3.2	0.1	0.1	2	3.2	18.8	5.8
C111786 <sup>‡</sup>	45	2.46	0.2	0.15	1	6	40	6.6
C111787 <sup>‡</sup>	53	1.5	0.4	0.27	2	18.1	62.3	3.4
C111788 <sup>‡</sup>	41	1.41	0.4	0.3	1	21	58.5	2.8
C79902 <sup>‡</sup>	37	1.42	0.4	0.2	1	13.7	62.2	4.5
C79904 <sup>‡</sup>	50	1.18	0.5	0.32	1	27.5	56	2
C79905 <sup>‡</sup>	43	1.31	0.4	0.24	1	18.1	60.5	3.3
C79907 <sup>‡</sup>	50	2.02	0.3	0.28	2	13.9	56	4

**Table S1.** Continuation.

C79909 <sup>‡</sup>	41	1.88	0.3	0.19	1	10.1	46.3	4.6
C79910 <sup>‡</sup>	46	1.95	0.3	0.21	1	10.9	50	4.6
C79916 <sup>‡</sup>	34	3.7	0.1	0.12	1	3.1	23.5	7.5
C79919 <sup>‡</sup>	45	1.67	0.3	0.19	1	11.2	44.4	4
C79930 <sup>‡</sup>	40	1.96	0.3	0.28	2	14.2	50	3.5

<sup>‡</sup>Season 2019–2020.

<sup>§</sup>Percent territory = proportion contained in core areas.

<sup>#</sup>Percent use = proportion of locations in core areas.

<sup>††</sup>Intensity = ratio between percent use and percent territory.

**Table S2.** Number of birds captured, banded, and re-sighted per breeding season (2018–2022) at two grasslands areas in Southern Brazil. C: capture; X: sighting.

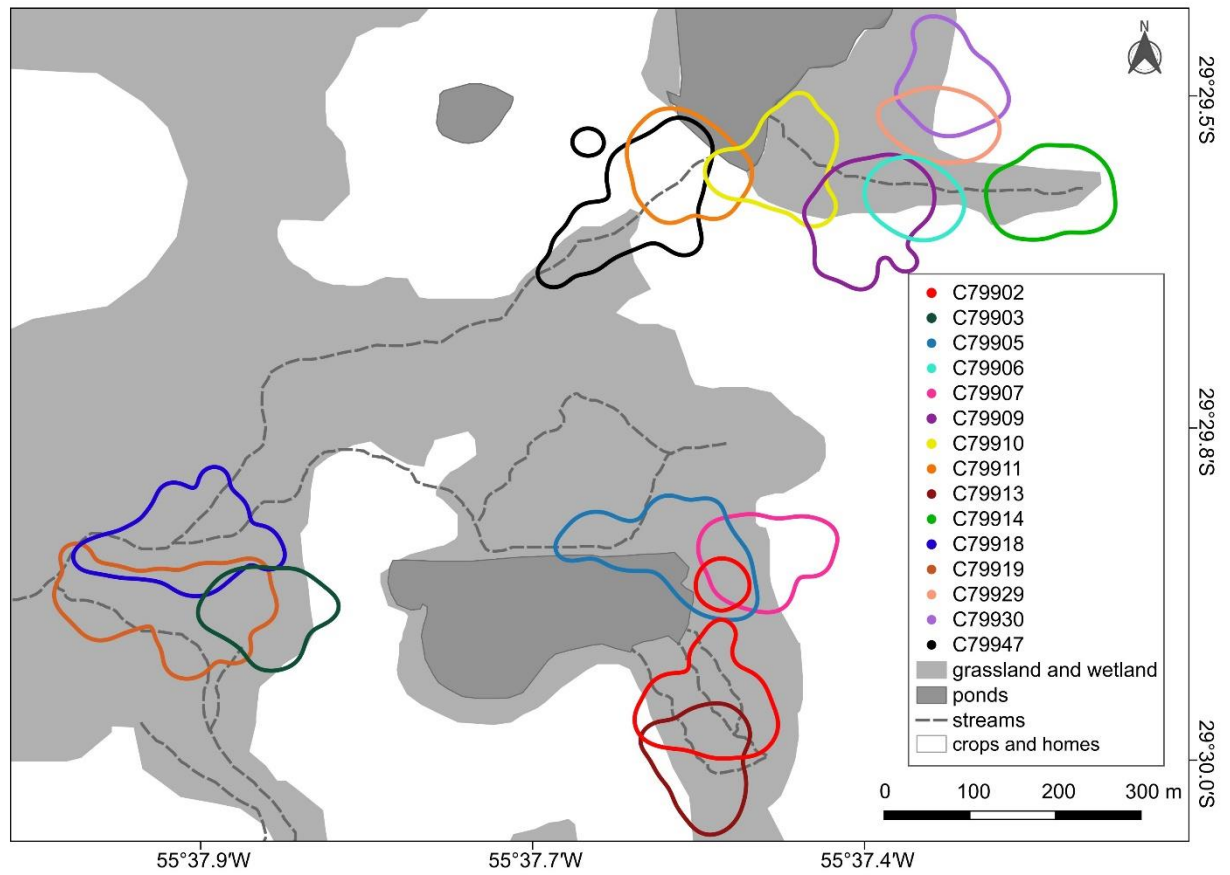
Band number	Sex	2018–2019	2019–2020	2020–2021	2021–2022
C79902	M	C	X	X	
C79903	M	C	X	X	
C79904	M	C	X	X	X
C79905	M	C	X	X	
C79906	M	C	X		
C79907	M	C	X	X	X
C79908	F	C	X	X	X
C79909	M	C	X	X	X
C79910	M	C	X	X	X
C79911	M	C			
C79912	F	C		X	
C79913	M	C	X		
C79914	M	C			
C79915	M	C	X		
C79916	M	C	X		
C79917	M	C			
C79918	M	C			
C79919	M	C	X		
C79922	M	C			
C79923	M	C			X
C79929	M	C		X	X
C79930	M	C	X	X	X
C79933	M	C	X	X	X
C79934	M	C			
C79947	M	C	X		
C79948	M	C	X		X
C79949	M	C	X		X
C111757	M	C	X		
C111761	F	C			
C111762	M	C			
C111768	M		C	X	
C111769	M		C		
C111770	M		C	X	
C111771	M		C		
C111772	M		C	X	
C111773	M		C		
C111774	M		C		
C111775	F		C		
C111776	M		C		
C111777	M		C	X	
C111778	M		C	X	X
C111779	M		C		X



**Table S2.** Continuation.

C111780	M	C		
C111781	M	C	X	X
C111783	F	C		
C111785	M	C		
C111786	M	C	X	X
C111787	M	C		
C111788	M	C		
C111789	M	C		
C111791	M	C		
C111792	M	C		
C111793	M	C	X	
C111796	F	C		
C111797	M	C		
C111798	M	C		
C111799	M	C		
C57804	F	C		
C57813	M	C	X	X
C57814	F	C		
C57819	M	C		
C110451	M		C	
C110452	M		C	
C110453	F		C	
C110454	M		C	X
C110455	M		C	
C110456	M		C	
C110457	M		C	X
C57820	M		C	
C57821	M		C	X

**Fig. S1.** Kernel contour of the estimated breeding territory (95th isolines shown) and recorded locations of 15 adult male Chestnut Seedeaters (*Sporophila cinnamomea*) during the breeding season (2018–2019) in the grasslands of Rio Grande do Sul, in the Pampas grasslands of southern Brazil.



## Appendix 2

### Territoriality, breeding philopatry, and nest site selection of Chestnut Seedeater (*Sporophila cinnamomea*) in grasslands of southern Brazil

```
##### Home Ranges and UD estimators #####
# Set your working directory
setwd("<path to chosen working directory")
# Load the required libraries
library (adehabitatHR)
library (sp)
library (mapview)
library (maptools)
# Load database
xy_1819_utm <- read.csv ('2018_2019_utm.csv', header=T, sep=",")
xy_1920_utm <- read.csv ('2019_2020_utm.csv', header=T, sep=",")
xy_1819_dec <- read.csv ('2018_2019_dec.csv', header=T, sep=",")
xy_1920_dec <- read.csv ('2019_2020_dec.csv', header=T, sep=",")
# Validating numbers as coordinates
coordinates (xy_1819_utm) <- ~ y + x # It corresponds to the column name
coordinates (xy_1920_utm) <- ~ y + x
# Adding a Coordinate Reference System - SRC
proj4string (xy_1819_utm) <- CRS ("+proj=utm +zone=21 +south +ellps=GRS80")
proj4string (xy_1920_utm) <- CRS ("+proj=utm +zone=21 +south +ellps=GRS80")
##### Validating lat long #####
##### 2018/2019 #####
coordinates (xy_1819_dec) <- ~ Long + Lat # Changing the position of the columns
# Adding a Coordinate Reference System - SRC
proj4string (xy_1819_dec) <- CRS ("+proj=longlat +ellps=GRS80
+towgs84=0,0,0,0,0,0,0 +no_defs") #SIRGAS 2000
# Calling an object
xy_1819_dec
##### 2019/2020 #####
coordinates (xy_1920_dec) <- ~ Long + Lat # Changing the position of the columns.
# Adding a Coordinate Reference System - SRC
proj4string (xy_1920_dec) <- CRS ("+proj=longlat +ellps=GRS80
+towgs84=0,0,0,0,0,0,0 +no_defs") #SIRGAS 2000
```

```

# Calling an object
xy_1920_dec
##### Kernel Density Estimation KDE #####
##### First, to build the KDE with smoothing factor href para 99%, 95% and 50% with
contour
##### isolines using the kernelUD() and getverticeshr() function
#####
##### Breeding season 2018-2019 #####
#####
# The href was previously used to generate all reference bandwidth, and after obtaining the
# parameter's value. The median was obtained to be used as the "population smoothing
# parameter."
Khref <- kernelUD(xy_1819_utm, h = 15.92, grid = 800, extent = 2.2)
Khref10 <- getverticeshr(Khref, 10, unin = "m", unout = "ha")
Khref20 <- getverticeshr(Khref, 20, unin = "m", unout = "ha")
Khref30 <- getverticeshr(Khref, 30, unin = "m", unout = "ha")
Khref40 <- getverticeshr(Khref, 40, unin = "m", unout = "ha")
Khref50 <- getverticeshr(Khref, 50, unin = "m", unout = "ha")
Khref60 <- getverticeshr(Khref, 60, unin = "m", unout = "ha")
Khref70 <- getverticeshr(Khref, 70, unin = "m", unout = "ha")
Khref80 <- getverticeshr(Khref, 80, unin = "m", unout = "ha")
Khref90 <- getverticeshr(Khref, 90, unin = "m", unout = "ha")
Khref95 <- getverticeshr(Khref, 95, unin = "m", unout = "ha")
# Finding the value of the parameter used for each animal
Khref[["name"]@h
# Save value in .csv
write.csv(as.data.frame(Khref10), 'namespecies_10 2018_2019.csv')
write.csv(as.data.frame(Khref20), 'namespecies_20 2018_2019.csv')
write.csv(as.data.frame(Khref30), 'namespecies_30 2018_2019.csv')
write.csv(as.data.frame(Khref40), 'namespecies_40 2018_2019.csv')
write.csv(as.data.frame(Khref50), 'namespecies_50 2018_2019.csv')
write.csv(as.data.frame(Khref60), 'namespecies_60 2018_2019.csv')
write.csv(as.data.frame(Khref70), 'namespecies_70 2018_2019.csv')
write.csv(as.data.frame(Khref80), 'namespecies_80 2018_2019.csv')
write.csv(as.data.frame(Khref90), 'namespecies_90 2018_2019.csv')
write.csv(as.data.frame(Khref95), 'namespecies_95 2018_2019.csv')
# Obtaining the areas for each of the calculated values

```

```

kernelareashref_10_90 <- kernel.area (Khref, percent = seq (10,90, by=10),
                                     unin = "m", unout = "ha")
# Obtaining the area for 95% calculated values
kernelareashref_95 <- kernel.area (Khref, percent = c(95), unin = "m", unout = "ha")
# Save value in .csv
write.csv (as.data.frame (kernelareashref_10_90), 'kernelareashref 2018_2019_10_90.csv')
# Save value in .csv
write.csv (as.data.frame (kernelareashref_95), 'kernelareashref 2018_2019_95.csv')
# Plotting the limits for each percentage
mapview (Khref95) + mapview (Khref90) + mapview (Khref80) + mapview (Khref70) +
mapview (Khref60) + mapview (Khref50) + mapview (Khref40) + mapview (Khref30) +
mapview (Khref20) + mapview (Khref10) + mapview (xy_1819_utm)
# Save shape file
writePolyShape (Khref95,'Khref95 2018_2019')
writePolyShape (Khref90,'Khref90 2018_2019')
writePolyShape (Khref80,'Khref80 2018_2019')
writePolyShape (Khref70,'Khref70 2018_2019')
writePolyShape (Khref60,'Khref60 2018_2019')
writePolyShape (Khref50,'Khref50 2018_2019')
writePolyShape (Khref40,'Khref40 2018_2019')
writePolyShape (Khref30,'Khref30 2018_2019')
writePolyShape (Khref20,'Khref20 2018_2019')
writePolyShape (Khref10,'Khref10 2018_2019')
# Counting the number of points per estimated polygon
ndpontos <- over (xy_1819_utm, Khref10) # 10 to 95
table (ndpontos$id)
# Overview the lines in different % and cores
mapview (Khref95, col.regions = "red", map.type = "OpenStreetMap") + mapview (Khref90,
col.regions = "cyan1") + mapview (Khref80, col.regions = "blue") + mapview (Khref70,
col.regions = "green") + mapview (Khref60, col.regions = "yellow") + mapview (Khref50,
col.regions = "pink") + mapview (Khref40, col.regions = "orange") + mapview (Khref30,
col.regions = "white") + mapview (Khref20, col.regions = "grey") + mapview (Khref10,
col.regions = "chocolate1") + mapview (xy_1819_utm, alpha = 0.1, cex = 3)
#####
##### Breeding season 2019-2020 #####
#####
Khref <- kernelUD (xy_1920_utm, h = 15.92, grid = 800, extent = 2.2)

```

```

Khref10 <- getverticeshr (Khref, 10, unin = "m", unout = "ha")
Khref20 <- getverticeshr (Khref, 20, unin = "m", unout = "ha")
Khref30 <- getverticeshr (Khref, 30, unin = "m", unout = "ha")
Khref40 <- getverticeshr (Khref, 40, unin = "m", unout = "ha")
Khref50 <- getverticeshr (Khref, 50, unin = "m", unout = "ha")
Khref60 <- getverticeshr (Khref, 60, unin = "m", unout = "ha")
Khref70 <- getverticeshr (Khref, 70, unin = "m", unout = "ha")
Khref80 <- getverticeshr (Khref, 80, unin = "m", unout = "ha")
Khref90 <- getverticeshr (Khref, 90, unin = "m", unout = "ha")
Khref95 <- getverticeshr (Khref, 95, unin = "m", unout = "ha")

# Finding the value of the parameter used for each animal
Khref[["name"]@h

# Save value in .csv
write.csv (as.data.frame (Khref10), 'namespecies_Khref10 2019_2020.csv')
write.csv (as.data.frame (Khref20), 'namespecies_Khref20 2019_2020.csv')
write.csv (as.data.frame (Khref30), 'namespecies_Khref30 2019_2020.csv')
write.csv (as.data.frame (Khref40), 'namespecies_Khref40 2019_2020.csv')
write.csv (as.data.frame (Khref50), 'namespecies_Khref50 2019_2020.csv')
write.csv (as.data.frame (Khref60), 'namespecies_Khref60 2019_2020.csv')
write.csv (as.data.frame (Khref70), 'namespecies_Khref70 2019_2020.csv')
write.csv (as.data.frame (Khref80), 'namespecies_Khref80 2019_2020.csv')
write.csv (as.data.frame (Khref90), 'namespecies_Khref90 2019_2020.csv')
write.csv (as.data.frame (Khref95), 'namespecies_Khref95 2019_2020.csv')

# Obtaining the areas for each of the calculated values
kernelareashref_10_90 <- kernel.area (Khref, percent = seq (10,90, by=10),
                                     unin = "m", unout = "ha")

# Obtaining the area for 95% calculated values
kernelareashref_95 <- kernel.area (Khref, percent = c(95), unin = "m", unout = "ha")

# Save value in .csv
write.csv (as.data.frame (kernelareashref_10_90),
          'kernelareashref 2019_2020_10_90.csv')

# Save value in .csv
write.csv (as.data.frame (kernelareashref_95),
          'kernelareashref 2019_2020_95.csv')

# Plotting the limits for each percentage

```

```
mapview (Khref95) + mapview (Khref90) + mapview (Khref80) + mapview (Khref70) +  
mapview (Khref60) + mapview (Khref50) + mapview (Khref40) + mapview (Khref30) +  
mapview (Khref20) + mapview (Khref10) + mapview (xy_1920_utm)  
# Save shape file  
writePolyShape (Khref95,'Khref95 2019_2020')  
writePolyShape (Khref90,'Khref90 2019_2020')  
writePolyShape (Khref80,'Khref80 2019_2020')  
writePolyShape (Khref70,'Khref70 2019_2020')  
writePolyShape (Khref60,'Khref60 2019_2020')  
writePolyShape (Khref50,'Khref50 2019_2020')  
writePolyShape (Khref40,'Khref40 2019_2020')  
writePolyShape (Khref30,'Khref30 2019_2020')  
writePolyShape (Khref20,'Khref20 2019_2020')  
writePolyShape (Khref10,'Khref10 2019_2020')  
# Counting the number of points per estimated polygon  
ndpontos <- over (xy_1920_utm, Khref40)  
table (ndpontos$id)  
# Overview the lines in different % and cores  
mapview (Khref95, col.regions = "red", map.type = "OpenStreetMap") + mapview (Khref90,  
col.regions = "cyan1") + mapview (Khref80, col.regions = "blue") + mapview (Khref70,  
col.regions = "green") + mapview (Khref60, col.regions = "yellow") + mapview (Khref50,  
col.regions = "pink") + mapview (Khref40, col.regions = "orange") + mapview (Khref30,  
col.regions = "white") + mapview (Khref20, col.regions = "grey") + mapview (Khref10,  
col.regions = "chocolate1") + mapview (xy_1920_utm, alpha = 0.1, cex = 3)
```