



Effects of cattle on habitat use and diel activity of large native herbivores in a South American rangeland

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ABSTRACT

Cattle grazing usually has negative effects on wildlife. We studied the effects of cattle on the patterns of habitat use and diel activity of three neotropical herbivores, capybara (*Hydrochoerus hydrochaeris*), gray brocket (*Mazama gouazoubira*) and marsh deer (*Blastocerus dichotomus*) across the savannas of northeastern Argentina. We conducted a camera-trap survey with 234 stations in three habitat types (grasslands, forests, pine plantations) with and without cattle. We used occupancy models to evaluate the effect of cattle (presence or recording rate), season (winter vs spring-summer, and its interaction with cattle), distance to water bodies, and density of vegetation, on the probability of detection (p) of the native herbivores. We also evaluated the effect of cattle, habitat type, structural complexity of the vegetation, proportion of marshlands and forests in the landscape, season, and relative accessibility by humans, on the probability of occupancy (ψ) of these species. Using kernel density functions, we estimated the overlap in diel activity pattern of native herbivores with that of cattle, comparing this overlap between stations with and without cattle. Several variables, but not cattle, affected ψ . For marsh deer p was lower when cattle were present. For brocket, p was much lower in winter when cattle were present. For the marsh deer and the capybara, ψ increased (but in brocket decreased) with increasing cost of human access, a proxy of hunting. For capybara and brocket, ψ increased with increasing structural complexity of vegetation. Cattle were more diurnal in grasslands than in forests and plantations. Native herbivores became more nocturnal, with lesser overlap with the diel pattern of cattle, in grasslands, when cattle were present. Cattle seem to exert interference competition on the native herbivores which used less frequently areas where cattle were present and avoided being simultaneously active with grazing cattle. Livestock production should strive to mitigate the negative competitive effects of cattle on wildlife by maintaining cattle free protected areas. It should also reduce some indirect effects of this activity on the native herbivores (e.g., hunting of wildlife and frequent burning of grasslands).

1. Introduction

Livestock grazing affects more than 25 % of the earth surface, about half of the grasslands and savannas, and have pervasive impacts on ecosystem processes and biodiversity (Bernardi, Staal, Xu, Scheffer, & Holmgren, 2019; MEA, 2005; Reid et al., 2013; Robinson et al., 2014). In the next few years, cattle grazing is predicted to increase in South

America, where two of the main beef producers are located (Brazil and Argentina; FAO, 2020), and where forests (e.g., the Amazon) are being cleared for cattle grazing (Alkemade, Reid, van den Berg, de Leeuw, & Jeuken, 2013; Reid et al., 2013). Under certain grazing regimes, livestock may have some positive effects on ecological processes and biodiversity, and it can sometimes be used to promote ecosystem or landscape restoration, especially in ecosystems with a long history of

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human use (Ba Diao, 2020; FAO, 2009; Verdú, Crespo, & Galante, 2000, 2007). Thus, when considering the impacts of cattle ranching on nature, negative (e.g., Pringle, Young, Rubenstein, & McCauley, 2007; Young et al., 2018) and positive (Allan et al., 2017; Keesing et al., 2018) effects of livestock production should be weighted. Even though livestock grazing may sometimes have positive (i.e. facilitation) effects on other large herbivores (Augustine, 2004; Willms, McLean, Tucker, & Ritcey, 1979; Yeo, Peek, Wittinger, & Kvale, 1993), negative (i.e., competitive) effects seem to be more common (Schielz & Rubenstein, 2016), especially when productivity is reduced (e.g., during a dry season), and at high stocking rates (Kinnaid & O'Brien, 2012; Odadi, Karachi, Abdulrazak, & Young, 2011; Young et al., 2018). Consequently, the global increase in livestock production and, particularly, of cattle, has been associated with a dramatic reduction of the biomass of large herbivores worldwide (Ripple et al., 2015).

A thorough review of the literature on the competitive effects of livestock on wildlife found that only three studies on the effect of livestock on ungulates have been conducted in South America (Schielz & Rubenstein, 2016). These three studies (Baldi, Albon, & Elston, 2001; Borgnia, Vilá, & Cassini, 2008; Burgi, Marino, Rodríguez, Pazos, & Baldi, 2011), found negative effects of livestock on native camelids. In our review of the literature, we found six other articles on the competitive effect of livestock on habitat use by large native herbivores in South America, three on the pampas deer, *Ozotoceros bezoarticus* (Bilenca et al., 2012; Perez Carusi, Beade, & Bilenca, 2017; Vila, Beade, & Barrios Lamunière, 2008), and three on the gray brocket, *Mazama gouazoubira* (Nanni, 2015; Noss & Cuéllar, 2000; Puechagut et al., 2018), all showing avoidance of livestock by these cervids. However, there seems to be no information on the effect of cattle on other large Neotropical herbivores. Given the extent and impact of cattle grazing in Latin America, it is very important to better understand how large herbivores respond to the presence of livestock in South American rangelands.

The potential competitive effects of cattle on large herbivores depend on their body size (Illius & Gordon, 1992), the proportional consumption of grasses vs other vegetation (Desbiez, Santos, Alvarez, & Tomas, 2011), and the type of digestive system (foregut fermenters or ruminants vs hindgut fermenters, Illius & Gordon, 1992; Schielz & Rubenstein, 2016). Based on these characteristics, it is possible to predict if pairs of species will compete for food when the conditions for this interaction occur (Butt & Turner, 2012; Illius & Gordon, 1992). As cattle (*Bos taurus*) are large ruminant grazers, herbivores with these characteristics are expected to compete with them. In South American grasslands there are no grazers morphologically similar to cattle, but browsers (e.g., cervids) could be as negatively affected by cattle as grazers, probably as a result of the latter heavy consumption of protein-rich forbs (Young et al., 2018), or by interference, rather than exploitative competition (Schielz & Rubenstein, 2016).

The effects of interspecific competition can be observed in different behavioral and ecological aspects. Competition may eventually lead to changes in density, habitat use, or to the exclusion of the lower competitor from certain areas (Perez Carusi et al., 2017; Stewart, Bowyer, Kie, Cimon, & Johnson, 2002). The coexistence of syntopic species with similar diets could be facilitated by a reduction in resource use overlap along ecological niche axes (Griffin & Silliman, 2011; Schoener, 1974). This resource partitioning has been usually studied along three axes representing trophic niche (diet), habitat use, and diel activity (Begon, Harper, & Townsend, 1990). It has been suggested that resource partitioning will be more frequent in the diet and habitat use axes (Schoener, 1974). However, when competition is of the interference type and the lower competitor is cathemeral, a shift in its diel activity pattern is a valid mechanism to reduce competition (Carothers & Jaksic, 1984; Di Bitetti, Di Blanco, Pereira, Paviolo, & Jiménez Pérez, 2009).

Cattle can not only compete with wildlife for resources but can modify the environment (e.g., through grazing or trampling) and affect habitat conditions (Eldridge, Poore, Ruiz-Colmenero, Letnic, &

Soliveres, 2016). Vegetation structure and composition can be simplified, not only through the physical effects exerted by cattle but by the use of fires to facilitate forage regrowth (Cingolani et al., 2013), or the implantation of potentially invasive exotic pastures (D'Antonio & Vitousek, 1992). Cattle management also implies the presence of people, sometimes accompanied by dogs, which may hunt or harass wildlife and particularly seek large herbivores as preferred prey (Jerzolimski & Peres, 2003). These indirect effects of cattle raising can represent a good proportion of the negative effects of livestock production on native herbivores.

Our research aimed at studying the spatial and temporal responses of three large native herbivores, the capybara (*Hydrochoerus hydrochaeris*), the gray brocket deer, and the marsh deer (*Blastoceros dichotomus*) to the presence of cattle in the subtropical savannas of northeastern Argentina. We used occupancy models to study their patterns of habitat use in relation to cattle presence. We evaluated the potential temporal avoidance of cattle by comparing the diel activity patterns of these herbivores when livestock was present vs absent. If conditions for food competition prevail (i.e., during the winter), we expected a potentially stronger competitive effect of cattle on capybaras, which consume mostly grasses (Desbiez et al., 2011; Quintana, Monge, & Malvárez, 1998; Quintana, 2003), than on the two cervids, characterized as browsers or selective feeders (Black-Décima et al., 2010; Marin et al., 2020; Piovezan et al., 2010). Between the two cervids, we expected a potentially stronger negative effect of cattle on marsh deer than on gray brocket, due to the larger body weight of the former and its higher overlap in grassland use with livestock. We also predicted indirect negative effects of cattle management through changes in vegetation structure.

2. Materials and methods

2.1. Study area

The study was conducted in northeastern Corrientes province and southern Misiones province, Argentina, comprising an area of about 25,000 km² (Fig. 1). Two ecoregions are included within the study area, the Southern Cone Mesopotamian Savannas and the Iberá Marshes ecoregions (Brown & Pacheco, 2006; Burkart, Bárbaro, Sánchez, & Gómez, 1999). The weather is subtropical without a dry period, with an average annual precipitation of 1400–1800 mm and more rainfall during the summer (Neiff & Neiff, 2013; Sampedro & Calvi, 2018). Frequent frosts occur during the winter (May–September) when forage productivity and quality decreases (Orozco, Marull, Jiménez, & Gürtler, 2013; Pizzio & Bendersky, 2018). The study area is dominated by subtropical savannas, temporarily flooded grasslands and wetlands, and includes other natural environments such as woodlands and gallery forests, and lands devoted to crop production (Fig. 1). Cattle ranching on both natural and introduced grasses, and tree plantations (*Pinus* sp. and *Eucalyptus* sp.) are the main productive activities. Cattle are sometimes produced under low-density tree plantations in silvopastoral systems. The area also includes protected areas under national, provincial or private jurisdiction where productive activities, except tourism, are not allowed. People and, probably, dogs are the main predators of adult large herbivores in the study area, since pumas (*Puma concolor*) are rare (Iezzi, De Angelo, & Di Bitetti, 2020; Soler & Cáceres, 2009) and jaguars (*Panthera onca*) became locally extinct (Caruso & Jiménez Pérez, 2013). Consequently, populations of capybaras and marsh deer in strictly protected areas are regulated by bottom-up forces and parasites, and have higher infection and mortality rates during the winter (Gorosábel, Loponte, & Corriale, 2016; Orozco et al., 2013).

2.2. Studied species

We focus this study on the effects of cattle on a large rodent, the capybara, and two native cervids, the gray brocket and the marsh deer. Due to the low number of camera-trap records of other large herbivores,

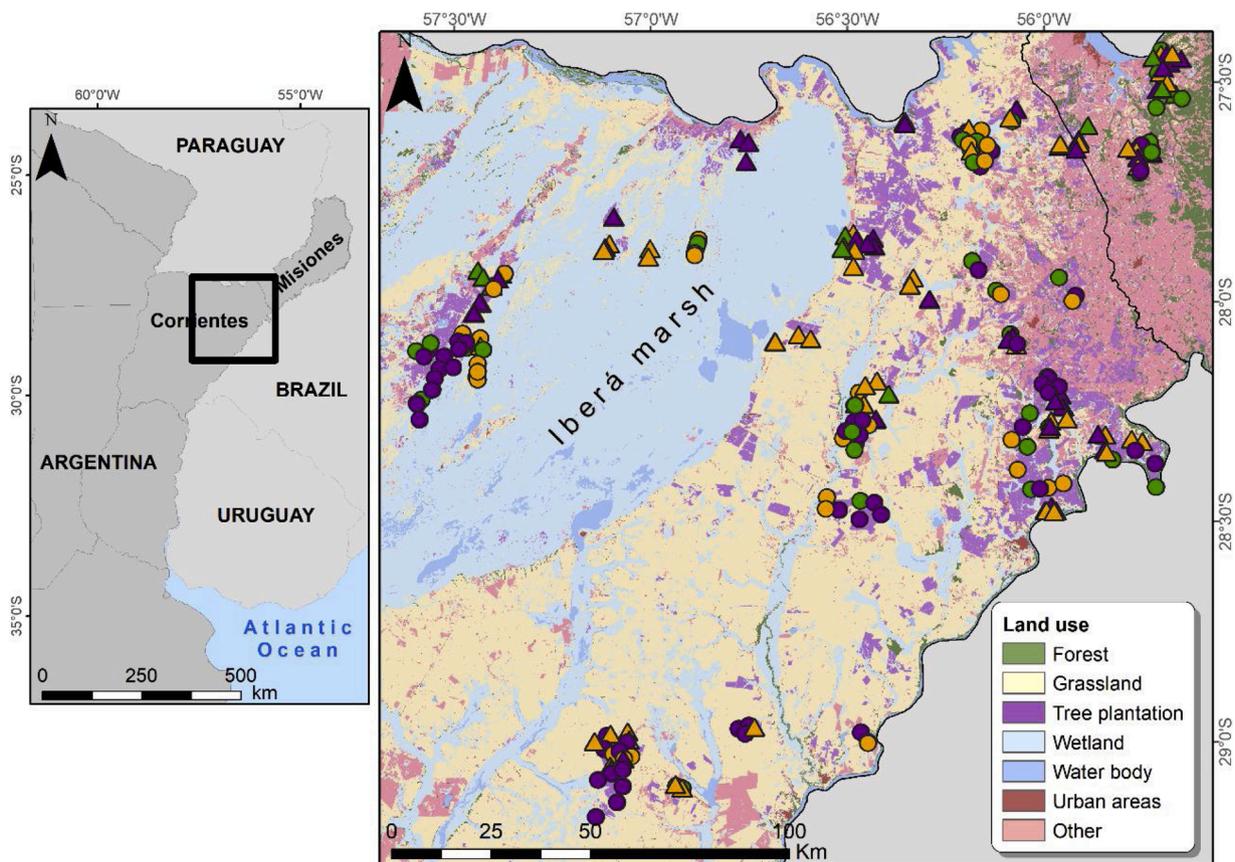


Fig. 1. Location of the study area in Northeastern Argentina and map showing the different main habitat types and land uses, and the location of the camera-trap stations. Green dots correspond to camera-trap stations located in forests, orange dots to those located in grasslands, and purple ones to those located in pine plantations. Circles correspond to stations without cattle and triangles to those with cattle.

including the pampas deer (*Ozotoceros bezoarticus*) and the exotic chital (*Axis axis*), they were not included in the study (for a complete list of recorded species see [Iezzi et al., 2020](#)).

The capybara is a large South American rodent (35–65 kg) associated with aquatic environments such as rivers, marshes, and lagoons ([Lacher, 2016](#)). It has a diet based on grasses, sedges, and aquatic vegetation which it grazes in the borders of water bodies ([Desbiez et al., 2011](#); [Quintana et al., 1998](#); [Quintana, 2003](#)). The capybara is a social polygynous mammal living in groups of up to 30 individuals. Dominant males defend territories of 5–200 ha along water bodies ([Corriale, Muschetto, & Herrera, 2013](#); [Herrera & Macdonald, 1989](#)). Capybaras are usually more active at dawn and dusk but, being cathemeral, they may become mostly nocturnal when they are persecuted ([Lacher, 2016](#)). Both globally and in Argentina, the conservation status of this species is categorized as low concern ([Bolkovic et al., 2019](#)).

The gray brocket is a small cervid (11–25 kg) that occupies open forest formations and wooded savannas in South America, with a preference for borders between grasslands and forests ([Black-Décima et al., 2010](#); [Mattioli, 2011](#)). It is a typical browser with a diet composed of leaves, twigs, flowers, buds, and fruits ([Black-Décima et al., 2010](#)). It is a mostly solitary and cathemeral species that occupies territories of 30–348 ha ([Black-Décima et al., 2010](#); [Mattioli, 2011](#)). The gray brocket is categorized as of low concern both globally and in Argentina ([Juliá et al., 2019](#)).

The marsh deer is a large cervid (70–130 kg) that inhabits marshes and seasonally flooded grasslands south of the Amazon river ([Mattioli, 2011](#); [Piovezan et al., 2010](#)). It has been characterized as a grazer-browser or as a concentrate selector that consumes grasses and easily digestible food, such as aquatic and semiaquatic plants and browse ([Beccaceci, 1996](#); [Marin et al., 2020](#); [Tomas & Salis, 2000](#)). It is

cathemeral and weakly gregarious, with solitary males but females sometimes forming small family groups ([Mattioli, 2011](#)). Marsh deer occupy relatively large home ranges, varying from 300 to up to 5000 ha ([Mattioli, 2011](#); [Piovezan et al., 2010](#)). It is a threatened species categorized as vulnerable both globally and in Argentina ([Pereira et al., 2019](#)).

Cattle were the dominant livestock in the study area, with 2807 independent camera-trap records at 100 camera-trap sites. Most herds belonged to Braford and Brangus breeds (hybrids between *Bos taurus taurus* and *Bos taurus indicus*). Stock densities vary between 0.8 and 1.5 individuals/ha and, during the winter, even at moderate stocking rates, cattle may lose weight and their body condition deteriorate as a result of forage shortage ([Sampedro, 2018](#)). Cattle are kept in large enclosures of usually >50 ha frequently containing small forest patches and, occasionally, pine plantations. Other livestock present in the study area included horses (*Equus ferus caballus*, 244 records), water buffalos (*Bubalus arnee bubalis*, 36 records), sheep (*Ovis aries*, 7 records), and goats (*Capra aegagrus hircus*, 4 records). Due to similarities in diet, digestive system and body weight, records of water buffalo were considered as cattle in the analyses.

2.3. Camera-trap surveys

We conducted a 9.5-month-long camera-trap survey between 27 May 2016 and 10 March 2017. We deployed 234 non-baited camera-trap stations, each represented by a single Reconyx® model HC500 unit. Camera-traps were located at distances of >50 m from dirt roads or human trails. Each unit was attached to a wooden stake or to the base of a tree at a height of about 30–40 cm above ground level. The location of stations was decided opportunistically (depending on permits and

accessibility) but following a balanced design among three a priori selected habitat types and considering the proviso that stations located in the same habitat type should be at least 2 km apart. Camera-trap stations were deployed in three habitat types: native grassland (89 stations; 50 with cattle records), native forest (54 stations; 17 with cattle records), and pine plantations (91 stations, 36 with cattle records, Fig. 1). Stations located in pine plantations were immersed in 1- to 30-year-old plantation stands with densities ranging from 200 to 1666 trees/ha. Camera-traps were set to have a delay of 30 s between detections and to take 10 successive pictures per trigger. Stations were active, on average, for 44.9 consecutive days (range: 21–67 days), totaling a camera-trap effort of 10,494 days. Records were considered an independent event if >1 h had passed between successive pictures of the same species.

2.4. Occupancy modeling

To analyze the patterns of habitat use by the three native herbivores we used single-species single-season occupancy models, based on a likelihood approach, that were run in package *unmarked* (Fiske & Chandler, 2011) in *R* (version 3.5). These models estimate the probability that a site is being used (occupied) by a species (ψ), and the probability of detecting the species (p) at each camera-trap station when the station is being used (MacKenzie et al., 2002). Given the relatively small size of the area effectively surveyed by a camera trap in relation to the size of the home range of the studied species, ψ should be interpreted as the probability that the camera-trap station is located within the home range of at least one individual of the target species (Efford & Dawson, 2012; Neilson, Avgar, Burton, Broadley, & Boutin, 2018), which corresponds to Johnson's (1980) second-order habitat selection. Since the target species have relatively large home ranges and can move freely in and out of the surveyed area, the parameter ψ , even though usually referred to as the probability of occupancy, in camera-trap studies should be more properly interpreted as the probability that the species use the area rather than as the species being continuously present in that habitat patch (MacKenzie, 2005). The probability of detection (p) is estimated by the pattern of repeated visits to the surveying area by a target species. The time frame during which a visit can occur is defined as a sampling occasion (MacKenzie et al., 2006). For this analysis, we divided the sampling period in up to nine successive sampling occasions of seven consecutive days each. The probability of detection can be affected by several factors, of which animal density and activity are biologically relevant (Neilson et al., 2018; Rowcliffe, Field, Turvey, & Carbone, 2008), and are related to Johnson's (1980) third-order habitat selection.

We evaluated the effect on p , ψ or both, of several covariates. We predicted the effect of these variables on p or ψ based on specific hypotheses aimed at explaining the patterns of habitat use by large herbivores, including the potential competitive effect of cattle or its management (Table A.1). Some of the variables were measured *in situ* and some others in a Geographic Information System (GIS, using ArcGIS 10.3.1, ESRI Inc., Table A.1), for which we used a land-use layer (pixels of 30 × 30 m) created for the period 2013–2014 by Zuleta et al. (2015).

Four variables were tested for their potential effect on p (see details in Table A.1). 1) Cattle. To assess the effect of cattle on p we used either the presence/absence of cattle or its camera-trap recording rate (# of independent photographic records of cattle per day). The latter was used as an index of cattle relative abundance, even though we were not able to adjust this index to an estimate of abundance (Kinnaird & O'Brien, 2012; O'Brien, 2011). We predicted avoidance of cattle by large herbivores and thus, lower activity (lower p) in areas with cattle or as cattle recording rate increases. 2) Vegetation density. As the density of the vegetation at a camera-trap station may act as a physical barrier that obstructs the visual field, we predicted a decline of p with increasing vegetation density. This variable was estimated, with the point contact method (Catchpole & Wheeler, 1992), as the mean number of contacts of

the vegetation with a 1-m height vertically placed rod measured 12 times (see Table A.1 for details). 3) Distance to water. For capybara and marsh deer, we assessed the effect on p of the lineal distance to the nearest permanent fresh-water body due to their high dependency on water bodies for establishing their territories, mating, thermoregulation and seeking refuge from predators (Corriale & Herrera, 2013; Herrera & Macdonald, 1989). We predicted a decreasing gradient of activity and thus, of p , with increasing distance to water. 4) Season. The study area is under strong seasonality which may affect, through temporal changes in food availability, the level of activity of animals. We predicted higher p during winter due to higher step rates when forage decreases (Young et al., 2018). For this variable, we classified camera-trap stations into two groups: those active during the winter and those active during the spring-summer months (Table A.1). Finally, we assessed a potential interaction between cattle and seasonality. We predicted an increase of p during the winter, but especially so in areas without cattle. We could not model this interaction for marsh deer due to low number of records.

We assessed the effect of seven independent variables on ψ (Table A.1). 1) Cattle. If cattle negatively affect forage quality/quantity or interfere with native herbivores we predicted lower ψ in areas with cattle. 2) Habitat type. We included this variable due to known associations of these herbivores with certain habitat types (Iezzi et al., 2020; Table A.1). 3) Relative representation of keystone habitats (marshlands and forests). We predicted a positive association of ψ with the proportional representation of marshland, for capybara and marsh deer, and forests, for brockets. The percentage area of these habitat types was measured, in a GIS, in concentric circles around camera-trap stations (at radii of 200, 500, 1000, 2000 and 5000 m) in order to detect scale-dependent species responses (e.g. Nagy-Reis et al., 2017; Zurita, Pe'er, & Bellocq, 2017). 4) Distance to water. As a result of the high dependency of capybara and marsh deer on fresh-water ecosystems, the probability of habitat use should decrease with increasing lineal distance to the nearest permanent water body. 5) Structural complexity and composition of vegetation. We predicted an increase of ψ with more complex and diverse understory vegetation, which may provide refuge, concealment, or resources. We measured, *in situ*, the structural complexity and diversity of understory vegetation following the procedures described in Iezzi et al. (2020). We summarized this variable with the second axis of a principal components analysis (PCA) that discriminate stations with higher abundance of shrubs, ferns, and short grasses and forbs (positive values) from stations with scarce undergrowth and cespitose grasses (negative values, Table A.1). 6) Season. During times of food scarcity, mammals may increase the size of their home ranges (Di Bitetti, 2001; McNab, 1963). This was reported for capybaras and other mammals in our study area (Corriale et al., 2013; Di Blanco, Jiménez Pérez, & Di Bitetti, 2015), for which we predicted higher ψ during winter. We tested for an interaction between season and cattle, predicting an increase of ψ during winter but, exceptionally so, in areas without cattle. This interaction was not modelled for marsh deer for lack of records. 7) Cost of access. Through hunting, and by their association with dogs, people can have negative effects on large mammal populations (Lessa et al., 2016; Peres & Palacios, 2007). The cost of access, computed as the hours needed to access a camera-trap station from the nearest town or city, represents the probability of occurrence of people and was used as a proxy of hunting. The cost of access was estimated following Iezzi et al. (2020, Table A.1). We predicted an increase in ψ with increasing cost of access.

As a first step in the analysis, we checked for covariation between all pairs of variables affecting p and ψ by running bivariate Spearman correlations. No correlation had a coefficient >0.6 for which we included them in the models in different combinations (Burnham & Anderson, 2002). Due to the large number of variables, scales, and potential models, we followed a three-step modeling approach (Cruz, Iezzi, De Angelo, Varela, & Di Bitetti, 2019; Estevo, Nagy-Reis, & Nichols, 2017). The first step was to select the best alternative (radii) for the variable representing the percentage of marshlands (capybara and

marsh deer models) and forests (gray brocket models) in the landscape, and the option that best represents the effect of cattle (presence-absence vs camera-trap recording rate). For this, we compared the 20 models that result from the combination of the five landscape radii with the two cattle alternatives (for p and ψ) in a general model that hold present all other covariates. We selected the alternatives present in the best-ranked model. In a second step, we run all potential combinations of the six (gray brocket) or seven (capybara and marsh deer) variables that may affect ψ , in a general model that included all three (marsh deer) or four (capybara and brocket) covariates of p . Finally, we run models with all the combinations of variables that could affect p in a general model that included all the covariates of ψ selected in the previous step (see Estevo et al., 2017 for a similar three-step procedure). Models were ranked in ascending order of their Akaike's Information Criterion value adjusted for small sample size (AICc). A variable included in the top-ranked models ($\Delta AICc \leq 2$) was considered important if the 95 % confidence intervals (CI) of its beta coefficient did not include zero (Anderson, 2007). Only the variables contained in the top-ranked model ended up being important for which we did not use average modeling. We checked for overdispersion of the data in the full models using the goodness-of-fit test; and no c -hat value was >1.00 (MacKenzie & Bailey, 2004).

2.5. Diel activity patterns

To describe and compare the diel activity patterns of the native herbivores in areas with and without cattle we used circular statistics (Frey, Fisher, Burton, & Volpe, 2017). The similarity between the activity patterns of two species could be quantified with a coefficient (Δ) that describes the extent of overlap (0 = no overlap, 1 = complete overlap) of their circular kernel density functions (Frey et al., 2017; Oliveira-Santos, Zucco, & Agostinelli, 2013; Ridout & Linkie, 2009). For the three species, we estimated and compared their Δ with cattle in stations where cattle were present vs absent. To estimate Δ , we used a smoothing parameter of 1.0 and the estimator Δ_1 , recommended for small samples (Meredith & Ridout, 2018). We estimated 95 % CIs through 1,000 bootstrap iterations.

If wild herbivores were avoiding cattle, we predicted that their Δ with cattle will be lower in stations where the latter were present. Given that the diel activity pattern of cattle differed among habitat types we performed these analyses for each one separately. Because due to thermoregulatory reasons the diel activity pattern of mammals may also change with the seasons (Di Blanco, Spørring, & Di Bitetti, 2017; Maloney, Moss, Cartmell, & Mitchell, 2005), we compared the Δ of the herbivores with cattle separately for the winter (May-September) and the spring-summer (October-March). Given that the diel activity of mammals may change with the degree of anthropogenic impact

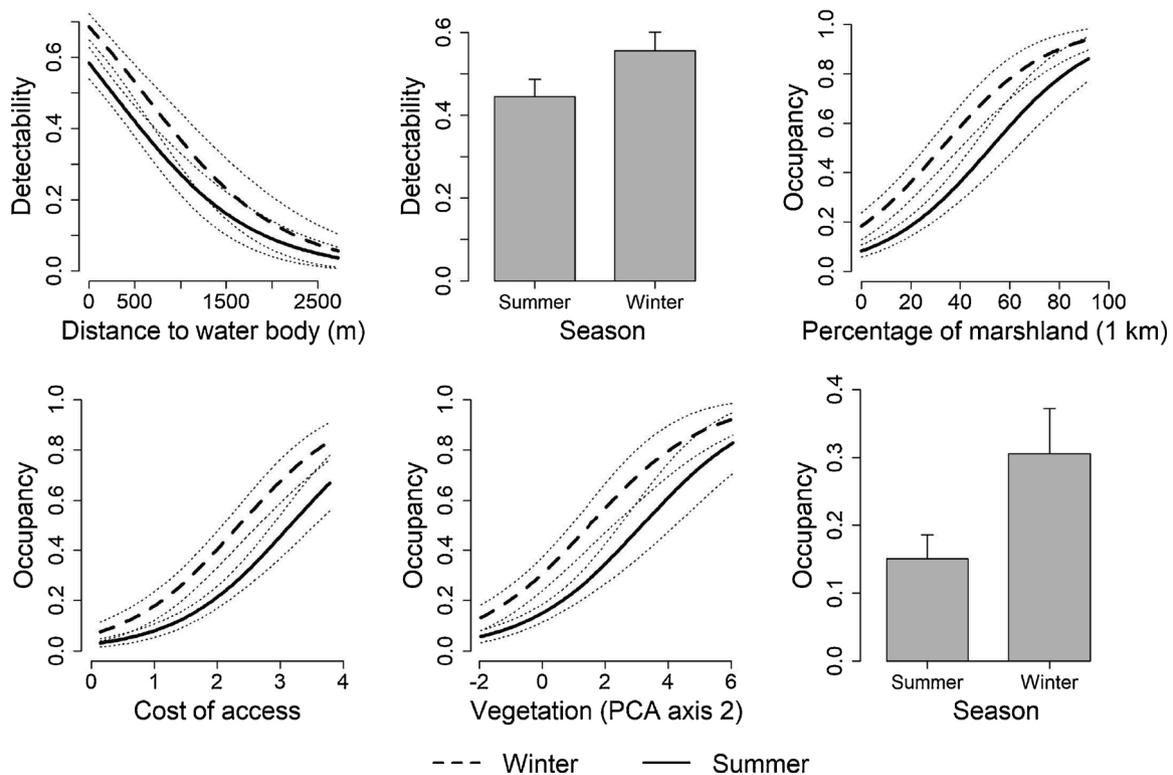


Fig. 2. Probability of occupancy and detectability of the capybara (*Hydrochoerus hydrochoeris*). Records for this occupancy analysis were obtained during a camera-trap survey conducted in the Southern Cone Mesopotamian Savannas and the Iberá marshes ecoregions of northern Corrientes and southern Misiones provinces of Argentina. Solid lines depict predicted values, broken lines and error bars represent the standard error of the mean.

(Gaynor, Hojnowski, Carter, & Brashares, 2018), we also compared the Δ of the herbivores with cattle separately in stations with low and high cost of access, using the median value as a cut-off point. For these analyses, we used packages *camtrapR* (Niedballa, Sollmann, Courtiol, & Wilting, 2016) and *overlap* (Meredith & Ridout, 2014) in R (version 3.5).

3. Results

3.1. Detectability and occupancy

For the capybara, the top-ranked model included the nearest distance to a water body and season affecting p and four variables affecting ψ : the percentage of marshland within 1 km, the cost of access, the structural complexity and diversity of vegetation, and season (Table A.2). The detectability of capybaras decreased with increasing distance to water bodies and was higher during the winter (Fig. 2, Table A.3). Their probability of site occupancy increased with increasing percentage of marshland, with increasing cost of access, with increasing structural complexity of vegetation, and during the winter (Fig. 2, Table A.4).

For the gray brocket, the top-ranked model included cattle presence and season as variables interacting and affecting p and four variables affecting ψ : cost of access, percentage of forest within 500 m, habitat type, and complexity of vegetation (Table A.5). The detectability of gray brockets decreased with cattle presence, but particularly so during the winter (Fig. 3, Table A.6). Contrary to expectations, ψ decreases with increasing cost of access. The probability of site occupancy of this cervid was higher in forests and pine plantations than in grasslands. It increases with increasing percentage of forests within 500 m, and with increasing structural complexity of vegetation (Fig. 3, Table A.7).

For marsh deer, the top-ranked model included the presence of cattle and distance to water affecting p , and three variables affecting ψ : the percentage of marshland within 5 km, habitat type and the cost of access (Table A.8). The probability of detecting marsh deer was much lower in

areas with cattle and decreased with increasing distance to a water body (Fig. 4, Table A.9). The probability of occupancy of marsh deer was lower in forests than in grasslands. It increased with increasing percentage of marshland within 5 km, and with increasing cost of access (Fig. 4, Table A.10).

3.2. Diel activity patterns

The four herbivores showed cathemeral diel activity, with records during both day and night. Cattle showed more diurnal records in grasslands, more nocturnal records in forests, and a pronounced evening peak in pine plantations (Fig. A.1). In contrast with cattle, native herbivores tended to be less diurnal in grasslands than in forests and plantations (Fig. A.1).

For the three herbivores, the overlap (Δ) in diel activity with cattle was variable and dependent on the habitat type. Overlap with cattle diel activity ranged from as low as 0.33 for gray brocket in grasslands when cattle were present, to as high as 0.79 for capybara in pine plantations when cattle were absent (Table 1). When considering the three habitat types together, capybaras and brockets showed lower Δ with cattle in stations with presence of the latter, while marsh deer showed a similar tendency (Table 1). This temporal avoidance of cattle was stronger in the grasslands, where Δ was much lower when cattle were present (a reduction of -0.25 for capybaras, -0.19 for brockets, and -0.31 for marsh deer) and where the diel activity of the three native herbivores became more nocturnal when cattle were present (Fig. 5). The reduction of Δ with cattle in grasslands when this livestock was present was observed both during summer and winter in capybaras (Table A.11). At least for capybara, the reduction in Δ when cattle were present was similar in areas of low ($\Delta = -0.30$) and high ($\Delta = -0.28$) cost of access (Table A.12). For the two other species, comparisons between seasons and levels of cost of access were not possible due to small sample sizes (Tables A.11 and A.12).

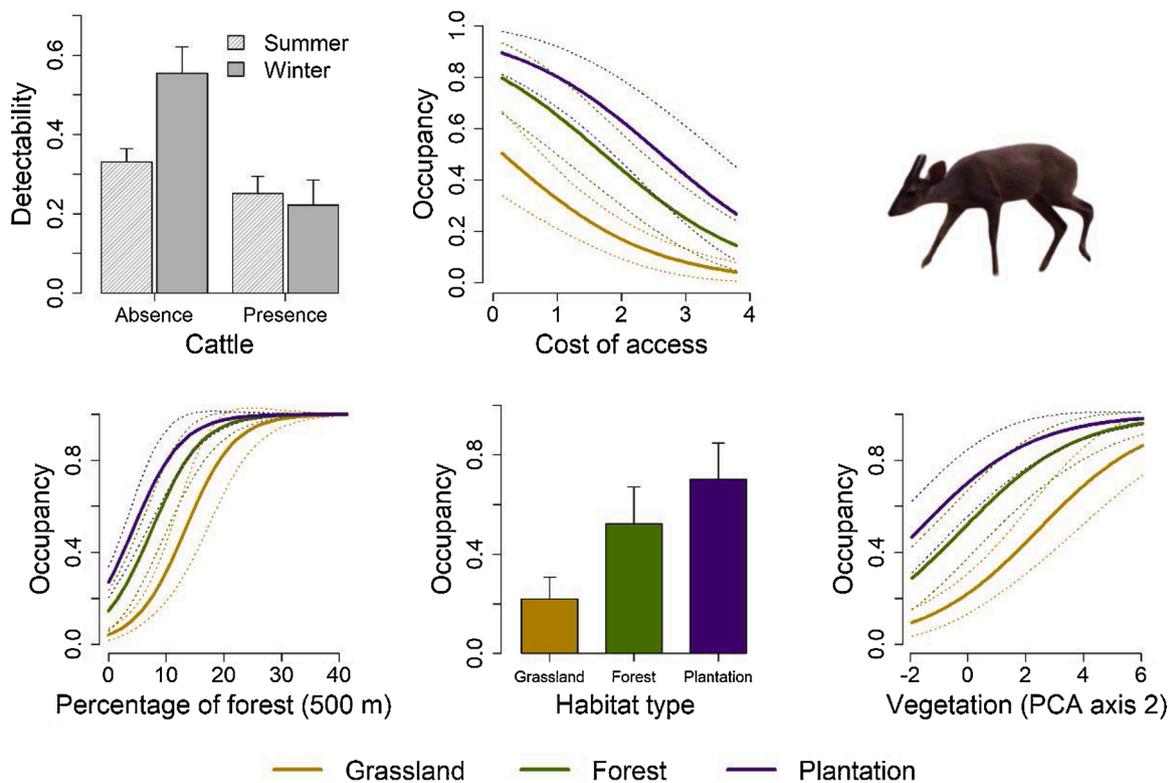


Fig. 3. Probability of occupancy and detectability of gray brocket deer (*Mazama gouazoubira*). Records for this analysis were obtained during a camera-trap survey conducted in the Southern Cone Mesopotamian Savannas and the Iberá marshes ecoregions of northern Corrientes and southern Misiones provinces of Argentina. Solid lines depict predicted values, broken lines and error bars represent the standard error of the mean.

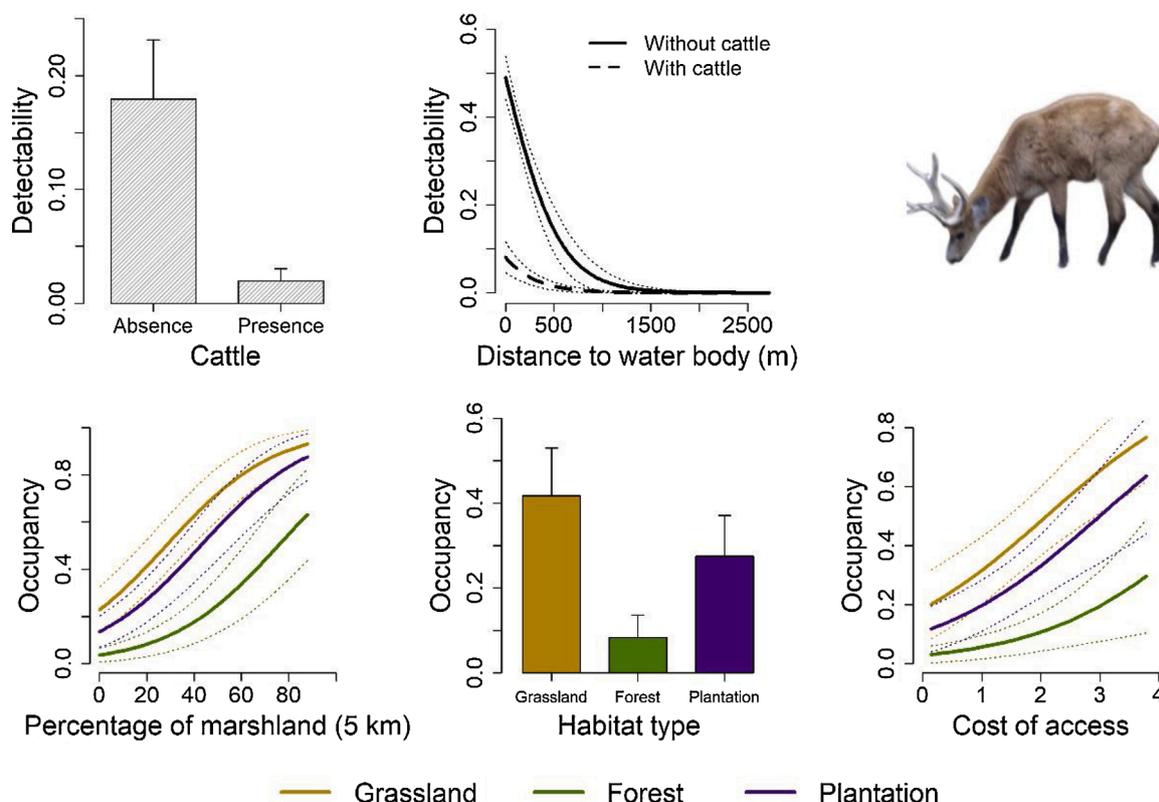


Fig. 4. Probability of occupancy and detectability of marsh deer (*Blastocerus dichotomus*). Records for this occupancy analysis were obtained during a camera-trap survey conducted in the Southern Cone Mesopotamian Savannas and the Iberá marshes ecoregions of northern Corrientes and southern Misiones provinces of Argentina. Solid lines depict predicted values, broken lines and error bars represent the standard error of the mean.

Table 1

Estimated coefficient of overlap (Δ) of daily activity records (and their 95 % confidence intervals) of three large herbivores (capybara, *Hydrochoerus hydrochaeris*, gray brocket *Mazama gouazoubira*, and marsh deer *Blastocerus dichotomus*) with the pattern observed for cattle (*Bos taurus*) in three different habitat types (grasslands, forests and pine plantations) in Northeastern Argentina. Overlap with cattle was estimated and compared between stations located in areas without cattle (Δ no cattle) and with cattle (Δ cattle). Δ difference = Δ no cattle - Δ cattle. Estimates of Δ were not performed for samples sizes <7. Comparisons for which 95 % confidence intervals do not overlap are highlighted in bold.

Species	Habitat type	Δ no cattle	N	95 % CI no cattle	Δ cattle	N	95 % CI cattle	Δ difference
Capybara	All	0.81	853	0.77 - 0.84	0.56	193	0.49 - 0.59	-0.25
	Grassland	0.74	669	0.70 - 0.78	0.49	153	0.39 - 0.51	-0.25
	Forest	0.71	111	0.58 - 0.77	0.77	33	0.65 - 0.89	0.06
	Plantation	0.79	73	0.73 - 0.92	0.50	7	0.36 - 0.75	-0.29
Gray brocket	All	0.80	232	0.77 - 0.86	0.67	73	0.56 - 0.75	-0.13
	Grassland	0.52	41	0.30 - 0.55	0.33	16	0.03 - 0.34	-0.19
	Forest	0.68	115	0.56 - 0.75	0.70	24	0.65 - 0.95	0.02
	Plantation	0.75	76	0.65 - 0.82	0.71	33	0.63 - 0.90	-0.04
Marsh deer	All	0.80	192	0.77 - 0.88	0.59	14	0.45 - 0.85	-0.21
	Grassland	0.71	101	0.64 - 0.80	0.40	10	0.21 - 0.59	-0.31
	Forest	0.73	49	0.67 - 0.90	NA	1	NA	NA
	Plantation	0.72	42	0.61 - 0.84	NA	3	NA	NA

4. Discussion

4.1. Detectability and occupancy

The two cervids strongly reduced their detectability when cattle were present. For the gray brocket, this reduction was stronger during the winter. Cattle, however, did not affect the probability of occupancy of the native herbivores. For the capybara and the marsh deer, detectability decreased, as predicted, with distance to the nearest water body. The probability of detection and occupancy of the capybara increased during the winter, suggesting a higher level of activity and increased

ranging during this season. The habitat type where camera traps were placed, amount of marshland or forest in the landscape, and the structural complexity of the vegetation were, as predicted, important determinants of ψ . For capybara and marsh deer, the observed relationship of ψ with the cost of access was expected, since the latter has a positive effect on wildlife in areas where poaching is frequent (Benítez-López et al., 2017; Peres, Emilio, Schiatti, Desmoulière, & Levi, 2016). However, for gray brocket, ψ showed an unexpectedly negative relationship with cost of access, even when this species is resilient to hunting (Juliá et al., 2019; Noss & Cuéllar, 2000). While the association of the native herbivores with certain habitat types were already known (of marsh

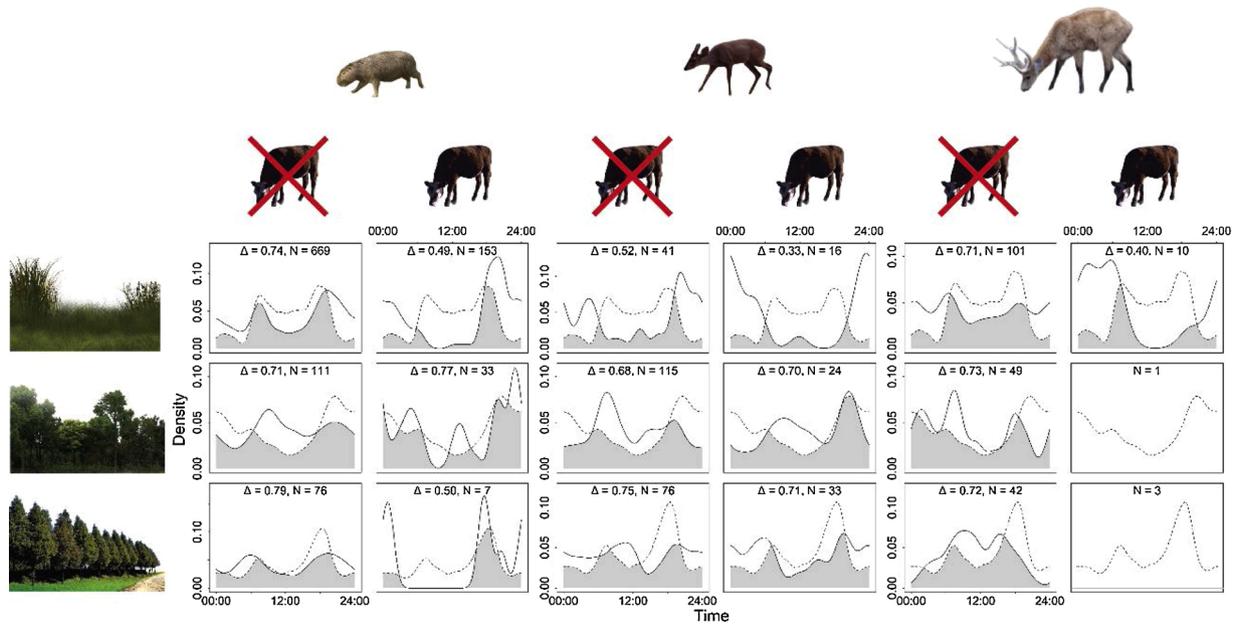


Fig. 5. Overlap in diel activity patterns of capybaras *Hydrochoerus hydrochoeris* (first two columns), gray brocket *Mazama gouazoubira* (two central columns) and marsh deer *Blastocercus dichotomus* (last two columns) with the diel activity pattern of cattle. The dotted lines correspond to the activity patterns of cattle, filled lines to those of the native herbivores. Diel activity patterns are based on the frequency of records obtained during a camera-trap survey conducted in the Southern Cone Mesopotamian Savannas and the Iberá marshes ecoregions of northern Corrientes and southern Misiones provinces of Argentina. For each species, the left column shows the overlap of the activity pattern of that species observed in camera-traps located in sites without cattle and the second column corresponds to the pattern observed in cameras placed at locations with cattle (in both cases compared to the pattern observed for cattle in the study area). The first line depicts the overlap observed camera-trap stations located in grasslands, the second in forests, and the third in pine plantations.

deer and capybara with marshlands, permanent water bodies and grasslands, and of gray brocket with forests; Herrera & Macdonald, 1989; Mattioli, 2011), the effects of the structural complexity of vegetation on ψ and the negative effect of cattle on the detectability of the cervids deserve a discussion in the context of beef production and cattle management.

The fact that cattle do not affect the occupancy of the native herbivores could be interpreted as evidence for lack of competition: their presence does not deplete or preclude access to critical resources needed for the native herbivores to establish their home ranges (second-order habitat selection *sensu* Johnson, 1980). However, the fact that cattle negatively affect the probability of detection of the two cervids suggests that, within their home ranges, the latter reduce their activity in areas with cattle (third-order habitat selection, Johnson, 1980). This change in activity by the native deer suggests that they are avoiding cattle and, therefore, that cattle may be exerting interference competition on them.

Capybaras, which *a priori* were expected to show the strongest negative spatial use response to cattle due to their similarities in diet, showed the lowest one. This differential response among species may result from the different social systems and space requirements of the three species. Social groups of capybaras have relatively small territories in relation to the larger home ranges of the cervids and in relation to the size of cattle enclosures. Deer home ranges may spread across several livestock enclosures, which allow them to mitigate the competitive effect of cattle by reducing the visitation rate to areas of their home ranges where cattle are temporarily present. Given their smaller size, home ranges of capybaras may be confined to one cattle enclosure (e.g., Desbiez et al., 2011), which may explain why cattle have no effect on their third-order habitat selection.

The probability of occupancy by capybaras and gray brocket increased with the complexity and diversity of the vegetation. Large herbivores may be vulnerable in open environments and may prefer to use complex vegetation that provides concealment against predators (Creel, Winnie, Maxwell, Hamlin, & Creel, 2005), including humans. Capybaras prefer areas with vegetation cover which provide them

shelter, resting sites, and forage (Corriale & Herrera, 2013; Fracassi, Buchter, Pereira, Borodowski, & Somma, 2015) and which are associated with female reproductive success (Herrera & Macdonald, 1989). Gray brockets may also find forests with a clean understory barren of refuge and browse. Cattle, through grazing, browsing and trampling, and people with the use of fires to promote regrowth of pastures, tend to reduce the amount and complexity of vegetation and thus, indirectly, the occupancy of habitat types by capybaras and brockets. It remains unclear why the occupancy of marsh deer was not affected by the structural complexity and diversity of vegetation but, given their large size, tall grasses may not be an effective shield in these grasslands, for which they may rely on other strategies to reduce detection by predators.

4.2. Diel activity patterns and overlap with cattle

A negative response to cattle was also observed in the diel temporal axis: the three species avoided cattle and became more nocturnal in the grasslands when the latter, mostly diurnal in this habitat type, were present. Nanni (2015) also reported that gray brockets become more nocturnal in areas with cattle in northwestern Argentina. Despite being usually characterized as a cathemeral or crepuscular species, cattle show more diurnal locomotor activity (Piccione, Giannetto, Caola, & Casella, 2010; Piccione, Giannetto, Schembari, Ganesella, & Morgante, 2011). This pattern was neatly observed in grasslands, but not in forests and plantations, where they were more active at night. Cattle tend to concentrate their grazing in diurnal hours, starting with a main meal near sunrise and finishing with another at sunset (Albright, 1993; Arnold, 1984; Gregorini, Tamminga, & Gunter, 2006). Cattle seem to be using forests and pine plantations as resting places or refuges more than as foraging areas. The observed patterns suggest that wild herbivores avoid cattle when the latter is active and grazing, something that mostly occurs in the grasslands. Even though diel activity patterns may change seasonally and with hunting pressure, avoidance of cattle in grasslands, at least for capybaras, was observed during winter and summer and in

areas of high and low cost of human access.

4.3. Conclusions and recommendations for nature conservation

The observed patterns of habitat use and diel activity suggest that cattle are exerting interference competition on wild herbivores. Based on their diets and morphological characteristics, we expected a larger effect of cattle on the capybara than on the two native cervids. However, both cervids responded as negatively or more than capybara to the presence of cattle, especially in the spatial domain. Avoidance of cattle by ungulates, and particularly by deer, has been frequently reported in the literature as behavioral or “social intolerance” (Perez Carusi et al., 2017; Schieltz & Rubenstein, 2016). Even though pampas deer and gray brocket can coexist with cattle at low stocking rates (Merino, Semeniuk, & Fa, 2011; Noss & Cuéllar, 2000), they are negatively affected by livestock (Nanni, 2015; Perez Carusi et al., 2017; Puechagut et al., 2018; Vila et al., 2008). The paucity of studies on the effect of cattle on Neotropical cervids does not allow yet to make strong generalizations on their responses to cattle.

Although our results suggest that cattle exert interference competition on the large native herbivores, it is not clear if this livestock may also be exerting exploitative competition on them. Exploitative competition by cattle is most frequently observed in low productive ecosystems, especially during the dry season and at high stocking rates (Young et al., 2018). We observed a strong effect of season in the capybara and the gray brocket, which corroborates that large herbivores at our study site are limited by food during the winter. However, we did not observe an effect of cattle on ψ . This suggests that stocking rates at our study site are not high and cattle do not exert strong exploitative competition on native herbivores, even during the winter. However, the observed positive effect on ψ of the structural complexity of vegetation, which is reduced by cattle through grazing and browsing (and also by the frequent burning of grasslands) may partially encapsulate this exploitative effect. Future studies should consider experimental approaches (e.g., Odadi et al., 2011) to evaluate the relative effects of direct competition, indirect competition, and facilitation by cattle on wild herbivores.

Since animals can associate cues with predation risk, an alternative explanation for the observed patterns is that wild ungulates are not avoiding competition by cattle but associating livestock with people. Thus, the spatial and temporal patterns observed may represent responses to a landscape of fear (Gaynor, Brown, Middleton, Power, & Brashares, 2019). Both the spatial (Cruz et al., 2018) and temporal avoidance (Di Bitetti, Paviolo, Ferrari, De Angelo, & Di Blanco, 2008; Gaynor et al., 2018) of people by wildlife have been well documented. Even though the cost of human access had no effect on the diel activity of the three herbivores, it remains possible that the presence of cattle serves as a clue used to assess the probability of encountering people that live within estancias managing livestock and whose effect were not contemplated in the cost of access model. This is something that should also be explored in future studies.

Indirect non-competitive effects of cattle grazing on native herbivores are probably as important or more than the competitive effects. The negative effect of the cost of human access on capybaras and marsh deer indicate that these two species are very sensitive to the presence of people. Employees of cattle ranches and estancias live with their families in isolated posts. Several dogs, an important source of mortality and harassment to native deer species (Juliá et al., 2019; Merino et al., 2019; Pereira et al., 2019), live with them and accompany them on their cattle caring activities. Legal and illegal hunting of wildlife by local people is common, especially of capybaras (Quintana & Bolkovic, 2012). While our cost of access model did not include the effect of individual posts, it suggests that native herbivores, except gray brockets, are overly sensitive to hunting. Ranchers frequently burn grasslands to promote regrowth of pastures (Di Bella, Fischer, & Jobbágy, 2011). While this may be beneficial to cattle, it reduces the structural complexity of

vegetation and has a negative effect on the probability of occupancy of capybara. Thus, management of cattle should strive to mitigate indirect negative effects of this activity: presence of dogs in rural areas, hunting of wildlife, and excessive burning of grasslands. These management actions should be accompanied with more stringent nature conservation actions, such as the creation of cattle-free protected areas where poaching is controlled, to preserve the diverse native grasslands and keystone habitats on which the native wild herbivores and other Neotropical mammals depend (Jezzi et al., 2020).

The direct and indirect negative effects of extensive and intensive cattle production on nature have been amply documented (Beschta et al., 2013; Steinfeld et al., 2006). Particularly, competition with livestock is one of the main threats to large herbivores (Ripple et al., 2015). Even though exploitative competition by cattle on large herbivores is only realized under certain circumstances (e.g., low productive environments and high stocking rates, Odadi et al., 2011), avoidance of cattle by native herbivores seems to be the rule globally (Schieltz & Rubenstein, 2016) and, as our results suggest, the Neotropical region is no exception. The conversion of forests to grasslands for beef production in South America is the main driver of deforestation (De Sy et al., 2015) and it has tremendous negative impacts on biodiversity (Alkemade et al., 2013). A global reduction of meat consumption is needed to ameliorate these pervasive global impacts of livestock production (Machovina, Feeley, & Ripple, 2015). However, the conversion of natural grasslands to crop systems (e.g., corn) or tree plantations (Di Giacomo et al., 2010) may generate even more negative impacts on wildlife than livestock production (Fargione et al., 2009). For this reason, in a regional context of natural savannas and grasslands, extensive and responsible livestock grazing is probably the best productive option which maximizes wildlife and economic benefits (Allan et al., 2017; Modernel et al., 2016). But this requires mitigation measures and best practices (Marino et al., 2013), such as low stocking rates to avoid overgrazing, cattle-free reserved areas, control of poaching, prescribed burns, and top-predator conflict resolution.

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Declaration of Competing Interest

The authors report no declarations of interest.

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Appendix A. Supplementary data

Supplementary material related to this article can be found, in the online version, at doi:<https://doi.org/10.1016/j.jnc.2020.125900>.

References

- Albright, J. L. (1993). Feeding behavior of dairy cattle. *Journal of Dairy Science*, 76, 485–498. [https://doi.org/10.3168/jds.S0022-0302\(93\)77369-5](https://doi.org/10.3168/jds.S0022-0302(93)77369-5).
- Alkemade, R., Reid, R. S., van den Berg, M., de Leeuw, J., & Jeuken, M. (2013). Assessing the impacts of livestock production on biodiversity in rangeland ecosystems. *Proceedings of the National Academy of Sciences of the USA*, 110, 20900–20905. <https://doi.org/10.1073/pnas.1011013108>.
- Allan, B. F., Tallis, H., Chaplin-Kramer, R., Hockett, S., Kowal, V. A., Musengezi, J., et al. (2017). Can integrating wildlife and livestock enhance ecosystem services in central Kenya? *Frontiers in Ecology and the Environment*, 15, 328–335. <https://doi.org/10.1002/fee.1501>.
- Anderson, D. R. (2007). *Model based inference in the life sciences: A primer on evidence*. Berlin/Heidelberg: Springer Science & Business Media.
- Arnold, G. W. (1984). Comparison of the time budgets and circadian patterns of maintenance activities in sheep, cattle and horses grouped together. *Applied Animal Behaviour Science*, 13, 19–30. [https://doi.org/10.1016/0168-1591\(84\)90048-0](https://doi.org/10.1016/0168-1591(84)90048-0).
- Augustine, D. J. (2004). Influence of cattle management on habitat selection by Impala on Central Kenyan Rangeland. *The Journal of Wildlife Management*, 68, 916–923. [https://doi.org/10.2193/0022-541X\(2004\)068\[0916:IOCMOHJ\]2.0.CO;2](https://doi.org/10.2193/0022-541X(2004)068[0916:IOCMOHJ]2.0.CO;2).
- Ba Diao, M. (2020). *Livestock production and conservation in and around protected areas: The Project for Integrated Ecosystem Management in Senegal* Accessed 29 January 2020 <http://www.fao.org/3/a0532e/A0532e04.htm>.
- Baldi, R., Albon, S., & Elston, D. (2001). Guanacos and sheep: Evidence for continuing competition in arid Patagonia. *Oecologia*, 129, 561–570. <https://doi.org/10.1007/s004420100770>.
- Beccaceci, M. D. (1996). Dieta del ciervo de los pantanos, *Blastocerus dichotomus*, en la Reserva del Iberá, Corrientes, Argentina. *Mastozoología Neotropical*, 3, 193–197.
- Begon, M., Harper, J. L., & Townsend, C. R. (1990). *Ecology: Individuals, populations and communities*. Oxford: Blackwell.
- Benítez-López, A., Alkemade, R., Schipper, A. M., Ingram, D. J., Verweij, P. A., Eikelboom, J. A. J., et al. (2017). The impact of hunting on tropical mammal and bird populations. *Science*, 356, 180–183. <https://doi.org/10.1126/science.aaj1891>.
- Bernardi, R. E., Staal, A., Xu, C., Scheffer, M., & Holmgren, M. (2019). Livestock herbivory shapes fire regimes and vegetation structure across the global tropics. *Ecosystems*, 22, 1457–1465. <https://doi.org/10.1007/s10021-019-00349-x>.
- Beschta, R. L., Donahue, D. L., DellaSala, D. A., Rhodes, J. J., Karr, J. R., O'Brien, M. H., et al. (2013). Adapting to climate change on western public lands: Addressing the ecological effects of domestic, wild, and feral ungulates. *Environmental Management*, 51, 474–491. <https://doi.org/10.1007/s00267-012-9964-9>.
- Bilenca, D., Codesido, M., González Fischer, C., Pérez Carusi, L., Zufiaurre, E., & Abba, A. (2012). Impactos de la transformación agropecuaria sobre la biodiversidad en la provincia de Buenos Aires. *Revista del Museo Argentino de Ciencias Naturales, n. s.*, 14, 189–198.
- Black-Décima, P., Vierra Rossi, R., Vogliotti, A., Cartes, J. L., Maffei, L., Barbanti Duarte, J. M., González, S., Juliá, J. P., et al. (2010). Brown brocket deer *Mazama gouazoubira* (Fischer 1814). In J. M. Barbanti Duarte, & S. González (Eds.), *Neotropical cervidology. Biology and medicine of latin American deer* (pp. 190–201). Jaboticabal, Brazil: Funep/IUCN.
- Bolkovic, M. L., Quintana, R., Cirignoli, S., Perovic, P. G., Eberhardt, A., Byrne, S., Bareiro, R., Porini, G., et al. (2019). *Hydrochoerus hydrochaeris*. In SAyDS-SAREM (Ed.), *Categorización 2019 de los mamíferos de Argentina según su riesgo de extinción*. Lista Roja de los mamíferos de Argentina. <https://doi.org/10.31687/SaremlR.19.339>.
- Borgnia, M., Vilá, B. L., & Cassini, M. H. (2008). Interaction between wild camels and livestock in an Andean semi-desert. *Journal of Arid Environments*, 72, 2150–2158. <https://doi.org/10.1016/j.jaridenv.2008.07.012>.
- Brown, A. D., & Pacheco, S. (2006). Propuesta de actualización del mapa de ecoregional de la Argentina. In A. D. Brown, U. Martínez Ortiz, M. Acerbi, & J. Corcuera (Eds.), *La situación ambiental argentina 2005* (pp. 28–31). Buenos Aires: Fundación Vida Silvestre Argentina.
- Burgi, M. V., Marino, A., Rodríguez, M. V., Pazos, G., & Baldi, R. (2011). Response of guanacos *Lama guanicoe* to changes in land management in Península Valdés, Argentine Patagonia: conservation implications. *Oryx*, 46, 99–105. <https://doi.org/10.1017/S0030605311000214>.
- Burkart, R., Bárbaro, N. O., Sánchez, R. O., & Gómez, D. A. (1999). *Ecoregiones de la Argentina*. Buenos Aires: Administración de Parques Nacionales.
- Burnham, K. P., & Anderson, D. R. (2002). *Model selection and multiple model inference: A practical information-theoretical approach* (2nd ed.). New York: Springer.
- Butt, B., & Turner, M. D. (2012). Clarifying competition: The case of wildlife and pastoral livestock in East Africa. *Pastoralism*, 2, 9. <https://doi.org/10.1186/2041-7136-2-9>.
- Carothers, J. H., & Jaksić, F. M. (1984). Time as a niche difference: The role of interference competition. *Oikos*, 42, 403–406. <https://doi.org/10.2307/3544413>.
- Caruso, F., & Jiménez Pérez, I. (2013). Tourism, local pride, and attitudes towards the reintroduction of a large predator, the jaguar *Panthera onca* in Corrientes, Argentina. *Endangered Species Research*, 21, 263–272. <https://doi.org/10.3354/esr00519>.
- Catchpole, W. R., & Wheeler, C. J. (1992). Estimating invasion biomass: A review of techniques. *Austral Ecology*, 17, 121–131. <https://doi.org/10.1111/j.1442-9993.1992.tb00790.x>.
- Cingolani, A. M., Vaieretti, M. V., Giorgis, M. A., La Torre, N., Whitworth-Hulse, J. I., & Renison, D. (2013). Can livestock and fires convert the sub-tropical mountain rangelands of central Argentina into a rocky desert? *The Rangeland Journal*, 35, 285–297. <https://doi.org/10.1071/RJ12095>.
- Corriale, M. J., & Herrera, E. A. (2013). Patterns of habitat use and selection by the capybara (*Hydrochoerus hydrochaeris*): a landscape-scale analysis. *Ecological Research*, 29, 191–201. <https://doi.org/10.1007/s11284-013-1113-2>.
- Corriale, M. J., Muschetto, E., & Herrera, E. A. (2013). Influence of group sizes and food resources in home-range sizes of capybaras from Argentina. *Journal of Mammalogy*, 94, 19–28. <https://doi.org/10.1644/12-MAMM-A-030.1>.
- Creel, S., Winnie, J., Jr., Maxwell, B., Hamlin, K., & Creel, M. (2005). Elk alter habitat selection as an antipredator response to wolves. *Ecology*, 86, 3387–3397. <https://doi.org/10.1890/05-0032>.
- Cruz, P., Iezzi, M. E., De Angelo, C., Varela, D., & Di Bitetti, M. S. (2019). Landscape use by two opossums is shaped by habitat preferences rather than by competitive interactions. *Journal of Mammalogy*, 100, 1966–1978. <https://doi.org/10.1093/jmammal/gyz133>.
- Cruz, P., Iezzi, M. E., De Angelo, C., Varela, D., Di Bitetti, M. S., & Paviolo, A. (2018). Effects of human impacts on habitat use, activity patterns and ecological relationships among medium and small felids of the Atlantic Forest. *PLoS One*, 13, Article e0200806. <https://doi.org/10.1371/journal.pone.0200806>.
- D'Antonio, C. M., & Vitousek, P. M. (1992). Biological invasions by exotic grasses, the grass/fire cycle, and global change. *Annual Review of Ecology and Systematics*, 23, 63–87.
- De Sy, V., Herold, M., Achard, F., Beuchle, R., Clevers, J. G. P. W., Lindquist, E., et al. (2015). Land use patterns and related carbon losses following deforestation in South America. *Environmental Research Letters*, 10, Article 124004. <https://iopscience.iop.org/article/10.1088/1748-9326/10/12/124004#erlaa0a70s3>.
- Desbiez, A. L. J., Santos, S. A., Alvarez, J. M., & Tomas, W. M. (2011). Forage use in domestic cattle (*Bos indicus*), capybara (*Hydrochoerus hydrochaeris*) and pampas deer (*Ozotoceros bezoarticus*) in a seasonal Neotropical wetland. *Mammalian Biology*, 76, 351–357. <https://doi.org/10.1016/j.mambio.2010.10.008>.
- Di Bella, C. M., Fischer, M. A., & Jobbágy, E. G. (2011). Fire patterns in north-eastern Argentina: Influences of climate and land use/cover. *International Journal of Remote Sensing*, 32, 4961–4971. <https://doi.org/10.1080/01431161.2010.494167>.
- Di Bitetti, M. S. (2001). Home range use by the tufted capuchin monkey, *Cebus apella nigratus*, in a subtropical rainforest of Argentina. *Journal of Zoology*, 253, 33–45. <https://doi.org/10.1017/S0952836901000048>.
- Di Bitetti, M. S., Di Blanco, Y. E., Pereira, J. A., Paviolo, A., & Jiménez Pérez, I. (2009). Time partitioning favors the coexistence of sympatric crab-eating foxes (*Cerdocyon thous*) and pampas foxes (*Lycalopex gymnocercus*). *Journal of Mammalogy*, 90, 479–490. <https://doi.org/10.1644/08-MAMM-A-113.1>.
- Di Bitetti, M. S., Paviolo, A., Ferrari, C. A., De Angelo, C., & Di Blanco, Y. (2008). Differential responses to hunting in two sympatric species of brocket deer (*Mazama americana* and *M. nana*). *Biotropica*, 40, 636–645. <https://doi.org/10.1111/j.1744-7429.2008.00413.x>.
- Di Blanco, Y. E., Jiménez Pérez, I., & Di Bitetti, M. S. (2015). Habitat selection in reintroduced giant anteaters: The critical role of conservation areas. *Journal of Mammalogy*, 96, 1024–1035. <https://doi.org/10.1093/jmammal/gyv107>.
- Di Blanco, Y. E., Spörring, K. L., & Di Bitetti, M. S. (2017). Daily activity pattern of reintroduced giant anteaters (*Myrmecophaga tridactyla*): Effects of seasonality and experience. *Mammalia*, 81, 11–21. <https://doi.org/10.1515/mammalia-2015-0088>.
- Di Giacomo, A. S., Vickery, P. D., Casanas, H., Spitznagel, O. A., Ostrosky, C., Krapovickas, S., et al. (2010). Landscape associations of globally threatened grassland birds in the Aguapey river Important Bird Area, Corrientes, Argentina. *Bird Conservation International*, 20, 62–73. <https://doi.org/10.1017/S0959270909990177>.
- Efford, M. G., & Dawson, D. K. (2012). Occupancy in continuous habitat. *Ecosphere*, 3, 1–15. <https://doi.org/10.1890/ES11-00308.1>.
- Eldridge, D. J., Poore, A. G. B., Ruiz-Colmenero, M., Letnic, M., & Soliveres, S. (2016). Ecosystem structure, function, and composition in rangelands are negatively affected by livestock grazing. *Ecological Applications*, 26, 1273–1283. <https://doi.org/10.1890/1520-1234>.
- Estevo, C. A., Nagy-Reis, M. B., & Nichols, J. D. (2017). When habitat matters: Habitat preferences can modulate co-occurrence patterns of similar sympatric species. *PLoS One*, 12, Article e0179489. <https://doi.org/10.1371/journal.pone.0179489>.
- FAO. (2009). *Sustaining communities, livestock and wildlife: A guide to participatory land-use planning*. Rome: FAO.
- FAO. (2020). *Production of meat indigenous, cattle: Top 10 producers* Accessed 31 January 2020 <http://www.fao.org/faostat/en/#data/QL/visualize>.
- Fargione, J. E., Cooper, T. R., Flaspohler, D. J., Hill, J., Lehman, C., Tilman, D., et al. (2009). Bioenergy and wildlife: Threats and opportunities for grassland conservation. *BioScience*, 59, 767–777. <https://doi.org/10.1525/bio.2009.59.9.8>.

- Fiske, I., & Chandler, R. (2011). Unmarked: An R package for fitting hierarchical models of wildlife occurrence and abundance. *Journal of Statistical Software*, 43, 1–23. <https://www.jstatsoft.org/article/view/v43i10>.
- Fracassi, N. G., Buchter, W. M., Pereira, J. A., Borodowski, E. D., & Somma, D. (2015). Determinants of capybara presence in afforestations of the lower delta of the Paraná river, Argentina. *Studies on Neotropical Fauna and Environment*, 50, 229–237. <https://doi.org/10.1080/01650521.2015.1110406>.
- Frey, S., Fisher, J. T., Burton, A. C., & Volpe, J. P. (2017). Investigating animal activity patterns and temporal niche partitioning using camera-trap data: Challenges and opportunities. *Remote Sensing in Ecology and Conservation*, 3, 123–132. <https://doi.org/10.1002/rse2.60>.
- Gaynor, K. M., Brown, J. S., Middleton, A. D., Power, M. E., & Brashares, J. S. (2019). Landscapes of fear: Spatial patterns of risk perception and response. *Trends in Ecology & Evolution*, 34, 355–368. <https://doi.org/10.1016/j.tree.2019.01.004>.
- Gaynor, K. M., Hohnowski, C. E., Carter, N. H., & Brashares, J. S. (2018). The influence of human disturbance on wildlife nocturnality. *Science*, 360, 1232–1235. <https://doi.org/10.1126/science.aar7121>.
- Gorosábel, A., Loponte, D., & Corriale, M. J. (2016). Mortalidad estacional de carpinchos (*Hydrochoerus hydrochaeris*) y su relación con la disponibilidad de recursos en los Esteros del Iberá, Argentina. *Mastozoología Neotropical*, 23, 401–413. <http://www.redalyc.org/articulo.oa?id=45750282015>.
- Gregorini, P., Tamminga, S., & Gunter, S. A. (2006). Behavior and daily grazing patterns of cattle. *The Professional Animal Scientist*, 22, 201–209. [https://doi.org/10.15232/S1080-7446\(15\)31095-0](https://doi.org/10.15232/S1080-7446(15)31095-0).
- Griffin, J. N., & Silliman, B. R. (2011). Resource partitioning and why it matters. *Nature Education Knowledge*, 3(10), 49.
- Herrera, E. A., & Macdonald, D. W. (1989). Resource utilization and territoriality in group-living capybaras (*Hydrochoerus hydrochaeris*). *The Journal of Animal Ecology*, 58, 667–679. <https://doi.org/10.2307/4855>.
- Iezzi, M. E., De Angelo, C., & Di Bitetti, M. S. (2020). Tree plantations replacing natural grasslands in high biodiversity areas: How do they affect the mammal assemblage? *Forest Ecology and Management*, 473, Article 118303. <https://doi.org/10.1016/j.foreco.2020.118303>.
- Illius, A. W., & Gordon, I. J. (1992). Modelling the nutritional ecology of ungulate herbivores: Evolution of body size and competitive interactions. *Oecologia*, 89, 428–434. <https://doi.org/10.1007/BF00317422>.
- Jerozolinski, A., & Peres, C. A. (2003). Bringing home the biggest bacon: A cross-site analysis of the structure of hunter-kill profiles in Neotropical forests. *Biological Conservation*, 111, 415–425. [https://doi.org/10.1016/S0006-3207\(02\)00310-5](https://doi.org/10.1016/S0006-3207(02)00310-5).
- Johnson, D. H. (1980). The comparison of usage and availability measurements for evaluating resource preference. *Ecology*, 61, 65–71. <https://doi.org/10.2307/1937156>.
- Juliá, J. P., Varela, D., Periago, M. E., Cirignoli, S., Muzzachiodi, N., Camino, M., Barri, F., Iezzi, M. E., de Bustos, S., et al. (2019). *Mazama gouazoubira*. In SAyDS-SAREM (Ed.), *Categorización 2019 de los mamíferos de Argentina según su riesgo de extinción*. Lista Roja de los mamíferos de Argentina. <https://doi.org/10.31687/SaremLR.19.211>.
- Keesing, F., Ostfeld, R. S., Okanga, S., Huckett, S., Bayles, B. R., Chaplin-Kramer, R., et al. (2018). Consequences of integrating livestock and wildlife in an African savanna. *Nature Sustainability*, 1, 566–573. <https://doi.org/10.1038/s41893-018-0149-2>.
- Kinnaird, M. F., & O'Brien, T. G. (2012). Effects of private-land use, livestock management, and human tolerance on diversity, distribution, and abundance of large African mammals. *Conservation Biology*, 26, 1026–1039. <https://doi.org/10.1111/j.1523-1739.2012.01942.x>.
- Lacher, T. E., Jr. (2016). Family Caviidae (Cavies, Capybaras and Maras). In D. E. Wilson, T. E. Lacher, Jr., & R. A. Mittermeier (Eds.), *Handbook of the mammals of the world. Vol. 6. Lagomorphs and rodents I* (pp. 406–439). Barcelona: Lynx Edicions.
- Lessa, I., Corrêa Seabra Guimarães, T., de Godoy Bergallo, H., Cunha, A., & Vieira, M. E. (2016). Domestic dogs in protected areas: a threat to Brazilian mammals? *Natureza & Conservação*, 14, 46–56. <https://doi.org/10.1016/j.ncon.2016.05.001>.
- Machovina, B., Feeley, K. J., & Ripple, W. J. (2015). Biodiversity conservation: The key is reducing meat consumption. *The Science of the Total Environment*, 536, 419–431. <https://doi.org/10.1016/j.scitotenv.2015.07.022>.
- MacKenzie, D. I. (2005). What are the issues with presence-absence data for wildlife managers? *The Journal of Wildlife Management*, 69, 849–860. [https://doi.org/10.2193/0022-541X\(2005\)069\[0849:WATWPI\]2.0.CO;2](https://doi.org/10.2193/0022-541X(2005)069[0849:WATWPI]2.0.CO;2).
- MacKenzie, D. I., & Bailey, L. L. (2004). Assessing the fit of site-occupancy models. *Journal of Agricultural, Biological, and Environmental Statistics*, 9, 300–318. <https://doi.org/10.1198/108571104X3361>.
- MacKenzie, D. I., Nichols, J. D., Lachman, G. B., Droege, S., Royle, A. J., & Langtimm, C. A. (2002). Estimating site occupancy rates when detection probabilities are less than one. *Ecology*, 83, 2248–2255. [https://doi.org/10.1890/0012-9658\(2002\)083\[2248:ESORWD\]2.0.CO;2](https://doi.org/10.1890/0012-9658(2002)083[2248:ESORWD]2.0.CO;2).
- MacKenzie, D. I., Nichols, J. D., Royle, J. A., Pollock, K. H., Bailey, L. L., & Hines, J. E. (2006). *Occupancy estimation and modeling: Inferring patterns and dynamics of species occurrence*. San Diego, USA: Academic Press, 344 p.
- Maloney, S. K., Moss, G., Cartmell, T., & Mitchell, D. (2005). Alteration in diel activity patterns as a thermoregulatory strategy in black wildebeest (*Connochaetes gnou*). *Journal of Comparative Physiology A*, 191, 1055–1064. <https://doi.org/10.1007/s00359-005-0030-4>.
- Marin, V. C., Fernández, V. A., Dacar, M. A., Gutiérrez, D. G., Fergnani, D., & Pereira, J. A. (2020). Diet of the marsh deer in the Paraná River Delta, Argentina—A vulnerable species in an intensive forestry landscape. *European Journal of Wildlife Research*, 66, 16. <https://doi.org/10.1007/s10344-019-1358-3>.
- Marino, D., Preliasco, G. P., Martínez Ortiz, U., Aiello, F., Sosa, L. L., Marani, M. D., Miñarro, F., et al. (2013). Las buenas prácticas ganaderas para el manejo sustentable de pastizales del centro y noreste argentinos. In G. D. Marino, F. Miñarro, M. E. Zaccagnini, & B. López-Lanús (Eds.), *Pastizales y sabanas del cono sur de Sudamérica: iniciativas para su conservación en la Argentina* (pp. 467–479). Buenos Aires: Aves Argentinas, Fundación Vida Silvestre Argentina e INTA. Temas de Naturaleza y Conservación, Monografía de Aves Argentinas 9 https://www.avesargentinas.org.ar/sites/default/files/Monograf%C3%ADa%20Nro.%20209.%20Cap%20XIX_reducido.pdf.
- Mattioli, S. (2011). Family Cervidae (Deer). In D. E. Wilson, & R. A. Mittermeier (Eds.), *Handbook of the mammals of the world vol 2, hoofed mammals* (pp. 350–443). Barcelona: Lynx Edicion.
- McNab, B. K. (1963). Bioenergetics and the determination of home range size. *The American Naturalist*, 97, 133–140.
- MEA, Millennium Ecosystem Assessment. (2005). *Ecosystems and human well-being: Biodiversity synthesis*. Washington, DC: World Resources Institute.
- Meredith, M., & Ridout, M. (2014). *Overlap: Estimates of coefficient of overlapping for animal activity patterns*. Package version 0.2.4. Available at: <http://CRAN.R-project.org/package=overlap>.
- Meredith, M., & Ridout, M. (2018). *Overview of the overlap package*. *R Proj.* (pp. 1–9).
- Merino, M. L., Cirignoli, S., Perez Carusi, L., Varela, D., Kin, M. S., Pautasso, A., Demaría, M., Beade, M. S., Uhart, M., et al. (2019). *Ozotoceros bezoarticus*. In SAyDS-SAREM (Ed.), *Categorización 2019 de los mamíferos de Argentina según su riesgo de extinción*. Lista Roja de los mamíferos de Argentina. <https://doi.org/10.31687/SaremLR.19.213>.
- Merino, M. L., Semeñuk, M. B., & Fa, J. E. (2011). Effect of cattle breeding on habitat use of Pampas deer *Ozotoceros bezoarticus celer* in semiarid grasslands of San Luis, Argentina. *Journal of Arid Environments*, 75, 752–756. <https://doi.org/10.1016/j.jaridenv.2011.03.002>.
- Modernel, P., Rossing, W. A. H., Corbeels, M., Dogliotti, S., Picasso, V., & Tiftonell, P. (2016). Land use change and ecosystem service provision in Pampas and Campos grasslands of southern South America. *Environmental Research Letters*, 11, 113002. <https://iopscience.iop.org/article/10.1088/1748-9326/11/11/113002>.
- Nagy-Reis, M. B., Estevo, C. A., Setz, E. Z. F., Ribeiro, M. C., Chiarello, A. G., & Nichols, J. D. (2017). Relative importance of anthropogenic landscape characteristics for Neotropical frugivores at multiple scales. *Animal Conservation*, 20, 520–531. <https://doi.org/10.1111/acv.12346>.
- Nanni, A. S. (2015). Dissimilar responses of the Gray brocket deer (*Mazama gouazoubira*), Crab-eating fox (*Cerdocyon thous*) and Pampas fox (*Lycalopex gymnocercus*) to livestock frequency in subtropical forests of NW Argentina. *Mammalian Biology*, 80, 260–264. <https://doi.org/10.1016/j.mambio.2015.04.003>.
- Neiff, J. J., & Neiff, M. (2013). *Evaluación de los impactos del cambio climático sobre el ecosistema natural y la biodiversidad Esteros del Iberá (Argentina)*. Serie Medio Ambiente y Desarrollo N° 152. Santiago de Chile: Naciones Unidas. ISSN:1564-4189.
- Neilson, E. W., Avgar, T., Burton, A. C., Broadley, K., & Boutin, S. (2018). Animal movement affects interpretation of occupancy models from camera-trap surveys of unmarked animals. *Ecosphere*, 9(1), e02092. <https://doi.org/10.1002/ecs2.2092>.
- Niedballa, J., Sollmann, R., Courtiol, A., & Wilting, A. (2016). camtrapR: An R package for efficient camera trap data management. *Methods in Ecology and Evolution*, 7, 1457–1462. <https://doi.org/10.1111/2041-210X.12600>.
- Noss, A., & Cuéllar, E. (2000). Índices de abundancia para fauna terrestre en el Chaco boliviano: huellas en parcelas y en brechas barridas. In E. Cabrera, C. Mercolli, & R. Resquin (Eds.), *Manejo de fauna silvestre en Amazonia y Latinoamérica* (pp. 73–82). Asunción, Paraguay: CITES Paraguay, Fundación Moises Bertoni, University of Florida.
- O'Brien, T. G. (2011). Abundance, density and relative abundance: A conceptual framework. In A. F. O'Connell, J. D. Nichols, & K. U. Karanth (Eds.), *Camera traps in animal ecology: Methods and analyses* (pp. 71–96). Tokyo, Japan: Springer. https://doi.org/10.1007/978-4-431-99495-4_6.
- Odadi, W. O., Karachi, M. K., Abdurazak, S. A., & Young, T. P. (2011). African wild ungulates compete with or facilitate cattle depending on season. *Science*, 333, 1753–1755. <https://doi.org/10.1126/science.1204498>.
- Oliveira-Santos, L. G. R., Zucco, C. A., & Agostinelli, C. (2013). Using conditional circular kernel density functions to test hypotheses on animal circadian activity. *Animal Behaviour*, 85, 269–280. <https://doi.org/10.1016/j.anbehav.2012.09.033>.
- Orozco, M. M., Marull, C., Jiménez, I., & Gürtler, R. E. (2013). Winter mortality of marsh deer (*Blastocercus dichotomus*) in wetlands of northeastern Argentina. *Mastozoología Neotropical*, 20, 163–170. <https://www.redalyc.org/articulo.oa?id=45728549015>.
- Pereira, J. A., Varela, D., Aprile, G., Cirignoli, S., Orozco, M. M., Lartigau, B., De Angelo, C., Giraudo, A. R., et al. (2019). *Blastocercus dichotomus*. In SAyDS-SAREM (Ed.), *Categorización 2019 de los mamíferos de Argentina según su riesgo de extinción*. Lista Roja de los mamíferos de Argentina. <https://doi.org/10.31687/SaremLR.19.207>.
- Peres, C. A., & Palacios, E. (2007). Basin-wide effects of game harvest on vertebrate population densities in Amazonian forests: Implications for animal-mediated seed dispersal. *Biotropica*, 39, 304–315. <https://doi.org/10.1111/j.1744-7429.2007.00272.x>.
- Peres, C. A., Emilio, T., Schiatti, J., Desmoulière, S. J., & Levi, T. (2016). Dispersal limitation induces long-term biomass collapse in overhunted Amazonian forests. *Proceedings of the National Academy of Sciences of the United States of America*, 113, 892–897. <https://doi.org/10.1073/pnas.1516525113>.
- Perez Carusi, L. C., Beade, M. S., & Bilenca, D. N. (2017). Spatial segregation among pampas deer and exotic ungulates: A comparative analysis at site and landscape scales. *Journal of Mammalogy*, 98, 761–769. <https://doi.org/10.1093/jmammal/gyx007>.
- Piccone, G., Giannetto, C., Caola, G., & Casella, S. (2010). Daily locomotor activity in five domestic animals. *Animal Biology*, 60, 15–24. <https://doi.org/10.1163/157075610X12610595764057>.

- Piccione, G., Giannetto, C., Schembari, A., Gianesella, M., & Morgante, M. (2011). A comparison of daily total locomotor activity between the lactation and the dry period in dairy cattle. *Research in Veterinary Science*, *91*, 289–293. <https://doi.org/10.1016/j.rvsc.2010.12.011>.
- Piovezan, U., Tiepolo, L. M., Tomas, W. M., Barbanti Duarte, J. M., Varela, D., & Marinho Filho, J. S. (2010). Marsh deer *Blastocercus dichotomus* (Illiger 1815). In J. M. Barbanti Duarte, & S. Gonz ales (Eds.), *Neotropical cervidology. Biology and medicine of latin American deer* (pp. 66–76). Jaboticabal, Brazil: Funep/IUCN.
- Pizzio, R., & Bendersky, D. (2018). Base forrajera y potencial de mejora en el NEA. In P. Barbera (Ed.), *Cr a vacuna en el NEA* (pp. 19–44). Ciudad Aut noma de Buenos Aires: Ediciones INTA. ISBN 978-987-521-955-957.
- Pringle, R. M., Young, T. P., Rubenstein, D. I., & McCauley, D. J. (2007). Herbivore-initiated interaction cascades and their modulation by productivity in an African savanna. *Proceedings of the National Academy of Sciences*, *104*, 193–197. <https://doi.org/10.1073/pnas.0609840104>.
- Puechagut, P. B., Politi, N., Ruiz de los Llanos, E., Lizarraga, L., Bianchi, C. L., Bellis, L. M., et al. (2018). Association between livestock and native mammals in a conservation priority area in the Chaco of Argentina. *Mastozoolog a Neotropical*, *25*, 407–418. <https://doi.org/10.31687/saremMN.18.25.2.0.19>.
- Quintana, R. D. (2003). Seasonal effects on overlap trophic niche between capybara (*Hydrochaeris hydrochaeris*) and livestock, and on trophic niche breadths in a rangeland of Central Entre R os, Argentina. *Mammalia*, *67*(1), 33–40.
- Quintana, R. D., & Bolkovic, M. L. (2012). Use of Capybaras in Argentina. In J. Moreira, K. Ferraz, E. Herrera, & D. Macdonald (Eds.), *Capybara* (pp. 345–356). New York, NY: Springer. https://doi.org/10.1007/978-1-4614-4000-0_21.
- Quintana, R. D., Monge, S., & Malv rez, A. I. (1998). Feeding patterns of capybara *Hydrochaeris hydrochaeris* (Rodentia, Hydrochaeridae) and cattle in the non-insular area of the Lower Delta of the Paran  River, Argentina. *Mammalia*, *62*, 37–52.
- Reid, R. S., Bedelian, C., Said, M. Y., Kruska, R. L., Mauricio, R. M., Castel, V., Olson, J., Thornton, P. K., et al. (2013). Global livestock impacts on biodiversity. In H. Steinfeld, H. A. Mooney, F. Schneider, & L. E. Neville (Eds.), *Livestock in a changing landscape, volume 1: Drivers, consequences, and responses* (pp. 111–138). Washington, DC: Island Press.
- Ridout, M. S., & Linkie, M. (2009). Estimating overlap of daily activity patterns from camera trap data. *Journal of Agricultural, Biological, and Environmental Statistics*, *14*, 322–337. <https://doi.org/10.1198/jabes.2009.08038>.
- Ripple, W. J., Newsome, T. M., Wolf, C., Dirzo, R., Everatt, K. T., Galetti, M., et al. (2015). Collapse of the world's largest herbivores. *Science Advances*, *1*, Article e1400103. <https://doi.org/10.1126/sciadv.1400103>.
- Robinson, T. P., Wint, G. R. W., Conchedda, G., Van Boeckel, T. P., Ercoli, V., Palamara, E., et al. (2014). Mapping the global distribution of livestock. *PLoS One*, *9*, Article e96084. <https://doi.org/10.1371/journal.pone.0096084>.
- Rowcliffe, J., Field, J., Turvey, S., & Carbone, C. (2008). Estimating animal density using camera traps without the need for individual recognition. *The Journal of Applied Ecology*, *45*, 1228–1236. <https://doi.org/10.1111/j.1365-2664.2008.01473.x>.
- Sampedro, D. (2018). Tecnolog as de manejo ordenadoras de los sistemas de cr a en el NEA. In P. Barbera (Ed.), *Cr a vacuna en el NEA* (pp. 45–65). Ciudad Aut noma de Buenos Aires: Ediciones INTA. ISBN 978-987-521-955-957.
- Sampedro, D., & Calvi, M. (2018). Caracterizaci n de la ganader a vacuna del nordeste argentino (NEA). In P. Barbera (Ed.), *Cr a vacuna en el NEA* (pp. 8–18). Ciudad Aut noma de Buenos Aires: Ediciones INTA. ISBN 978-987-521-955-957.
- Schieltz, J. M., & Rubenstein, D. I. (2016). Evidence based review: positive versus negative effects of livestock grazing on wildlife. What do we really know? *Environmental Research Letters*, *11*, Article 113003. <https://doi.org/10.1088/1748-9326/11/11/113003>.
- Schoener, T. W. (1974). Resource partitioning in ecological communities. *Science*, *185*, 27–39.
- Soler, L., & C ceres, F. (2009). Breve an lisis sobre la presencia del puma (*Puma concolor*) en la Provincia de Corrientes. *Biologica*, *10*, 67–69. <https://www.museoameghino.gov.ar/publicaciones-revista-biologica.php>.
- Steinfeld, H., Gerber, P., Wassenaar, T., Castel, V., Rosales, M., & de Haan, C. (2006). Livestock's long shadow: Environmental issues and options Food and Agriculture Organization of the United Nations (FAO). *Rome*.
- Stewart, K. M., Bowyer, R. T., Kie, J. G., Cimon, N. J., & Johnson, B. K. (2002). Temporospatial distributions of elk, mule deer, and cattle: Resource partitioning and competitive displacement. *Journal of Mammalogy*, *83*, 229–244. [https://doi.org/10.1644/1545-1542\(2002\)083<0229:TDOEMD>2.0.CO;2](https://doi.org/10.1644/1545-1542(2002)083<0229:TDOEMD>2.0.CO;2).
- Tomas, W. M., & Salis, S. M. (2000). Diet of the marsh deer (*Blastocercus dichotomus*) in the Pantanal wetland, Brazil. *Studies on Neotropical Fauna and Environment*, *35*, 165–172. <https://doi.org/10.1076/snfe.35.3.165.8861>.
- Verd , J. R., Crespo, M. B., & Galante, E. (2000). Conservation strategy of a nature reserve in Mediterranean ecosystems: The effects of protection from grazing on biodiversity. *Biodiversity and Conservation*, *9*, 1707–1721. <https://doi.org/10.1023/A:1026506725251>.
- Verd , J. R., Moreno, C. E., S nchez-Rojas, G., Numa, C., Galante, E., & Halffter, G. (2007). Grazing promotes dung beetle diversity in the xeric landscape of a Mexican Biosphere Reserve. *Biological Conservation*, *140*, 308–317. <https://doi.org/10.1016/j.biocon.2007.08.015>.
- Vila, A. R., Beade, M. S., & Barrios Lamuni re, D. (2008). Home range and habitat selection of pampas deer. *Journal of Zoology*, *276*, 95–102. <https://doi.org/10.1111/j.1469-7998.2008.00468.x>.
- Willms, W., McLean, A., Tucker, R., & Ritcey, R. (1979). Interactions between mule deer and cattle on big sagebrush range in British Columbia. *Journal of Range Management*, *32*, 299–304.
- Yeo, J. J., Peek, J. M., Wittinger, W. T., & Kvale, C. T. (1993). Influence of rest-rotation cattle grazing on mule deer and elk habitat use in east-central Idaho. *Journal of Range Management*, *46*, 245–250.
- Young, T. P., Porensky, L. M., Riginos, C., Veblen, K. E., Odadi, W. O., Kimuyu, D. M., et al. (2018). Relationships between cattle and biodiversity in multiuse landscape revealed by Kenya long-term enclosure experiment. *Rangeland Ecology & Management*, *71*, 281–291. <https://doi.org/10.1016/j.rama.2018.01.005>.
- Zuleta, G. A., Gauto, O. A., Varela, D. M., De Angelo, C., Johnson, B. G., Lor n, D., et al. (2015). *Evaluaciones Ambientales Estrat gicas y Programa de Monitoreo de la Biodiversidad en las Regiones de Mesopotamia y Delta del Paran . Proyecto de Conservaci n de la Biodiversidad en Paisajes Productivos Forestales (GEF TF 090118). Technical report. Buenos Aires, Argentina.*
- Zurita, G. A., Pe'er, G., & Bellocq, M. I. (2017). Bird responses to forest loss are influenced by habitat specialization. *Diversity and Distribution*, *23*, 650–655. <https://doi.org/10.1111/ddi.12559>.