TOWSON UNIVERSITY
OFFICE OF GRADUATE STUDIES

REINTRODUCTION SUCCESS AND ECOLOGICAL ASPECTS OF REINTRODUCED PECCARIES (*Pecari tajacu*) IN THE IBERA NATURAL RESERVE, CORRIENTES, ARGENTINA

by

Cindy M. Hurtado Martinez

A thesis

Presented to the faculty of

Towson University

in partial fulfillment

of the requirements for the degree

Master of Science

Department of Biological Sciences

Towson University
Towson, Maryland 21252

May 2017

TOWSON UNIVERSITY
OFFICE OF GRADUATE STUDIES

THESIS APPROVAL PAGE

This is to certify that the thesis prepared by Cindy Meliza Hurtado Martinez entitled: "Reintroduction success and ecological aspects of reintroduced peccaries (*Pecari tajacu*) in the Ibera Natural Reserve- Corrientes, Argentina" has been approved by the thesis committee as satisfactorily completing the thesis requirements for the degree Master of Science.

Harald Beck Harald Beck 4.27.2017
Dr. Harald Beck Signature Type Name Date

Paporn Thebpanya 4/27/2017
Dr. Paporn Thebpanya Signature Type Name Date

Altrichter Mariana Altrichter April 27th, 2017
Dr. Mariana Altrichter Signature Type Name Date

Ronald R. Swaisgood April 27th, 2017
Dr. Ronald Swaisgood Signature Type Name Date

Janet V DeLany Janet DeLany May 10, 2017
Dean of Graduate Studies Signature Type Name Date

ACKNOWLEDGMENTS

I thank my major advisor Dr. Harald Beck for taking me as his student and guiding me through this project. He has provided me with guidance and support, helped me obtain funding, and discussed research ideas. I also thank the members of my committee, Drs. Paporn Thebpanya, Mariana Altrichter, and Ronald Swaisgood for their intellectual contributions to this project.

This project was funded by the Phoenix Zoo, the Towson University Graduate Student Association, The Biological Sciences Department, and The Conservation Land Trust with the help and support from Ignacio Jimenez, Sofia Heinonen, and Sebastian Di Martino. Also, this project would not have been possible without the support and advice of Emanuel Galetto, who not only help in the field and shared his expertise about mammals and plants from Ibera but also provided accommodations in the field.

Thanks to Alvaro Garcia for his constant support and advice, and Yamil Di Blanco for suggestions and assistance with data analysis. Thanks to field assistants and veterinarians, and the Conservation Land Trust team, especially to Noelia Insaurralde, Antonella Lema, Cristian Schneider, Cecilia Accattoli, Jorge Peña, Nicolas Medrano, Amanda Street, and Carolina Rosas.

ABSTRACT

The collared peccary (*Pecari tajacu*) is distributed from southern USA to northern Argentina; however, in some Argentinean localities it went extinct over 50 years ago. As part of a rewilding project, two peccary herds (one captive-bred family group and one mixed group of rescued or zoo individuals) were reintroduced to the Ibera Natural Reserve, northeastern Argentina. I evaluated initial reintroduction success by assessing survival rates, site fidelity, and behavioral changes; and studied ecological aspects of the reintroduced individuals such as activity patterns, diet, home range, and habitat selection. After quarantine time, 10 individuals (3 males and 7 females) were brought to a pen for a soft-release in June, 2015, and each individual was fitted with a VHF collar. A second herd (5 males and 5 females) was released following the same protocol in May, 2016. I monitored post-release movements of all the individuals to obtain GPS locations every 100 min. The initial reintroduction success was confirmed with high fidelity to release site for the first group and monthly survival rates over 60% for both reintroduced groups. The first group showed a behavioral change from 30% to 52% foraging time and 20% to 15% traveling time, for 2015 and 2016, respectively. Furthermore, diurnal activity patterns were consistent with previous studies in other areas and no effect of feral pig was observed in the activity patterns of the release individuals. The peccaries' home range varied from 1.2 to 2 km^2 and they showed preference for forested areas and the forest edge. These results indicate a successful reintroduction of peccaries in Argentina and I provide recommendations for future reintroductions.

Table of Contents

List of Tables vi

List of Figures vii

Introduction 1

Materials and Methods 6

Results 19

Discussion 26

Management Considerations 41

Literature Cited 70

Curriculum Vitae 97

List of Tables

Table 1. Diet proportion of the first group of released collared peccaries in Corrientes, Argentina, from May 2015 - August 2016........ 44

Table 2. Plant species consumed by released collared peccaries in Corrientes, Argentina, from May 2015 - August 2016........ 45

Table 3. Home range estimates for released collared peccaries in the Ibera Natural Reserve (INR) Corrientes, Argentina 47

Table 4. Main habitat types and number of locations of the released collared peccaries in the Ibera Natural Reserve, Corrientes, Argentina. 49

Table 5. Second order of selection models for the released collared peccaries in the Ibera Natural Reserve, Corrientes, Argentina 50

Table 6. Third order of selection models for the released collared peccaries in the Ibera Natural Reserve, Corrientes, Argentina 51

Table 7. Home range estimates for collared peccaries from other studies 52

List of Figures

Figure 1. Study area, Rincon del Socorro Ranch, Corrientes, Argentina 54

Figure 2. Distribution of the habitat types in El Rincón del Socorro Ranch, Corrientes, Argentina.. 55

Figure 3. Habitat types within the release site in the Ibera Natural Reserve, Corrientes, Argentina.. 56

Figure 4. Pre-release pen for both released groups and circular mobile pen for the soft release of peccaries in the Ibera Natural Reserve, Corrientes, Argentina 57

Figure 5. Camera trap locations in the Ibera Natural Reserve, Corrientes, Argentina for the first and second released peccary group 58

Figure 6. Monthly survival rates for the first and second group of released peccaries in the Ibera Natual Reserve, Corrientes, Argentina. 59

Figure 7. Site fidelity analyses showing mean square distance and linearity index for the released collared peccaries in the Ibera Natural Reserve, Corrientes, Argentina............ 60

Figure 8. GPS locations from released collared peccaries in the Ibera Natural Reserve, Corrientes, Argentina 61

Figure 9. Activity budget of the released collared peccaries in the Ibera Natural Reserve, Corrientes, Argentina.. 62

Figure 10. Comparison of collared peccary group activities' right after release in the Ibera Natural Reserve, Corrientes, Argentina. 63

Figure 11. Collared peccaries’ behaviors for the first group in the Ibera Natural Reserve, Corrientes, Argentina, right after reintroduction (2015) and one year later (2016). 64

Figure 12. Daily activity pattern of released collared peccaries in the Ibera Natural Reserve, Corrientes, Argentina. 2015 (A) and 2016 (B). 65

Figure 13. Comparison of daily activity pattern of the released collared peccaries and feral pig in the Ibera Natural Reserve, Corrientes, Argentina. 66

Figure 14. Feral pig activity pattern in the Ibera Natural Reserve, Corrientes, Argentina during 2015 (A) and 2016 (B). 67

Figure 15. Home range estimation with Autocorrelated Kernel Density Estimator (AKDE) for released collared peccaries in the Ibera Natural Reserve, Corrientes, Argentina.. 68

Figure 16. Coefficients ± 90% confidence intervals for different habitat types in the Ibera Natural Reserve, Corrientes, Argentina. 69

Introduction

In 1992 Kent Redford coined the term “Empty forest” which refers to the extirpation of large mammals and keystone species in tropical forests (Redford 1992). Despite the appearance of intact forests, the lack of large mammals prevents myriad complex animal-plant interactions (Galetti & Dirzo 2003), which can also result in trophic cascading effects. Those effects can negatively impact pollination, density of herbivores, tree recruitment, seed predation and dispersal, and may also cause mesopredator release which refers to the dramatic increase of medium size predators (Wright 2003, Guimaraes et al. 2008, Terborgh et al. 2008, Beck et al. 2013, Jorge et al. 2013, Terborgh 2013, LaPoint et al. 2014). For example, in the Peruvian Amazon, overhunting has impacted seed predation and dispersal, which led to changes in the tree sapling community and species richness (Terborgh et al. 2008).

Rewilding is one conservation approach to mitigate the negative effects of defaunation (Brown et al. 2011, Seddon et al. 2014). The term rewilding initially consisted of reintroducing keystone species, primarily large carnivores, to restore the ecosystem functionality and prevent biodiversity loss (Soulé & Noss 1998, Griffiths & Harris 2010, Griffiths et al. 2010); however rewilding is currently used as returning wilderness and connecting large areas with or without active management or intervention (e.g. reintroductions, Corlett 2016).

Planning is an important component of translocation success, and a large number of factors need to be considered prior, during, and after translocating individuals (Batson et al., 2015; IUCN, 2013). Some of the factors that affect post-release survival and therefore reintroduction success include: intra-specific conflict (Linklater et al. 2011, Shier & Swaisgood 2012), long-distance dispersal away from the release site (Stamps & Swaisgood 2007, Shier & Swaisgood 2012), high predation levels due to predator naïvity (Shier & Owings 2006, Frair et al. 2007, Zidon et al. 2009), disease (Ballou 1993, Viggers et al. 1993), and post-release stress (Teixeira et al. 2007, Linklater et al. 2011, (Dickens et al., 2010). In order to determine whether factors such as these influence post-release success, so that lessons can be learned and applied to future reintroductions, intensive post release monitoring is required (Batson et al., 2015; Germano et al., 2015; IUCN, 2013). These best practices must be followed if reintroductions are to serve their role in restoration of ecosystem functions.

A rewilding process was recently started in Corrientes, Argentina, within the Ibera Natural Reserve (INR), where several species of large mammals are locally extinct (Giraudo & Provedano 2003, Giraudo et al. 2006). The NGO Conservation Land Trust is managing this process with the final objective of restoring all the species that once lived in Ibera, culminating in the reintroduction of jaguars (*Panthera onca*). Before the reintroduction of a top predator can be achieved, several extinct prey species needed to be reintroduced to restore the reserve´s ecological balance (Nicholls 2006). The pampas deer (*Ozotoceros bezoarticus*) and giant anteater (*Myrmecophaga tridactyla*) have already been successfully reintroduced into the area since 2007 (Perez et al. 2013, Di Blanco et al. 2012) followed by the reintroduction of collared peccaries in 2015.

Previous peccaries’ reintroductions efforts in other countries have not been published but were described in reports or dissertations (Litondo 1993, Porter 2006, Toone et al. 2003, Campos pers. com). Reintroductions of collared peccaries in Texas were successful in establishing a population (Litondo 1993, Richter 2012); however, the reintroduction of the Chacoan peccary in Paraguay was unsuccessful (Toone et al. 2003), and no follow up information was provided after the release of two individuals in the Encrucijada Biosphere Reserve in Mexico (Rivadeneyra & Naranjo 2004).

The objectives of this project were to assess reintroduction success and describe the behavioral adjustments and ecological interactions that the reintroduced individuals had with their environment. I evaluated the initial reintroduction success of two groups of collared peccaries reintroduced in the Ibera Natural Reserve by quantifying survivorship, fidelity to release site, and behavioral changes. Also, I investigated spatial and ecological aspects of the reintroduced individuals, including activity patterns, diet, home range, and habitat selection, to provide recommendations for future collared peccary reintroductions.

I used survival as one measure of initial reintroduction success (Seddon 1999, Jule et al. 2008, King et al. 2012) and to identify the time period of the reintroduction process in which mortality was higher: transportation and quarantine, acclimation in the release pen, and post-release period. Moreover, I evaluated site fidelity, inferred when an organism's movements are limited to a smaller area compared that expected from random movement (Munger 1984), which indicates the individual or group has a constant home range. Thus, I used site fidelity as another measure of initial reintroduction success (Yott et al. 2011, Berger-Tal & Saltz 2014, Jackson et al. 2016), indicating possible long-term persistence of the individuals in the release area (Flanagan et al. 2016). Establishment in the release site is a good indicator of initial reintroduction success since individuals that disperse away from the study area have a higher probability of mortality, primarily because of predation or starvation (Biggins et al. 1999, Calvete & Estrada 2004, Stamps & Swaisgood 2007, Sarkar et al. 2016).

Moreover, animals adjust their behavior after environmental changes to cope with their new environment and these changes can decrease foraging, resting time, and social interactions causing low reproduction success and alter the stability of social groups (Rubenstein 1991, Dunbar 1992). Changes in behavior following release into a novel environment can also be used as an indicator to evaluate whether an animal is “settled” and interacting with its environment in a way that will promote fitness (Kleiman 1989, Teixeira et al. 2007), with the expectation that early behavior profiles will be more maladaptive and potentially indicative of stress, but these behavior patterns will gradually change to approximate that observed in wild animals of the same species. Altrichter et al. (2002) used activity budget as a measure to determine foraging stress in a population of white-lipped peccaries in Costa Rica, where individuals at high levels of stress (determined by reduced food availability) increased time traveling and decreased time engaging in social interactions. For these reasons, I compared activity budget as a function of time post-release as a measure of determining short-term reintroduction success.

Intraspecific competition, especially with introduced species, may negatively affect reintroduction outcomes. Because feral pigs (*Sus scrofa*) alter soil structure and processes, decrease vegetation cover and plant communities, and can compete with native fauna for resources (Barrios-Garcia & Ballari 2012), their presence at the release site may compromise reintroduction success. Even though low niche overlap between feral pig and peccaries was observed in the Atlantic forest of Brazil (Desbiez et al. 2008), previous studies in Argentina showed a negative effect of feral pigs on pampas deer density (Perez Carusi et al. 2009). One was to evaluate potential competition between feral pigs and reintroduced peccaries is to measure daily activity patterns to determine if there is temporal niche separation. Thus, I also studied the activity pattern of feral pigs to determine if they had a negative effect on the released collared peccaries.

By evaluating different post-release measures and changes through time, I will be able to determine the best method for monitoring initial success of peccary reintroductions. Also, with these analyses I will determine if the released individuals are adapting to their new environment. The results of this study will help improve future reintroduction protocols by establishing best practices that will help increase survival rates and select appropriate release sites. Sustaining healthy peccary populations will not only restore missing ecosystem functions including seed dispersal and predation, but is also a critical precursor for the successful reintroduction of other keystone species.

Materials and Methods

Study area

I conducted this study at the Ibera Natural Reserve (INR), which is located in the Corrientes province, of northeastern Argentina (28°32’S, 57°10’W, Fig. 1). The reserve consists of provincial (government owned) and private land which is primarily used for extensive cattle and sheep ranching. Overall, INR encompasses 13,900 km^2 of various habitats within the Chaco Biogeographical Province (Morrone 2001, Canziani et al. 2003). The mean annual temperature ranges from 16 to 27°C (Neiff & Poa de Neiff 2006). During summer months (December to March) the temperature can reach up to 44°C, whereas the in the winter temperature may drop to -2°C (Beccaceci 1994, Neiff & Poa de Neiff 2006). Annual precipitation varies from 1300 to 1700 mm per year and during the summer months it varies from 106 to 442 mm (Ubeda et al. 2013).

The common native fauna within the INR includes capybaras (*Hydrochoerus hydrochaeris*), marsh deer (*Blastocerus dichotomus*), gray brockets (*Mazama gouazoubira*), crab-eating fox (*Cerdocyon thous*), various small mammals including rodents, greater rheas (*Rhea americana*), and strange-tailed tyrants (*Alectrurus risora)*. Invasive species, such as feral pigs, axis deer (*Axis axis*), and red deer (*Cervus elaphus*) also occur in this area. This research took place on the Rincon Del Socorro Ranch (RSR), which is 124 km^2 located in the southeastern area of the INR, and is owned and managed by The Conservation Land Trust (www.theconservationlandtrust.org). RSR is a conservation area without livestock, and it is where the rewilding project started (Perez et al. 2013, Di Blanco et al. 2015). Although hunting of native species is forbidden, invasive species hunting is allowed and strictly controlled.

RSR contains five different habitat types: hygrophilous forest, closed savanna, open savanna, grassland, and palmar (Fig. 2). The hygrophilous forest (Fig. 3A) is formed along small seasonal streams, typically gallery forest; the canopy reaches 15 m with some emergent trees, such as *Enterolobium contortisiliquum,* which reach up to 25 m. This habitat also contains vines and epiphytes (Tressens et al. 2002). The understory is dominated by *Pavonia sepium* and *Dicliptera tweediana* and it shows different impacts of previous anthropogenic disturbances such as debris from old farms, wells, and fence wires. Tree and palm species are typical of the Atlantic Forest and the most common species include *Patagonula americana*, *Phytolacca dioica*, and *Syagrus romanzoffiana* with some emergent trees like *Tipuana tipu*, and *Enterolobium contortisiliquum* (Tressens et al. 2002).

The closed savanna habitat (Fig. 3B) includes trees and herbaceous species within forest patches < 200 m^2 (Tressens et al. 2002). Dominant trees species include *Prosopis affinis, P. nigra* and *Acacia caven* with important cactus species like *Cereus argentinensis* and *Opuntia cardiosperma* (Tressens et al. 2002, Di Blanco et al. 2015). Open savannas (Fig. 3C) are open areas, devoid of forest, with medium to short grass species, some scattered palms (*Copernicia alba),* and two legume trees (*Prosopis affinis* and *Acacia caven*) sparsely distributed (Di Blanco et al. 2015). Also, it is characterized by herbaceous vegetation and shrubs with predominant species such as *Elyonurus muticus* and *Andropogon lateralis* (Tressens et al. 2002, Di Blanco et al. 2015). Grasslands (Fig. 3D) are seasonally flooded areas dominated by *Andropogon lateralis* and cover the southern portion of the study site (Di Blanco et al. 2015).

Palmar habitats (Fig. 3E) are smaller areas usually bordering the wetland. They are characterized by constant flooding and evident palm aggregations of caranday, *Copernicia alba* and Pindo, *Syagrus romanzoffiana*.

Study species: Collared peccaries

Collared peccaries are social animals with a wide distribution range, from southern USA to northern Argentina. Thus, peccaries are found in a great variety of habitats ranging from deserts, tropical moist forest, cloud forest, tropical dry forest, thorn-forest, oak grasslands, palm savannas, and mangroves (Taber et al. 2011). They form small herds of 6-30 individuals composing of females, males, and juveniles (Taber et al. 2011). Peccaries also have a well-developed sense of olfaction, presumably to locate food. Because of the large distribution range, they feed from at least 150 fruit species (Beck 2006). Their diet includes roots, leaves, invertebrates, fruits, and seeds; and it varies depending on resource availability and seasons (Barreto et al. 1997, Perez-Cortez & Reyna-Hurtado 2008, Taber et al. 2011).

Seasonality may drive differences in resource availability which can produce behavioral changes in the peccaries' activity budget (Bigler 1974, Altrichter et al. 2002, Carrillo et al. 2002, Keuroghlian & Eaton 2008). In addition, females may delay reproduction or not reproduce at all because of stress caused by food scarcity (Lochmiller et al. 1986). Females normally mate with several males in the group, with the males establishing a hierarchy (Taber et al. 2011). Collared peccaries reproduce year round and may have one to four young, while two being most common (Taber et al. 2011).

Currently, collared peccaries are classified as least concern by the International Union for Conservation of Nature (Gongora et al. 2011) and as vulnerable by

Argentinean red list (Ojeda et al. 2012). Although collared peccaries were frequent during the 1820's in the Corrientes province, they went locally extinct over 50 years ago because of overhunting and habitat loss (Giraudo et al. 2006). Hunting is now regulated in Corrientes and also in the Ibera Natural Reserve (Di Blanco et al. 2015).

One potential threat for the release individuals is the presence of feral pig in the study area. Feral pigs were introduced in Argentina for hunting purposes during the 20th century, and its rooting behavior have impacted perennial species (Cuevas et al. 2010), affected seed survival and regeneration of seedlings (Sanguinetti & Kitzberger, 2010), and affected population density of vertebrates (Perez Carusi et al. 2009).

Reintroduction protocol

Because no wild genetically similar population exists in the Corrientes province, the reintroduced individuals were captive-born and confiscated animals from the adjacent Yungas biogeographical region from the Tucuman and Salta provinces. The entire translocation process, vaccinations, veterinary and quarantine monitoring were conducted by the Conservation Land Trust staff.

First collared peccary group

Ten individuals (three adult males and seven adult females) were obtained from the Horco Molle Experimental Reserve located in the montane forest of the San Miguel de Tucuman province in northwestern Argentina. The peccaries were born and kept in a semi-captivity 18 ha fenced area. They fed from native plants such as camalote (*Eichornia* sp.), repollo de agua (*Pistia* sp.), roots and seasonal fruits such as mora (*Rubus imperialis*) and guayaba (*Psidium* sp.). Furthermore, as supplemental food they were given corn, carrots, potatoes, squash, and balanced swine food.

Ten peccaries were sedated and transported to a quarantine facility in the Corrientes Province on March 3rd, 2015. Also, a second transport was made for two additional individuals (one male and one female) on March 10th, 2015. Upon arrival, the animals received the required veterinarian examinations (to rule out Tuberculosis, Brucelosis, Leptospirosis, among others), and were equipped with a MOD 500 or MOD 400 VHF radio-collared transmitter (Telonics Inc., Mesa, Arizona), numbered ear tags (Allflex USA Inc., Texas) and subdermal microchips (Rosenbusch S.A., Ciudad Autonoma de Buenos Aires).

On April 25th, 2015, after 52 days in quarantine, ten individuals were transported into a 0.3 ha pre-release enclosure located within their release site (Fig. 4). As part of the soft-release protocol, the animals remained in the pen for 44 days to acclimate to their new local conditions (Kleiman 1989, Mitchell et al. 2011, IUCN 2013). This protocol was done in an attempt to improve survival rates of the peccaries after release (Davis 1983, Wanless et al. 2002, Porter 2006). During the acclimation period, the peccaries received approximately 1 kg of supplemental food per individual, which consisted of corn, balanced swine food, and some native fruits. The supplemental food was provided once a day, around 4 pm to reduce the possibility of post-release dispersal, which could be associated with high mortality rates (Stamps & Swaisgood 2007). This schedule was followed to train the animals on when food was going to be available after release.

Second collared peccary group

A second group of ten individuals (five adult females, four adult and one juvenile male) were translocated on November 19th, 2015, from the Estación de Fauna Autóctona de Salta, a rescue center in the Salta province. These individuals were not related; some had been kept as pets for several years and others were recently captured from the wild. Thus, they did not form a cohesive group, which may have been the reason for frequent fights and injuries prior to release. As opposed to the first group, this group was transported without sedation and was kept in a quarantine facility for 71 days. On January 29th, 2016, four adult females and two adult males were moved to the pre-release pen and were given supplemental food as described above.

After 17 days of acclimation, a first release attempt was made on February 14th, 2016. This attempt was unsuccessful because fights between both groups of peccaries over the supplemental food occurred frequently. A second release attempt was made on March 11th, 2016. Both release attempts were made from the same pen where the first group was kept and were released in the same area. However, after continuous fights between the two groups, one male from the second group was left with the first group on April 9th, 2016. The remaining individuals of the second group (one male and two females) were recaptured and later transported on April 30th, 2016 to a nearby area (8 km away) into a circular 4 m radius pen (Fig. 5) where they stayed for four days before the third and final release on May 3rd, 2016.

Radio tracking

The equipment consisted of one TR-4 164-168 MHz receiver, one RA-2AK VHF-antenna, and one Garmin 62sts GPS. The receiver and antenna were Telonics (Telonics Inc., Mesa, Arizona). To locate each individual, the “homing in” procedure was followed, which consisted of finding and approaching the animal until sighting to later obtaining a GPS location (White & Garrott 2012).

For each sighting I registered coordinates, date, time, behavior, and habitat type. Two radio tracking periods were used: intense radio tracking to quantify changes in activity budget, habitat selection, and home range patterns; and constant radio tracking to assess survival and changes in home range through time. The period of intense radio tracking was made between June and August 2015 and May and August 2016 right after release. In this period, I radio tracked all of the radio-collared individuals 5 days a week for a minimum of 4 hours per day. Moreover, in the constant radio tracking period, experienced park guards who work for the CLT, tracked the individuals to obtain at least two fixes a week.

Survival

Every mortality date was registered and when possible the body was recovered. I used number of death animals per month to create a binary matrix and calculate monthly survival rates using the Kaplan-Meier known-fate model with the logit function, in the Mark software version 8.1 (White & Burnham 1999). I did not consider age or sex in the analysis, only group identity. This analysis accounted for censored and staggered individuals (Pollock et al. 1989). I analyzed 18 monthly intervals for survival of the first group, starting immediately following translocation. For the second group, I analyzed ten months after translocation. The lower value of Akaike information Criterion corrected for small sample sizes (AICc) was used for model selection (Burnham & Anderson 2002, Mihoub et al. 2013).

Site fidelity

Site fidelity (SF) was calculated using the rhr package (Signer & Balkenhol 2015) in the R 3.3.1 software (R Core Team 2016) which followed the methods proposed by Danielson and Swihart (1987), and Spencer et al. (1990) with 100 bootstraps. SF was considered to exist when a significantly smaller difference was found between the Mean Square Distance (MSD) and the Linearity Index (LI) of the observed locations compared to generated random locations (Sarkar et al. 2016). A histogram generated from the distributions of random locations and the peccaries' locations provided a red dashed line indicating the alpha level of 0.05 (Signer & Balkenhol 2015). A red solid line below the dashed line indicated SF. Also, no site fidelity was considered if the individual left the study area or established a home range away from the INR.

Activity budget

To test the null hypothesis of no change in the activity budget through time, I recorded the activities of the each group during intensive radio-tracking. I used the scan sampling method with continuous recording (Martin & Baterson 2007) considering six mutually exclusive activities. 1) Foraging, described as the animal searching for food followed by active rooting, chewing, and eating, 2) Resting, when the animal was laying or sitting on the ground (not engaged in any other behaviors), 3) Traveling, referred to the animal’s movements with the group or by itself, following a specific direction, going from one foraging patch to another, resting site to foraging site, etc. 4) Walking, when the individuals is moving at a slower pace than when traveling and is not engaged in rooting or chewing. Social interactions were also recorded and divided in two groups, affiliative and aggressive behaviors. Affiliative behaviors were considered when the animals were playing, licking, mating, rubbing or smelling another member of the herd (Bissonette 1982). Aggressive behaviors included fights, aggressive bites, chase, vocalizations, and gapes (Bissonette 1982). The final behavioral category was “Others”, and considered any rare behavior different from the explained above (i.e. wallowing, drinking water).

For the data analyses, I only considered the days when radio tracking occurred more than three hours a day and when at least three individuals were recorded per scan. Furthermore, considering all the scans per day, I calculated the proportion of time spend in each behavioral category (Altrichter et al. 2002). Therefore, the sample unit (N) refers to daily activity budgets and number of observation days (Altrichter et al. 2002), which will allow for inferences in this group and not the species (i.e. pseudoreplication, Martin & Bateson 2007).

I used the non-parametric Wilcoxon rank test for paired samples to identify differences in mean percentages of time spent in each behavioral category, immediately following reintroduction versus one year later (Resende et al. 2014, Gine et al. 2015). Furthermore, to test for behavioral differences following reintroduction, I compared the activity budget among groups using the Kruskal-Wallis rank test, followed by the post-hoc Nemenyi test for multiple pair-wise comparisons (Keuling et al. 2009) to assess if the individuals behave similarly after release (i.e. all show homing behavior, more time traveling).

Activity patterns

Camera trap data can be a powerful tool for evaluating species differences in temporal activity patterns, and provide insights into possible temporal niche differentiation (Tobler et al. 2009). The seasonal activity pattern variations of the reintroduced individuals were investigated using 12 remote cameras (three Covert Illuminator, Lewisburg, Kentucky; one M-550 Moultrie, EBSCO Industries Inc., Birmingham, Alabama; three PC 800 Hyperfire, Reconyx Inc., Holmen, Wisconsin; and five Strike Force Browning, Prometheus Group, Birmingham, Alabama). From June 8th, 2015, to December 31st, 2015, five remote cameras were set near the release site of the first peccary group. From May 30th to August 20th, 2016, a total of 12 camera traps were set in the study area, across the ranges of both reintroduced groups (Fig. 5). The cameras were equipped with a motion-heat sensor and were set to take three photos per trigger at 5 min intervals. Each camera was separated by at least 200 m and was set in areas commonly utilized by peccaries. Camera locations were changed to follow the peccaries' movement through time.

All the photos were downloaded and analyzed using Camera Base 1.7 software (Tobler 2015) to quantify peccaries' activity pattern after release and to investigate changes through time. In addition, feral pig photos were analyzed to determine if there was temporal overlap with the released individuals.

Even though previous activity pattern research has used 30 min difference as time to independence for wild populations (i.e. Norris et al. 2010, Galetti et al. 2015), I evaluated activity patterns considering independent events as photographs in all the cameras separated by at least one hour (Gomez et al. 2005, Tobler et al. 2008, Bowkett et al. 2007, Martin et al. 2016). This approach was taken because of the certainty that only one peccary group exists within each of the release areas.

Furthermore, activity was quantified using the method proposed by Rowcliffe et al. (2014), which consisted of fitting a kernel circular distribution to the data to calculate proportion of time active. I calculated standard errors with non-parametric bootstrap (Rowcliffe et al. 2014). The bandwidth was selected as the (optimal bandwidth *2) which tends to have less bias with large sample sizes (Ridout & Linkie 2009, Rowcliffe et al. 2014). Comparisons of activity patterns between the two reintroduced groups and between collared peccaries and feral pig were made using the Wald test for proportion of time active, and randomization tests for comparisons between activity patterns over time and between species (Rowcliffe et al. 2014). I used R software version 3.3.1 and the packages circular (Agostinelli & Lund 2013) and activity (Rowcliffe 2016) for all the analyses.

Home range

For the spatial analysis, I used locations that were at least 100 min apart, following Keuroghlian et al. (2004), to reduce autocorrelation. I used R software version 3.3.1 (R Core Team 2016) to calculate monthly and seasonal home range (HR) estimates, using the minimum convex polygon (MCP) method (Mohr 1947, Calenge 2006), the fixed kernel density estimator (FKDE) method (Kernohan et al. 2001), and the autocorrelated kernel estimator (AKDE) method (Kie 2013). For the first method, I calculated 100% and 95% MCP to allow comparisons with previous studies (i.e. Taber et al. 1994, Ilse & Hellgren 1995a, Ticer et al. 1998, Richter 2012).

Moreover, FKDE was also calculated to compare HR estimates with recent studies (i.e. Keuroghlian et al. 2004, Reyna-Hurtado et al. 2009). I calculated 95% and 90% kernel as a more robust estimator of home range (Borger et al. 2006). In addition, 50% fixed kernel was calculated to identify core areas (Reyna-Hurtado et al. 2009, de Almeida Jacomo et al. 2013). Two bandwidths were selected as smoothing parameters for

FKDE, the Least Square Cross Validation (hlscv) and the reference bandwidth (href). The hlscv was selected for comparisons with other studies (Keuroghlian et al. 2004) and href to compare home ranges that tent to be underestimated when using hlscv (Worton 1989, Schuler et al. 2014, Fleming et al. 2015). Additionally, I calculated the autocorrelated fixed kernel density estimator method (AKDE), which is ideal for telemetry or GPS data because it is not affected by low sample sizes or autocorrelation (Fleming et al. 2015).

Habitat selection

Vegetation cover maps were obtained from the Conservation Land Trust database and used to calculate habitat selection at two spatial scales; second and third order selection (Johnson 1980). Second order selection includes all available habitats for a given population, whereas third order selection includes habitats within the animal's home range (Johnson 1980, Lesmeister et al. 2009, Reyna-Hurtado et al. 2009, Di Blanco et al. 2015).

To study the habitat selection of the reintroduced individuals, I used design II and III following Manly et al. (2002). Design II was used for second order selection (population level) while design III was used for third order selection (individual or group level). For design II, I used the largest dispersal distance to calculate a polygon of habitat availability, under the premise that each individual could have dispersed at least that much from the pre-release pen and therefore, all that area was available for the remaining released individuals.

For design III, I used each group's home range (100% MCP) with a buffer of 50 m to obtain a polygon of habitat availability (Manly et al. 2002). The locations were obtained from the VHF radio tracking, similar to the home range methodology. For second order selection, 5000 random locations were generated to represent availability while 2000 locations were used for third order selection (Di Blanco et al. 2015, Johnson et al. 2006).

The package MASS (Venables & Ripley 2002) and AICmodavg (Mazerolle 2016) from R software (R Core Team 2016) were used for these analyses. Resource selection functions (RSF) are generally used as the proportional probability of use of a resource (Marzluff et al. 2004, Millspaugh et al. 2006, Sheppard et al. 2010). One way to evaluate RSF is by using logistic regression in a generalized linear model where used locations are compared to available ones (Manly et al. 2002). The variables included in the global model were habitat type, distance to the edge of the hygrophilous forest (DFE), distance to main road (DR), distance to the edge of the wetland (DW), distance to the edge of the palmar habitat (DP), and distance to pre-release pen (DRP).

I generated the model with all the used independent locations from the first release group and random availability locations (training data) and used the independent locations of male 1 and male 2 to validate it (test data). Model selection was made using the Akaike Infrmation Criterion (AIC) which considers the number of parameters in the model and the significance of coefficients (Burnham & Anderson 2002), indicating that the lowest AIC value yields the best model. Selection was considered when the 90% confidence intervals did not include zero (Di Blanco et al. 2015).

Furthermore, validation of the best model was made following Di Blanco et al (2015). I compiled training and test data and predicted their probability of selection. These values were divided into 10 quantile bins and the frequency of use was calculated for each bin. A Spearman rank correlation was used to test the relationship of the bin ranks and the frequency of use. The model was considered to be valid if r > 0.7 (Boyce et al. 2002, Di Blanco et al. 2015).

Results

Survival

Of the initial 12 individuals of the first group, six died (two during transport and quarantine, two during acclimation, and two after release). During transport, two individuals had problems with the anesthesia and arrived dead. Therefore, one male and one female were brought from the same area to the quarantine as substitutes for the ones that died. However, these animals died during acclimation because of infected wounds caused by numerous fights with other members of the group. Another female died one month after release because of pneumonia; and the last male died one year after translocation because of territorial fights with individuals from the second group. Monthly survival rates for the first group varied from 0.83 (SE 0.11) to 1, with lower survival during the translocation and acclimation period (Fig. 6A). However, when considering only monthly survival rates after release, they varied from 0.88 to 1 (SE 0.12) (Fig. 6B).

From the initial ten individuals in the second group, six died (three during quarantine, unknown causes, and three after release) two were taken back to captivity and two males survived. These three individuals that died after release, left the group and went on their own to different locations close to the edge of the wetland where two of them died probably from drowning and the other animal was not found (only the collar). From the two individuals taken back to captivity, one male was injured from constant fights during quarantine and was kept there and one female that had been a pet for many years left the study area and was found near the road. Upon return she constantly left the release site and followed people. Thus, she was taken back to the quarantine facility and not reintroduced.

Only two males from the second group survived until the end of this study. One was released with the remaining females from the first group; and the other stayed in the second release area (male 2). The monthly survival rates for the second group varied from 0.66 (SE 0.27) to 1 with lower survival during post-release (Fig. 6A). However, when considering only monthly survival rates after reintroduction, they varied from 0.83 to 1 (SE 0.15) (Fig. 6B).

Site fidelity

For the first release group of peccaries (6 individuals), the mean square distance (MSD) from their center of activity was significantly different from random locations, however the Linearity Index (LI) was non-significant (Fig. 7A). One year later the same trend was maintained (Fig. 7B). Even though LI was not significantly different from random locations in both years, the animals remained within 2 km of the release site. When considering the data from both years, there is clear site fidelity by both indices (Fig. 7C).

Only one sub-adult male of the first group, male 1, dispersed during the first week after release and had several exploratory movements (Fig. 8A). After the first month of release, it established 8 km from the release site but still within the study area and showed site fidelity (Fig. 7D).

During 2016, all released individuals of the second group that dispersed at least 4 km from their release site did not survive. The only surviving male (male 2) remained in an area closer to the pre-release pen (2.8 km was the greatest distance from release pen, Fig. 8B) and also showed site fidelity according to the MSD but no LI (Fig. 7E).

Activity budget and diet

After 393.18 observational hours (100 h in 2015 and 293.18 h in 2016), I obtained 3422 scan entries for the released groups to determine changes in the activity budget throughout time. Right after reintroduction, the group's primary activity was Foraging (30% to 52%), followed by Resting (23% to 37%), Walking (14% to 23%), Traveling (2% to 16%), Social Interactions (0.5% to 2%), and Others (1.1% to 1.8%) (Fig. 9).

The comparisons of behavior between groups showed significant differences for Foraging (K-W= 11.57, $p < 0.01$), Traveling (K-W= 6.1058, $p < 0.05$), and Walking (K-W= 12.98, $p < 0.01$), Resting and Other behaviors were non-significant among groups (Fig. 10). The multiple comparison analysis indicated significant differences between the group's foraging behaviors compared to both solitary males ($p < 0.05$). Both males spent most of their time foraging (Fig. 10). For Walking and Traveling, the group proportion of time spent was significantly higher than male 2 ($p < 0.05$).

During 2015, I recorded 611 foraging scans for the group and male 1, and identified the food items for 255 events. The diet composition was similar between both groups; over 60% of their diet consisted of roots, followed by leaves (20% to 25%), and fruits (Table 1). Furthermore, in 2016, I identified 780 of 1426 foraging scans and recorded new items to the peccaries' diet. The group's diet composition consisted primarily of leaves (65%), and in lower proportion, fruits, flowers, roots, vines, and meat (Table 1). Scavenging was observed from all individuals; when they were feeding on dead feral pigs and capybaras. Male 2 fed primarily on fruits (71%), but also leaves and roots (Table1). Overall, 31 species of plants were eaten by the released individuals (Table 2).

When comparing the group's activity budget immediately after reintroduction to a year later, Foraging and Traveling behaviors were significantly different ($W = 848.5$, $p < 0.01$ and $W = 162$, $p < 0.01$, respectively). Traveling time was greater in 2015, while Foraging time increased for 2016 (Fig. 11). Also, for 2015 only 21 social interactions were registered for the first group and 19 of them were aggressive, the other two events were mating and rubbing behaviors. Whereas in 2016, more affiliative interactions were registered (66 of 110) mostly between the females, but no mating behavior was observed. Thus, the proportion of affiliative interaction increased during 2016 ($X^2 = 17.99$, d.f. = 1, $p < 0.1$).

Activity patterns

In total 1617 independent records for peccaries and feral pigs were obtained from 1837 camera days. In 2015, I obtained 635 independent events, 362 records for the first reintroduced group of collared peccaries and 273 for feral pigs. In 2016, 481 were independent events of the first group of peccaries, 376 from male 2, and 125 records of feral pig. The activity period of the collared peccaries right after release was concentrated in the afternoon without any defined peak of activity (Fig. 12A). However, after one year of release, during the same season, at least two peaks of activity were observed: one peak in the early morning and the other in the late afternoon (Fig. 11B).

Even though there were no significant differences in the activity of peccaries right after release compared to a year later ($W = 1.11$, $SE = 0.05$, $p = 0.294$), the proportion of time active during the first three months after release, was slightly greater when compared with one year after release, 52% (SE = 0.04) and 48% (SE = 0.09), respectively. When comparing activity between the first and the second group (male 2) in 2016, no significant differences were observed (W = 1.28, SE = 0.06, p = 0.26).

The comparison of activity between species did not show any significant differences during these two years (Fig. 13). Furthermore, the activity patterns of feral pig were similar in 2015 and 2016 (W = 2.8, SE = 0.11, p = 0.094) showing two evident peaks: one in the early morning and the other one in the late afternoon (Fig. 14) similar to the activity of the release peccaries in 2016.

Home range

A total of 482 telemetry recordings were obtained, 259 for the first peccary group, 68 for male 1, and 155 for male 2. The home range estimates varied greatly across the different methods used. The 95% kernell density estimator (KDE) using the hlscv as selected bandwidth always gave the smallest home range size, without considering 50% KDE for core areas, while 100% MCP and AKDE gave the largest estimates (Table 3).

During the first three months after release (2015) and with 112 fixes, the first group had a home range that varied from 0.25 km^2 (95% KDE, hlscv) to 1.59 km^2 (AKDE). A year after release, in 2016, with 147 fixes, the group´s home range varied from 0.12 km^2 (95% KDE, hlscv) to 3.54 km^2 (AKDE). The MCP and KDE with href showed that the home range in the second month after release was larger compared to the first and third month; however, with KDE (hlscv) the estimate was smaller (Table 3). All methods indicated that the peccaries' home range decreased in 2016 except for the AKDE, which showed an increase in home range from 1.59 ± 0.39 km^2 in 2015 to 3.54 ±

1.36 km^2 in 2016 (Fig. 14). Moreover, when using 259 fixes from 2015 and 2016 the home range was estimated to be 8.9 ± 1.7 km^2 (Fig. 15).

The monthly home range for male 1, the male that dispersed from the first group, showed a gradual decrease (Table 3). Its home range varied from 0.04 km^2 (95% KDE hlscv) to 6.42 km^2 (95% KDE href) for the first three months after release (Table 3). Male 2, the only surviving individual from the second released group, had a home range that varied from 0.49 km^2 (95% KDE hlscv) to 3.7 km^2 (95% KDE href) during the first three months of release.

Habitat Selection

The study area comprised a total of 117.1 km^2 of which the grassland habitat was the most extensive one (Table 4). Most of the locations, obtained from radio tracking, were concentrated in the hygrophilous forest and Closed savanna habitat type even though they only represent 22.2 % of the total area available (Table 4).

After testing for multicollinearity, distance to palmar habitat (DP) and distance to the wetland (DW) were removed from the global model because they were highly correlated with distance to forest edge (DFE). The best model for second order selection included all the other covariates tested (Table 5). The next best model had a AIC greater than 2 which suggests that even though it has less parameters it is not a better model (Table 5). Validating the model with male 1 and male 2 showed good model fit ($r = 0.81$). Using the Open savanna as the intercept for habitat types and 90% confidence intervals, the peccaries showed positive selection for only the hygrophilous forest (Fig. 16A).

The best model for third order selection, using the first group localities as training data, included habitat type, distance to road, distance to forest edge, and distance to release pen (Table 6). The 90% confidence intervals of the habitat coefficients showed negative selection for the grassland habitat type (Fig. 16B). However, when validating the model with male 1 observations it showed low model fit (r = 0.52). Similarly, low model fit was observed when validating the model with male 2 observations (r = 0.2).

Therefore, another model was generated using male 1 observations as training data and was validated with male 2 observations. The best model showed habitat type and distance to release pen as good predictors (Table 6). Also, validating the model with male 2 observations showed good model fit, r = 0.9 (Table 6). Moreover, using the Open savanna as the intercept for habitat types and with 90% confidence intervals, the peccaries showed negative selection for the grassland habitat type and marginally for the hygrophilous forest (Fig. 16C).

Discussion

Survival as a measure of initial reintroduction success

The purpose of species reintroductions is to establish a wild viable population back to parts of their historical range (Armstrong & Seddon 2008, Schaub et al. 2009, IUCN 2013). To accomplish this goal, years of effort, resources, and data collection are needed, especially for large mammal species (Dodd & Seigel 1991, Fischer & Lindenmayer 2000, Seigel & Dodd 2002, Saltz et al. 2000, Moseby et al. 2011). Therefore, it is difficult to determine reintroduction success (Dodd & Seigel 1991, Seigel & Dodd 2002, Armstrong & Seddon 2008), and in many cases unsuccessful projects are not documented nor published (Fischer & Lindenmayer 2000). Sarrazin (2007) suggested three phases to determine reintroduction success depending on the time after release: establishment, growth, and regulation. Thus, it is important to quantify the establishment of released individuals within the study area, their survival rates, and other critical factors including behavior and resource availability (Calenge 2005, Sarrazin 2007, Armstrong & Seddon 2008, Yott et al. 2011, Berger-Tal & Saltz 2014).

In ungulate reintroductions, the first month after release is critical because the highest mortality occurs typically during this time period; thus, individuals surviving the first month have a greater probability of success (Bedin & Ostrowski 1998, Slotta-Bachmayr et al. 2004, Calenge et al. 2005, Rosatte et al. 2007). The first group of released peccaries in the Ibera Natural Reserve had a high rate of survival after release 75% over the study period and monthly survival rates of 86% during the first month after release and later maintained at 100% for the first year. These high survival rates of the first group can be attributed to the soft release protocol (Porter 2006, Hardman & Moro

2006, Ryckman et al. 2010), social-cohesion of the family group (Porter 2006, Gusset et al. 2006, Blumstein et al. 2009) and lack of competition with other peccaries. On the other hand, the second group showed lower success after release with 40% of the animals remaining in the study area and monthly survival rates from 66% to 100%. The main reason of this lower success may be the result of the low familiarity and lack of a cohesive social structure and each individual's background. A previous study of collared peccary reintroduction in Texas, USA, assessed differences in survival between soft vs. hard release, and related vs. non-related social groups. This study showed higher survival in related social groups that were soft released (Porter 2006). It also showed that fidelity to release-site and survival decreased in the unrelated family groups, whereas related individuals survived longer, dispersed less, and formed larger family groups (Litondo 1993, Porter 2006).

The importance of familiarity and cohesive social groups has also been demonstrated in captivity, where studies have shown removing individuals from a social group, either related or long-time members of the herd, will change the social structure and risk individual acceptance by the herd if the animal were to return (Bastos da Silva et al. 2014). The conflict created when mixing or adding individuals to a herd can result in death of the new animal or other members of the herd (Bastos da Silva et al. 2014). Maintaining social relationships is also critical for solitary mammals (i.e. maintaining territorial neighbors) to reduce territorial fights and increase reintroduction success (Shier & Swaisgood 2011). In this project, because of aggressive interactions, two peccaries died during acclimation after been reunited with their group; and one male from the first group died after constant fights with the new members of the second group. This suggests that despite the availability of large areas for the individuals to occupy, territorial behavior is an important factor to consider.

Another factor to consider in the modest post-release mortality rates observed in this study is that the lack of predators at the release site which may have played a crucial role and should be considered in future reintroductions; captive born or unexperienced animals are naïve and will be at higher risk of predation (Priddel & Wheeler 1997, Frair et al. 2007, Moseby et al. 2011). A reintroduction attempt of Chacoan peccary (*Catagonus wagneri*) in Paraguay in 1999 resulted in an almost 90% mortality within the first 4 months, and 50% of those were caused by mountain lion (*Puma concolor)* predation (Toone et al. 2003).

Furthermore, meta-analyses show that wild-to-wild translocations and releasing large numbers of individuals (generally n > 100) leads to higher reintroduction success across species (Griffith et al. 1989, Fischer & Lindenmayer 2000). However, the use of wild individuals is not always possible even for non-endangered species (Jule et al. 2008, Rummel et al. 2016), especially because many wild populations are declining and in some cases no genetically similar population is available for release (Wilson & Price 1994). Also, even with a low number of released individuals, a reintroduction can be successful when there is low mortality and low dispersal, because surviving individuals have greater chances of success when remaining within the release site (Taylor et al. 2005). In 2004, 29 collared peccaries were released in Texas, and although several individuals left the release site right after the reintroduction, 27% of them survived and remained within the study site (Porter 2006). Even though no studies addressed the population density of the reintroduced individuals, several reproduction events and newly formed groups were evident after seven years of release potentially allowing the establishment of a wild population (Richter 2012).

Moreover, non-natural sources of mortality, such as hunting, can have large effects on population dynamics of large herbivores (Peres 2000, Jerozolimski & Peres 2013). When there is no hunting pressure, adult survival rates remain constant over time and species population dynamic is mainly dependent on juvenile survival rates (Gaillard et al. 1998). In the wild, adult collared peccaries show annual survival rates of about 90% in areas devoid of hunting activities **(**Hellgren et al. 1995), but monthly survival rates can naturally vary from 63% to 82% (Taber et al. 1994).

Post-release movements as a measure of initial reintroduction success

In this study, most peccaries settled near the release site, and established a home range, however, those that dispersed appeared to suffer higher mortality rates (individuals from the second group). Almost all the surviving individuals remained within 2 km of their release site and gradually showed site fidelity by establishing a home range, usually within three months after release (i.e. male 1). Of the eight released peccaries in the first group, only one dispersed away from the release site, sub adult male (male1), and survived until the end of the study. This is consistent with the male-biased dispersal characteristic for the species (Cooper et al. 2010), where related males may leave the group to join a neighboring herd (Byers & Bekoff 1981, Cooper et al. 2011). However, in the second group all three released females dispersed and died within the first month, while the only male of the group stayed in near de release site.

The relatively low dispersal rates in first group may be attributed to several factors: suitable habitat at the release site, sufficient acclimation in the pre-release enclosures, social cohesion in the group, and the provision of supplemental food following release and during the winter period of low food abundance in 2016.

Several factors may influence whether animals remain or disperse from a release site and a great deal of research in reintroduction biology is aimed at anchoring animals at the release site due to higher survival rates associated with reduced post-release dispersal (Stamps & Swaisgood 2007, Le Gouar et al. 2012). Several studies suggest that the greater the dispersal distance of reintroduced individuals, the lower their survival probability will be, because they may encounter less desirable habitat, expend more energy, risk aggressive contact with conspecifics, or be more vulnerable to predation (Biggins et al. 1999, Calvete & Estrada 2004, Yott et al. 2011) compared to the remaining individuals at the release site. Thus, the establishment of a home range within the release site improves reintroduction outcomes (Yott et al. 2011, Moehrenschlager & Macdonald 2003, Tweed et al. 2003, Berger-tal & Saltz 2014). Findings from other peccary reintroductions support this generalization. In previous reintroductions in Texas, peccaries' dispersal distance varied from 2 km (Litondo 1993) to 4 km (Porter 2006); however, the dispersing individuals either died or joined another herd.

Poor quality habitat is an important factor that can influence site fidelity (Bender et al. 2007, Linklater et al. 2011, Le Gouar et al. 2012) and reintroduction outcomes (Griffith et al., 1989). Because wild peccaries went extinct over 50 years ago in the region, it is difficult to determine what represents the best habitat for the species. In this study, the release site was chosen because of the greater hygrophilous forest area, and minimal human presence (e.g. only tourists and no hunting). However, if the animals had dispersed more than 20 km they likely would have experienced low survival because the habitat is comprised primarily of livestock pasture.

Soft-release techniques such as acclimating animals to the release area in release enclosures and supplemental feeding are two important tools that practitioners use to dampen dispersal from the release site (Stamps & Swaisgood 2007, LeGouar et al. 2012; McIntosh et al. 2014, Berger-Tal & Saltz 2014). Supplemental food has been shown to reduce dispersal of individuals and increase reintroduction success (Rickett et al. 2013, Sweikert & Phillips 2015). In addition, the release of individuals familiar with one another and social cohesion in general are important determinants of post-release movements (Armstrong 1995, Shier & Owings 2006, Shier & Swaisgood 2012, Porter 2006). Previous studies in gregarious herbivores showed that the animals that aggregate into cohesive family groups after release show shorter dispersal distances as compared to animals that disperse by themselves (Fryxell et al. 2008, Haydon et al. 2008, Yott et al. 2011). Thus, it appears that social conflict might be an important impetus of long-distance post-release dispersal and efforts to reduce social conflict may also decrease dispersal and its associated problems.

Home range

Several definitions of home range exist. Burt (1943), without considering exploratory movements, defines the home range as the area one individual uses for foraging, mating, and caring for its offspring. Van Winkle (1975) defines it as a frequency distribution of the locations of the individual or group over a defined period of time; he adds this time variable not specified by Burt. Powell and Mitchell (2012) defined home range as the cognitive map of an animal’s environment which the animal remembers and updates constantly. Thus, home ranges are dynamic and can change depending on new or seasonal environmental factors and should be directly linked to important resources (Powell & Mitchell 2012).

The home range of collared peccaries has been studied in different environments such as tropical forests (McCoy-Colton 1990, Fragoso 1994, Judas & Henry 1999, Keuroghlian et al. 2004), arid or semiarid regions (Ellisor & Harwell 1969, Taber et al. 1994, Ilse & Hellgren 1995a, Richter 2012). Other studies focused seasonality with varying food resources (Judas & Henry 1999, Reyna-Hurtado et al. 2009) and interspecific interactions with feral pigs or white-lipped peccary (Taber et al. 1994, Ilse & Hellgren 1995a). Most of these studies used one or two methods to measure home range size, usually MCP or KDE (Table 3). However, in the latest publications the trend is to use more informative statistical approaches such as KDE with either reference or lscv as a smoothing parameter, and use MCP just for comparison with older publications (Keuroghlian et al. 2004, Reyna-Hurtado et al. 2009).

When comparing MCP estimates with previous studies, the home range areas of collared peccaries in this study were similar to studies in Texas (Ellisor & Harwell 1969) and Costa Rica (McCoy et al. 1990) but smaller than those found in other tropical and reintroduced populations in Texas (Table 7). This finding may be attributed to high site fidelity due to supplemental food given during the first year of release.

There is a wide variation in the home range size of collared peccaries across the species distribution (Taber et al. 2011). In a study in Texas the home range of peccaries soft-released was 2.5 km^2 after six months compared to a hard released group which had a home range of 8.3 km^2 (Porter 2006). Porter (2006) attributed this difference in ranges to release type. Richter (2012) studied this population after seven years of reintroduction and found home range sizes from 2 to 12 km^2 (Table 7). However, the mean home range size right after reintroduction, was 4.5 km^2 (Porter 2006). The larger home ranges in Texas were from the herds that had more radio-tagged individuals which may suggest that there was an effect of sampling and marked individuals (Richter 2012). Furthermore, in other regions of Texas, the home range of peccaries was between 0.8 to 2 km^2 (Ellisor & Harwell 1969, Ilse & Hellgren1995). Habitat quality and population size may be the reason for home range changes through time (Ellisor & Harwell 1969, Day 1985, Judas & Henry 1999).

Seasonality may also play a role in peccaries’ home range (Reyna-Hurtado et al. 2009); in this study site, peccaries used three forest patches probably because of the exotic fruit availability during winter time in one of the patches, while the increase in native fruit availability during the same time in the other two (Fig. 14C). When looking at monthly home ranges, most methods show an increment in the home range during the second month. This increment could also be attributed to exploratory behaviors when fruits are scarce (Bigler 1974, Altrichter et al. 2002, Carrillo et al. 2002, Keuroghlian & Eaton 2008).

In this study MCP and KDE (href) helped to identify how peccaries changed their use of space through time, by providing a numerical representation of how the released individuals were settling in a fixed area, probably establishing a home range. Furthermore, AKDE did provide more information on the future use of space of the reintroduced peccaries and will help in selecting the areas and number of herds that can be released within INR.

Behavior as a measure of initial reintroduction success

When animals are released into a new environment they need to explore and gain knowledge that will allow them to assess their new site, settle in, and survive; this increased knowledge will yield behavioral changes (Berger-Tal and Saltz 2014). By identifying this behavioral changes and comparing them to wild populations we can determine reintroduction progress. In Ibera, the first released group spend more time traveling and lower foraging activity compared to the following year (2016), when the opposite trend was observed. This may indicate accumulated knowledge of location of resources.

The observed longer time invested in travel during the first months after release is probably because individuals had to explore the new area, and thus reduced resting time and time for social interactions (Rubenstein 1991, Dunbar 1992). Altrichter et al. (2002) used activity budget as a measure of nutritional stress in a population of white-lipped peccaries in Costa Rica, where individuals increased time traveling and decreased time engaging in social interactions during times of low food availability, indicating high levels of stress. In this study, the first group (familiar individuals) spent more time on social interactions (although not significant, $p = 0.7$) during the first three months after the release compared to one year later during the same months. This difference is consistent with greater traveling behavior right after release, peccaries traveling or moving show more social interactions than when foraging or resting (Byers & Bekoff 1981).

Activity budget assessments for native populations of collared peccaries are scarce in the literature, most studies focused their research on group size, social or foraging behavior (Bigler 1974, Byers & Bekoff 1981, Kiltie & Terborgh 1983, Robinson & Eisenberg 1985, Lochmiller et al. 1986). Most peccaries spent less time foraging during the first three months post-release than that observed for the species in Arizona (Bigler 1974), but one year later their behavior was more similar to that of established wild peccaries. However, one individual, male 1, showed similar foraging proportions to peccaries in Arizona and the first groups after one year of release. This more natural behavior exhibited by male one could be a reflection of sampling, because we were not able to find him for about 20 days after release, travel behavior was not recorded in the most critical phase. Therefore, more foraging activity is consistent with this animal establishing in its new area.

Finally, these behavioral data suggests that exploratory behavior of the first group was not greatly affected by the release. Even though the animals were exploring during the first three months in their new area, there is not a great difference in the activity budget between years. Furthermore, they potentially had enough food through the year since they showed no nutritional or social stress (no changes in foraging behavior through time) similar to white-lipped peccaries in Costa Rica (Altrichter et al. 2002).

Activity patterns

The activity patterns of the reintroduced individuals were consistent with studies from Mexico (Briceño-Mendez et al. 2016), Peru (Tobler et al. 2009), Bolivia (Gomez et al. 2005), and Ecuador (Blake et al. 2012, Espinosa & Salvador 2017) where they are mostly diurnal individuals. However, some studies have also shown activity at night (Taber et al. 1994, Weckel et al. 2006), especially in areas with marked seasonality where peccaries were more crepuscular during summer months (Bigler 1974, Taber et al. 1994).

Their activity peak in Mexico was during the middle of the day (Briceño-Mendez et al. 2016). In the Paraguayan Chaco they have one peak early in the day and one early at night (Taber et al. 1994). In the Ecuadorian rainforest their peak of activity is mostly in the early morning (Blake et al. 2012). Similar to the populations in the Paraguayan Chaco, in the INR the peccaries' activity peaks were at the early morning and late afternoon during 2016.

Because peccaries avoid extreme weather conditions, this difference in activity peaks may be the effect of temperature (Bigler 1974, Bissonette 1978, Taber et al. 1994, Hofmann et al. 2016) or water availability (Hofmann et al. 2016). Hunting pressure also affects daily activities of peccaries. Peccaries can have a more nocturnal activity when hunting is during the day (Espinosa & Salvador 2017) or a greater activity peak during the middle of the day when hunting occurs at night (Briceño-Mendez et al. 2016). Since there is no hunting in the INR, the activity patterns of the release peccaries should be more affected by temperatures. Similar to the activity pattern of peccaries' in the Paraguayan Chaco, the individuals in the INR would probably become more nocturnal during the summer months to avoid extreme temperatures (Taber et al. 1994), however, during this study they were primarily diurnal.

Interactions between collared peccaries and feral pig can potentially alter peccaries' abundance (Ilse & Hellgren1995a) and their activity; however, I found no evidence that the presence of feral pigs affected the activity pattern of the reintroduced peccaries. Feral pigs were cathemeral (active day and night) in the study area, and they were active at night in a greater proportion than peccaries (Fig. 13). Also, both species showed similar activity peaks during the day (Fig. 12). Galetti et al. (2015) found a similar situation, in the Atlantic forest, where collared peccaries and feral pigs coincided in their foraging activity, probably because there was not a great diet overlap (Desbiez et al. 2009). However, when feral pigs were not present collared peccaries had a greater activity during late afternoon (Galetti et al. 2015).

Collared peccary ecology at the Ibera Natural Reserve

Diet composition

Overall, peccaries increased their dietary composition and learned to feed from native fruits once the supplemental food was not available as often (Table 2). During the first three months after release the individuals received supplemental food and they were able to leave the pen to feed from native plants. Thus, right after release they fed primarily on supplemental food, roots, and leaves, even though palm fruits were available year round (Table 1). A year later, the peccaries consumed more fruits, flowers, and scavenged on dead feral pig, which is consistent with their diet composition in other tropical forests (Beck 2006). Surprisingly, the first group had a great diet shift from consuming primarily roots right after release compared to a greater proportion of leaves during 2016. This shift may be explained by the different resource availability in the areas occupied by the peccaries during this time, in the 2016 survey the peccaries were constantly in the open savanna feeding on grass and close to the edge of the wetland feeding on leaves, too.

During acclimation peccaries were also given several native species (Table 2) including palm species available during winter (Beck 2006). However, during acclimation and right after release peccaries would chew the fruit and expectorate the seed in almost all events. This behavior usually occurs when the endocarp is too hard

(Beck 2005, 2006, Motta et al. 2008). During winter of 2016, peccaries were observed eating the entire fruit and only sometimes spit out the seeds. Male 1, right after release, based his diet on roots and leaves while male 2 immediately started consuming palm and *Citrus* fruits (Table 2). Male 2 was probably used to palm fruits since he came from a montane forest in Salta, however, his specific origin is unknown.

Collared peccaries are omnivores (Kiltie 1981, Bodmer 1990, Barreto et al. 1997, Taber et al. 2011), although many authors consider them mainly frugivores (Olmos 1993, Salazar 2007, Keuroghlian & Eaton 2008). In Ibera the first group presented an omnivorous diet at least for most of the winter time, while male 2 was mostly frugivorous in the winter right after release (Table 1). In total the released individuals consumed different parts of 31 species of plants; including fruits from 14 species. These observations were collected across the year with more emphasis during winter time. During winter months, peccaries mostly consumed exotic fruits such as *Citrus sinensis*, *C. reticulata*, *C. paradisi*, and palm fruits, *Copernicia alba* and *Syagrus romanzoffiana*.

The palm fruits consumed during 2016 were previously recorded in wild collared peccary diet (Beck 2005). *S. romanzoffiana* was part of the peccaries' diet in Atlantic forest where peccaries act as seed predators and seed dispersers (Keuroghlian & Eaton 2008, Beck 2005, 2006). *Copernicia alba* was also part of the diet of the first group of peccaries. One study in Brazil determined that its seeds were destroyed after ingestion (Beck 2005). However, in Venezuela Robinson and Eisenberg (1985) and Barreto et al. (1997) registered collared peccaries feeding from *C. tectorum*, and their seeds were viable after ingestion suggesting probable dispersal. *C. alba* and *S. romanzoffiana* are the only palm species available in the release site; thus, more information is needed on the role of peccaries as predators or dispersers for these species.

Peccaries also feed from *Opuntia cardiosperma* and intact seeds were found in their feces. *Opuntia sp*. is a predominant part of the peccaries' diet in the Sonoran desert of Arizona and Texas (Everitt et al. 1981, Bissonette 1982, Corn & Warren 1985), probably because in these arid areas *Opuntia* is a valuable source of water (Corn & Warren 1985, Taber et al. 1994). For *Bromelia* spp., previous studies report peccaries eating their fruit, while their seed fate was either unknown or viable (Beck 2005). In INR peccaries consumed the roots of *Bromelia balansae* similar to a study from the Paraguayan Chaco (Taber et al. 1994) and the Pantanal (Desbiez et al. 2009), but fruits were not available during winter. Thus, more information is needed to determine the peccaries' ecological role.

Habitat selection

The habitat selection analysis showed that at larger spatial scales the reintroduced peccaries select the hygrophilous forest. This pattern is consistent with their feeding habits and greater fruit availability in this habitat type (Desbiez et al. 2009, Tressens et al. 2002). Furthermore, by validating the model with the other individuals with similar (male 1) and different (male 2) backgrounds (quarantine and acclimation time, release method) suggests that this model could potentially be used for predicting future release sites. At finer spatial scales (third order of selection), indicated a negative selection of grassland. This avoidance was not previously reported in the Pantanal which has similar habitat composition to Ibera (Desbiez et al. 2009).

Habitat selection studies in the Pantanal suggest that peccaries select forested habitats and are also found near the forest edge (Desbiez et al. 2009). Similarly, in Texas (Ilse & Hellgren 1995b) and the Peruvian Amazon collared peccaries selected areas with dense canopy cover and did not show a preference for terra firme forests or floodplain (Tobler et al. 2009). On a previous reintroduction of peccaries in Texas, where population tripled after about seven years, collared peccaries preferred areas with some forest cover (Richter 2012) as well as urban and suburban areas that provide some vegetation cover (Bellantoni & Krausman 1993); indicating that they can also adapt to small forest patches, a finding consistent with other studies (Canale et al. 2012).

Management Considerations

The successful initial reintroduction of collared peccaries in the Ibera Natural Reserve indicates that a population could be established. Because most mortality events occurred during the first three months of moving the individuals from its original location, more effort should be considered during this period, especially the transport and acclimation phases. Furthermore, efforts to ensure that animals remain within the release site has many benefits. In addition to reducing mortality (more deaths appear to be associated with long-distance dispersal), biologists can recognize and address problems in a timely manner and released animals can be provided with supplemental food or other intervention when needed.

Reintroduction protocol

Future reintroductions should consider large release sites (about 90 km^2), like RSR, with forested areas to ensure good habitat quality and account for the success of possible dispersal individuals, which would still be able to survive, even when dispersing. Furthermore, using a soft release and supplemental food after release is a good method to increase survival and site-fidelity, and help the animals in their transition to the new area. However, special consideration should be taken on the composition of food provided, considering that captive born individuals may not be familiar with native fruits, and provision of native foods may help train animals to recognize and forage on foods at the release site. Moreover, the time spent in the acclimation pen should only last long enough (about 30 days) for the animals to get used to the new native diet and environmental conditions, while this time should be reduced when supplementing with corn or balanced swine food to prevent the animals from depending on these feeding strategy.

The sex ratio in wild peccaries is 3:1 favoring females (Taber et al. 2011), this should be considered in future releases, because more males can increase injuries caused by fights among them. Also, different release pens should be used for different release groups to reduce territorial fights. The distance between pens should at least 8 km to provide sufficient home rage areas. To increase survival after release, only socially cohesive groups should be reintroduced, this will reduce dispersal and injuries from agonistic interactions. When family groups are not available, newly formed groups should be kept and monitored before translocation or in quarantine facility until they become familiar or strong social bonds have been formed. To reduce injuries from fighting, animals could be exposed to each other's' scent signals or allowed protected contact through fencing prior to housing them together, methods that have proven to successfully reduce aggression in zoo animals (Swaisgood & Schulte, 2010).

Technical problems recommendations

Sedation during transport, especially short ones, should be avoided to prevent losses from the anesthesia. Furthermore, the peccaries had problems with too loose fitting of the radio-collars. This problem was also encountered in similar peccary projects (Toone et al. 2003, Juan Campos pers. comm.). Special monitoring prior to release and proper adjustments of the collar are needed to prevent individuals having issues related to the proper fitting of radio collars (i.e. getting their foreleg stuck in it). For carnivores, the collar circumference is generally 2-4 cm larger than the neck circumference, and it should be smaller than the circumference of the head around the ears (B. Jansen, Arizona Game and Fish Department, pers. comm.). A similar method should be tested on peccaries, considering weight change after reintroduction and foreleg width, to avoid subjective or improper fitting of the radio-collars.

Long-term measures of success

The principal objective of a reintroduction is to establish a wild population in the study site, however, in the INR we are also interested in establishing lost ecological functions such as seed predation and dispersal. Thus, for the return of these lost functions and the long-term survival of collared peccaries', Rincon del Socorro ranch must be colonized by various groups of collared peccaries, allowing new individuals to disperse away from the reserve. In the presence of native predators and feral pigs in Texas, peccary population densities were on average 2.8 ind/km^2 thus up to 347 individuals could potentially establish within Ibera Natural Reserve. However, considering the home range of the released animals (up to 9 km^2) about 12 groups could settle in RSR.

Furthermore, to increase the probability of the Ibera peccary population adapting to changing environments, especial considerations should be taken to increase genetic diversity. Because mixed groups of non-related individuals are usually not preferred for reintroduction, I would recommend releasing groups from different Argentinian regions (i.e. Salta, Tucuman, Formosa). Thus, groups from different regions could be spatially closer in the release site to increase genetic variability after dispersal of juvenile males.

Table 1. Diet proportion (observations) of the first group of released collared peccaries in Corrientes, Argentina, from May 2015 - August 2016.

	2015				2016					
	Fruits	Leaves	Roots	Flowers	Fruits	Leaves	Roots	Flowers	Carcass	Vines
Group	0.1	0.25	0.65	-	0.15	0.65	0.05	0.005	0.11	0.03
Male 1	0.06	0.2	0.69	0.05	-	-	-	-	-	
Male 2	-	-	-	-	0.71	0.19	0.1	-	-	-

Table 2. Plant species consumed by reintroduced collared peccaries in Corrientes, Argentina, from May 2015 - August 2016. RT: indicates the time when peccaries where seen eating that species, whether before release (BR) or after release (AR). G: group

Species	**Common name**	**RT**	**G**	**Part eaten**
Pistia stratiotes	Repollito de agua	BR, AR	1	Leave and root
Hydrocotyle bonariensis	Paraguita de agua	AR	1	Leave
Eichhornia azurea	Camalote	BR, AR	1	Leave, stem
Eichhornia crassipes	Aguapé	BR, AR	1	Leaves, stem
Syagrus romanzoffiana	Pindo	BR, AR	1,2	Fruit, seed, leave
Copernicia alba	Caranday	BR	1	Fruit
Bromelia balansae	Caraguatá	AR	1	Root
Opuntia cardiosperma	Tuna	AR	1	Root, fruit, seed and flower
Solanum granuloso-leprosum	Fumo bravo	AR	1	Bark
Cereus argentinensis	Cardon de Montiel	AR	1,2	Stem
Citrus sinensis	Naranja	AR	1,2	Fruit
Citrus reticulata	Mandarina	AR	1,2	Fruit
Citrus paradisi	Pomelo	AR	1,2	Fruit
Scleria sp.	Navajuela	AR	1	Flower and leave
Philodendron tweedianum	Guembé de agua	AR	1	Root
Eugenia uniflora/sp.	Ñangapiri	BR, AR	1,2	Fruit
Allophylus edulis	Cocú	AR	1	Fruit
Hexachlamys edulis	Ubajay	AR	1	Fruit
Nectandra angustifolia	Laurel blanco	AR	1	Fruit
Smilax campestris	Zarza negra	BR, AR	1,2	Fruit
Psidium guajava	Guayabo	BR	2	Fruit
Passiflora caerulea	Mburucuyá	BR	2	Fruit
Ficus luschnathiana	Higuerón	BR	2	Fruit
Prosopis affinis	Ñandubay	BR	2	Fruit
Mysine laetevtens	Canelón	BR	2	Leave, stem, fruit
Celtis tala	Tala	BR	2	Fruit, leave

Tipuana tipu	Tipa	AR	1	Flower
Tradescantia sp.	Oreja de gato	AR	1	Leave, stem
Trifolium polymorfum	Trebolillo	AR	1	Leave, stem
Commelina diffusa	Santa Lucia	AR	1	Leave, stem
Commelina erecta	Santa Lucia	AR	1	Leave, stem

Table 3. Home range estimates for released collared peccaries at the Ibera Natural Reserve (INR) Corrientes, Argentina, using Minimum convex polygon (MCP), Kernel density estimator KDE with Least square cross validation bandwidth (hlscv), KDE* with reference bandwidth (href), and Autocorrelated kernel density estimator (AKDE). All areas in km^2. NF: number of fixes, NI: number of individuals per herd, RT: radio tracking time in months.

Reference	**NF**	**NI**	**RT**	**100% MCP**	**95% MCP**	**95% KDE**	**90% KDE**	**50% KDE**	**95% KDE***	**90% KDE***	**50% KDE***	**AKDE**
Group 1st month	71	6	1	1.07	0.35	0.13 (12.9)	0.1	0.01	0.98 (129.8)	0.72	0.15	-
Group 2nd month	30	6	1	1.25	1	0.12 (22)	0.09	0.02	2.68 (219.9)	2.06	0.54	-
Group 3rd month	11	6	1	0.15	0.05	0.14 (82.6)	0.13	0.05	1.83 (224)	1.45	0.42	-
Group 2015	112	6	3	1.90	0.80	0.25 (14.7)	0.2	0.03	1.73 (147.4)	1.29	0.27	1.59 ± 0.39
Group 2016	147	6	3	1.2	0.34	0.12 (10.8)	0.07	0.01	0.85 (107.9)	0.58	0.15	3.54 ± 1.36
Group 2015 & 2016	259	6	6	2.06	1.08	0.38 (14.83)	0.27	0.04	1.94 (148.3)	1.48	0.42	8.9 ± 1.7
male 1 1st month	27	1	1	1.48	1.36	0.61 (72.7)	0.45	0.06	18.2 (720.6)	13.86	3.33	-
male 1 2nd month	30	1	1	0.41	0.28	0.15 (26.5)	0.12	0.03	0.96 (123.24)	0.75	0.18	-

male 1 3rd month	11	1	1	0.07	0.04	0.29 (80.9)	0.22	0.06	0.31 (85.7)	0.24	0.07	-
Male 1	68	1	3	1.76	0.42	0.57 (42.7)	0.4	0.06	6.42 (426.5)	4.5	1.01	2.57 ± 0.85
Male 2	155	1	3	2.1	1.27	0.49 (25.2)	0.31	0.06	3.7 (252.1)	2.89	0.92	1.98 ± 0.5

Table 4. Main habitat types and number of locations of the released collared peccaries in the Ibera Natural Reserve, Corrientes, Argentina.

Habitat type	Area (km^2)	Area (%)	Group/ Individual	Number of locations	Locations (%)
Hygrophilous forest	5.8	4.95	1st Group	86	16.90
			Male 1	10	1.96
			Male 2	88	17.29
Closed savanna	20.2	17.25	1st Group	84	16.50
			Male 1	36	7.07
			Male 2	66	12.97
Open savanna	27.69	23.64	1st Group	83	16.31
			Male 1	20	3.93
			Male 2	18	3.54
Grassland	62.99	53.79	1st Group	2	0.39
			Male 1	1	0.20
			Male 2	7	1.38
Palmar	0.42	0.36	1st Group	5	0.98
			Male 1	2	0.39
			Male 2	1	0.20

Table 5. Second order selection models for the released collared peccaries in Ibera Natural Reserve, Corrientes, Argentina. DFE: Distance to forest edge, DR: Distance to main road, DRP: Distance to release pen. K: number of parameters used in the model, AICc: adjusted Akaike Information Criterion, ΔAIC = difference between the current model and the best model, AIC wt: model probability, cum wt = cumulative model probability.

Model	K	AICc	ΔAIC	AIC wt	Cum. wt
Habitat type + DFE + DR + DRP	8	505.04	0	0.81	0.81
DFE + DR + DRP	4	507.88	2.84	0.19	1
DR + DRP	3	530.89	25.85	0	1
Habitat type + DRP	6	657.36	152.32	0	1
DFE + DRP	3	673.6	168.56	0	1
DRP	2	782.62	277.58	0	1
Habitat type + DFE	6	1544.4	1039.37	0	1
Habitat type + DR	6	1545.51	1040.47	0	1
DFE + DR	3	1547.29	1042.25	0	1
DFE	2	1580.08	1075.04	0	1
Habitat type	5	1615.75	1110.71	0	1
DR	2	2066.76	1561.72	0	1

Table 6. Third order selection models for the first group of released peccaries and male 1 in Ibera Natural Reserve, Corrientes, Argentina. DFE: Distance to forest edge, DR: Distance to main road, DRP: Distance to release pen. K: number of parameters used in the model, AICc: adjusted Akaike Information Criterion, ΔAIC = difference between the current model and the best model, AIC wt: model probability, Cum wt = cumulative model probability.

Model (First group)	K	AICc	ΔAIC	AIC wt	Cum wt	Model (male 1)	K	AICc	ΔAIC	AIC wt	Cum wt
Habitat type + DFE + DR + DRP	8	1430.03	0	1	1	Habitat type + DRP	6	503.7	0	0.82	0.82
DFE + DR + DRP	4	1454.35	24.32	0	1	Habitat type + DFE + DR + DRP	8	506.8	3.14	0.17	1
DR + DRP	3	1486.09	56.06	0	1	Habitat type	5	515.6	11.98	0	1
Habitat type + DR	6	1546.46	116.43	0	1	Habitat type + DR	6	516	12.29	0	1
Habitat type + DRP	6	1551.16	121.13	0	1	Habitat type + DFE	6	517.4	13.78	0	1
Habitat type	5	1560.45	130.41	0	1	DR + DRP	3	540.2	36.54	0	1
Habitat type + DFE	6	1562.29	132.26	0	1	DFE + DR + DRP	4	542	38.38	0	1
DFE + DR	3	1570.45	140.42	0	1	DRP	2	544.8	41.14	0	1
DFE + DRP	3	1572.33	142.29	0	1	DFE + DRP	3	546.8	13.12	0	1
DFE	2	1583.33	153.3	0	1	DFE + DR	3	572.5	68.79	0	1
DR	2	1589.53	159.5	0	1	DR	2	573.1	69.4	0	1
DRP	2	1612.64	182.6	0	1	DRP	2	598.3	94.61	0	1

Table 7. Comparison of home range estimates for collared peccaries from other studies. Minimum convex polygon (MCP), Kernel density estimator KDE. All areas in km^2. NF: number of fixes, NI: number of radio-collar or tagged individuals (number of herds), RT: radio tracking time in months.

Reference	**Locality**	**NF**	**NI**	**RT**	**MCP (100%)**	**MCP (95%)**	**KDE (95%)**	**KDE (50%)**
Ellisor & Harwell 1969	Bee and Jim Wells County, Texas	19 - 69	66 (7)	-	1.25 - 2.21	-	-	-
McCoy et al. 1990	Palo Verde, Costa Rica	3115	6 (3)	10	0.83 - 1.41	0.64 - 1.09	-	-
Fragoso 1994	Maraca Island, Brazil	60-65	7 (2)	6-8	10.1 - 11.7	7.3 - 8.1	-	-
Taber et al. 1994	Boquerón, Paraguay	114	1 (1)	5.5	6.85	-	-	-
Ilse & Hellgren 1995	Welder Wildlife Refuge, Texas	27	5 (5)	3	-	0.81	-	-
Ilse & Hellgren 1995	Welder Wildlife Refuge, Texas	31	6 (6)	3	-	0.59	-	-
Ilse & Hellgren 1995	Welder Wildlife Refuge, Texas	51	6 (6)	3	-	0.98	-	-
Ilse & Hellgren 1995	Welder Wildlife Refuge, Texas	131	6 (6)	12	-	1.76	-	
Judas & Henry 1999	St. Eugene Field Station, French Guiana	112 - 620	5 (3)	12	-	-	1.57 - 2.43	-
Keuroghlian et al. 2004	Sao Paulo, Brazil	17 - 166	3 (1)	3-9	0.4 - 3.4	0.3 - 3.1	0.5 - 3.6	-
Keuroghlian et al. 2004	Sao Paulo, Brazil	139 - 167	2 (1)	9-11	1.25 - 1.44	0.98 - 1.17	1.59 - 1.83	-
Richter 2012	Mason Mountain, Texas	17	1 (1)	4-8	-	1.8	7.86	1.8
Richter 2012	Mason Mountain, Texas	77	11 (1)	4-8	-	9.59	16.2	2.8
Richter 2012	Mason Mountain, Texas	37	3 (1)	4-8	-	2.12	5.33	1.1
Richter 2012	Mason Mountain, Texas	8	3 (1)	4-8	-	2.1	8.2	2.1

Richter 2012	Mason Mountain, Texas	94	8 (1)	4-8	-	6.4	9.9	2.1
Richter 2012	Mason Mountain, Texas	39	7 (1)	4-8	-	5.3	12.8	2.6

Figure 1. Study area, Rincon del Socorro Ranch, Corrientes, Argentina represented by black circle. Blue areas represent the wetland and lakes while dark green areas represent private land owned by the Conservation Land Trust. Map author: Conservation Land Trust, 2011.

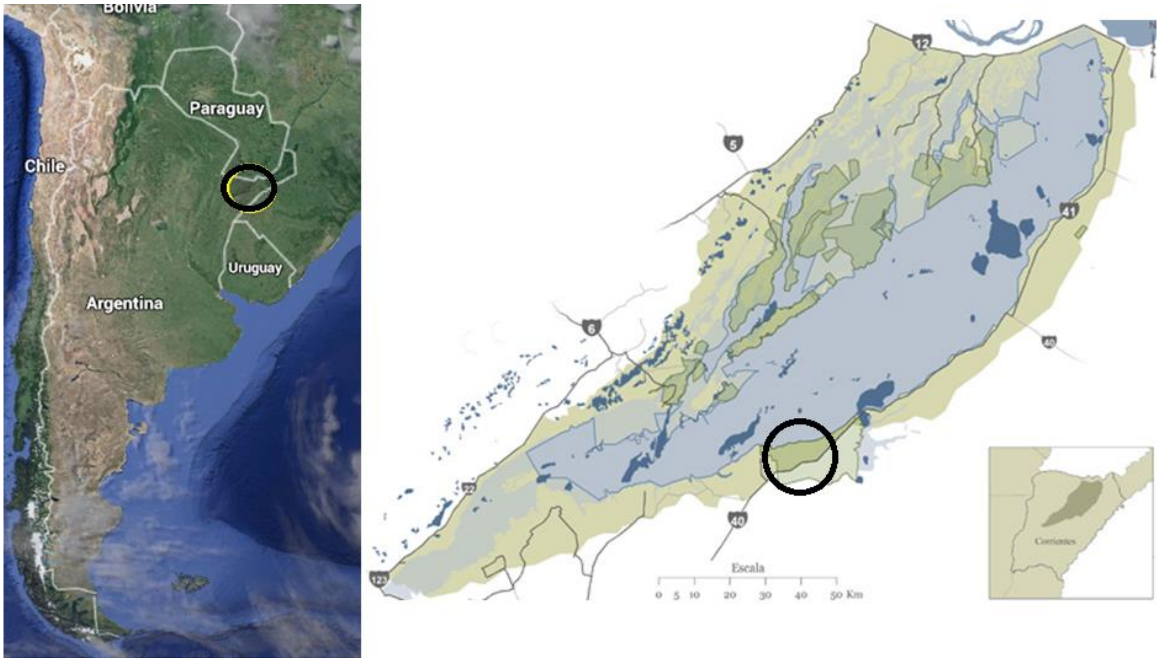


Figure 2. Distribution of the habitat types in El Rincón del Socorro Ranch, Corrientes, Argentina**.** Squares represent areas where pre-release pens were located. First group with a larger acclimation pen (red) and pre-release pen of the second group (yellow).

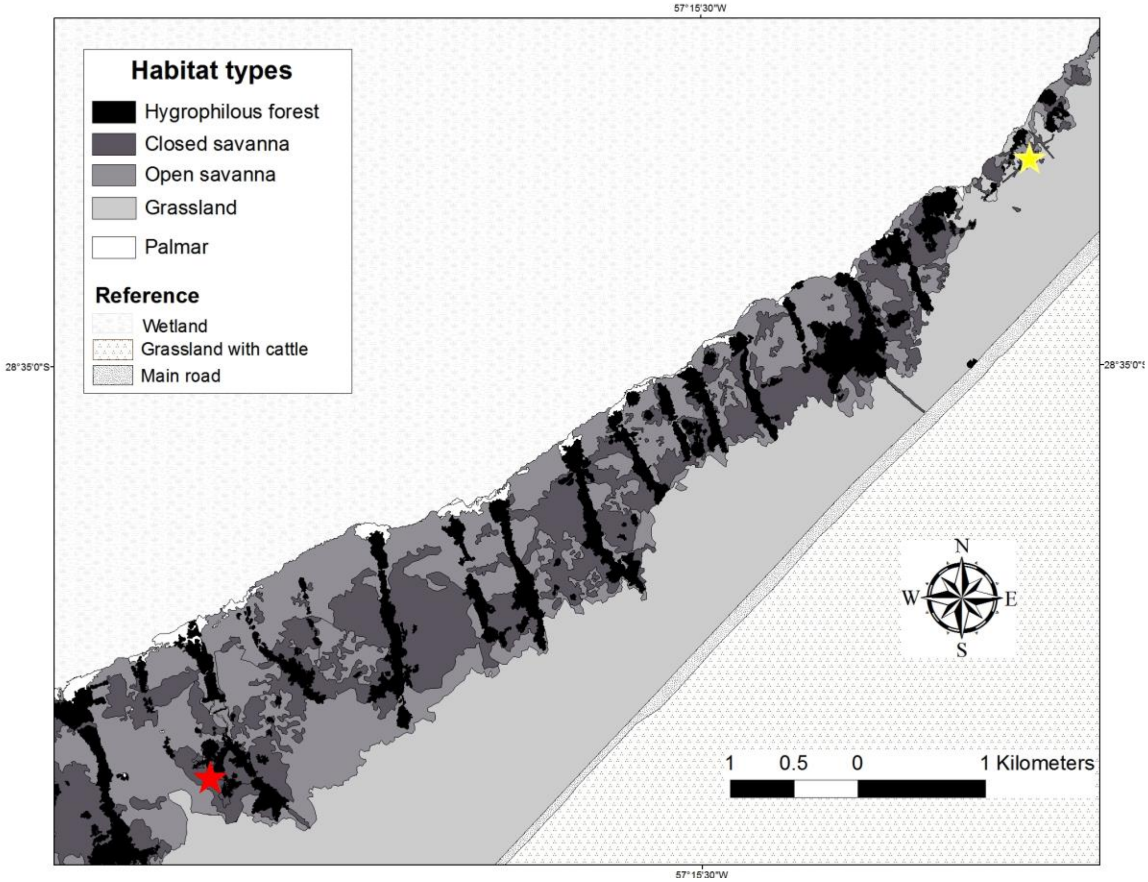


Figure 3. Habitat types within the release site in the Ibera Natural Reserve, Corrientes, Argentina. A) Hygrophilous forest, B) Closed savanna, C) Open savanna, D) Grassland, and E) Palmar.

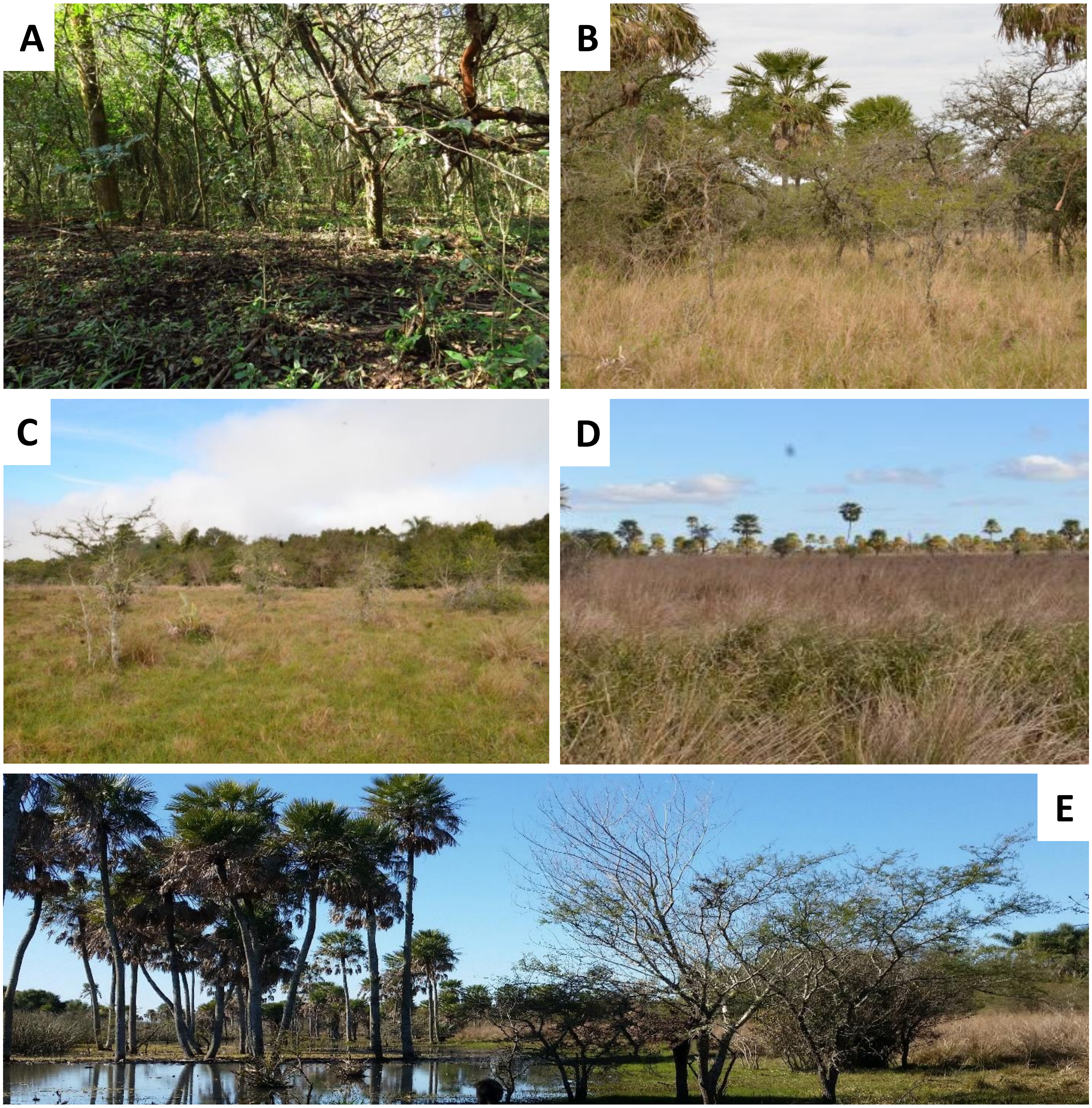


Figure 4. A) Pre-release pen for both released groups of peccaries at the Ibera Natural Reserve, Corrientes, Argentina. B) Circular mobile pen for the soft release of the second group of peccaries.

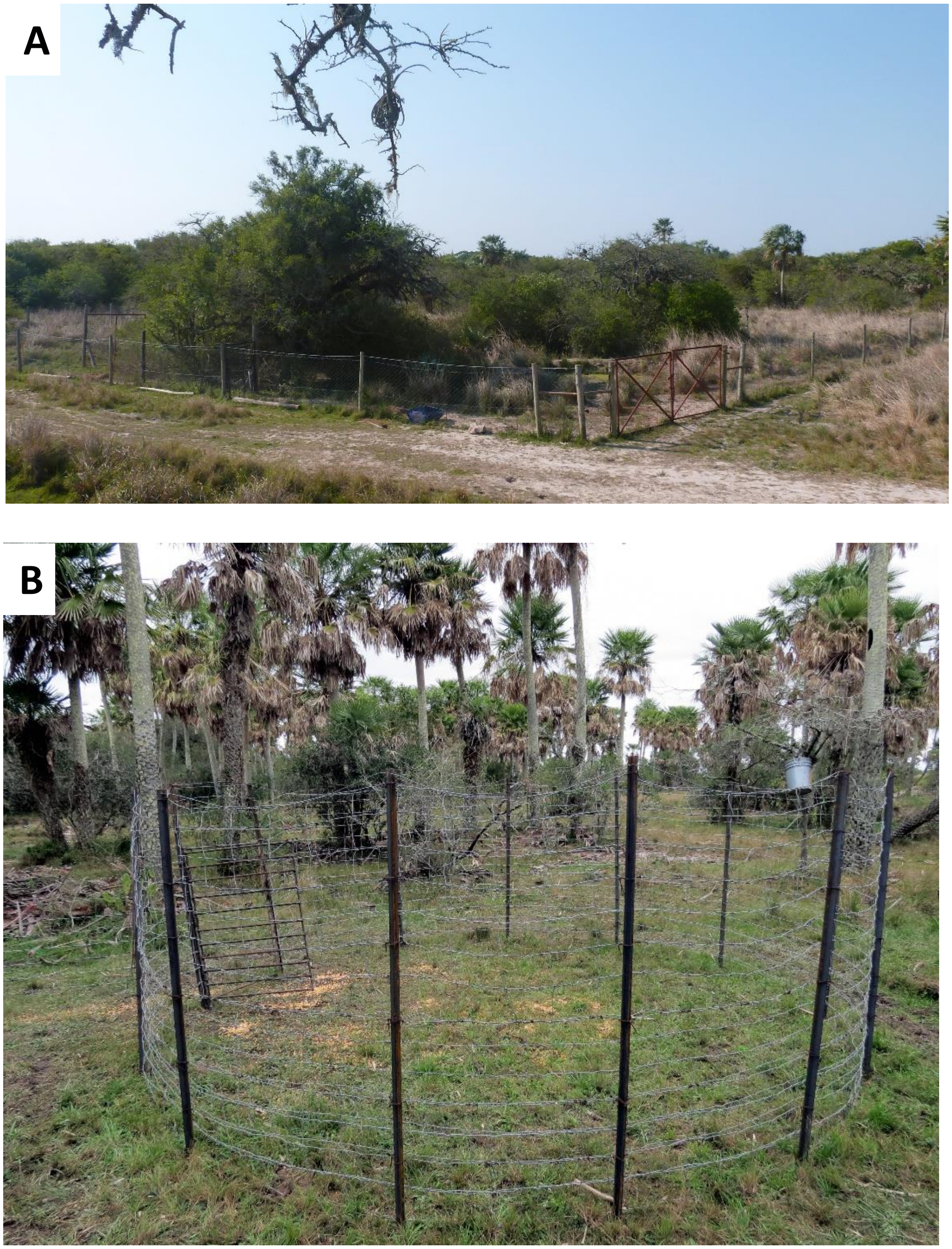


Figure 5. Camera trap locations at the Ibera Natural Reserve, Corrientes, Argentina for the first (A) and second released peccary group (B) and pre-release pen (red star).

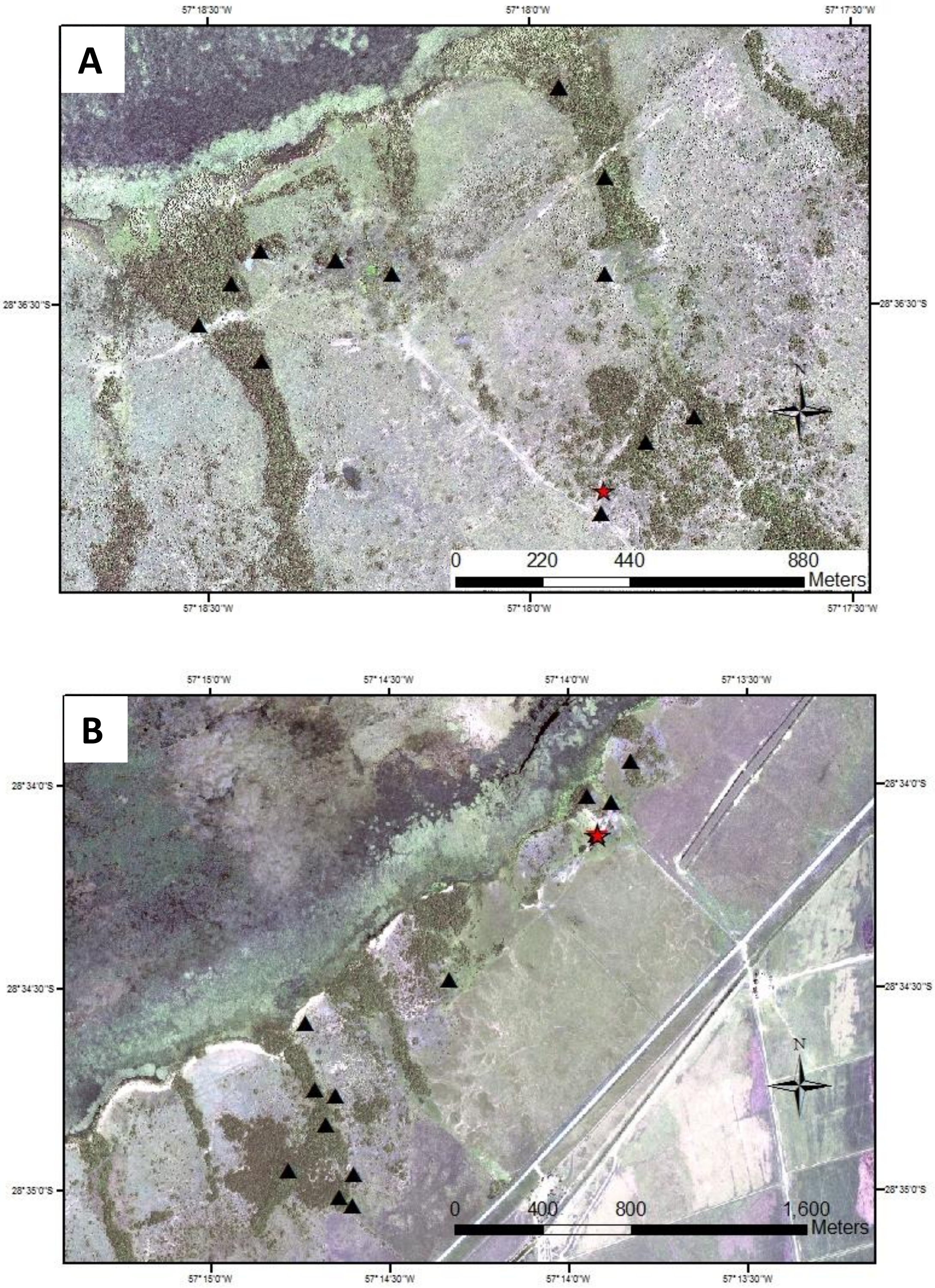


Figure 6. Monthly survival rates right after translocation (A) and post-release (B) for the first (red) and second group (green) of released peccaries in the Ibera Natual Reserve, Corrientes, Argentina. Black lines represent 95% confidence intervals.

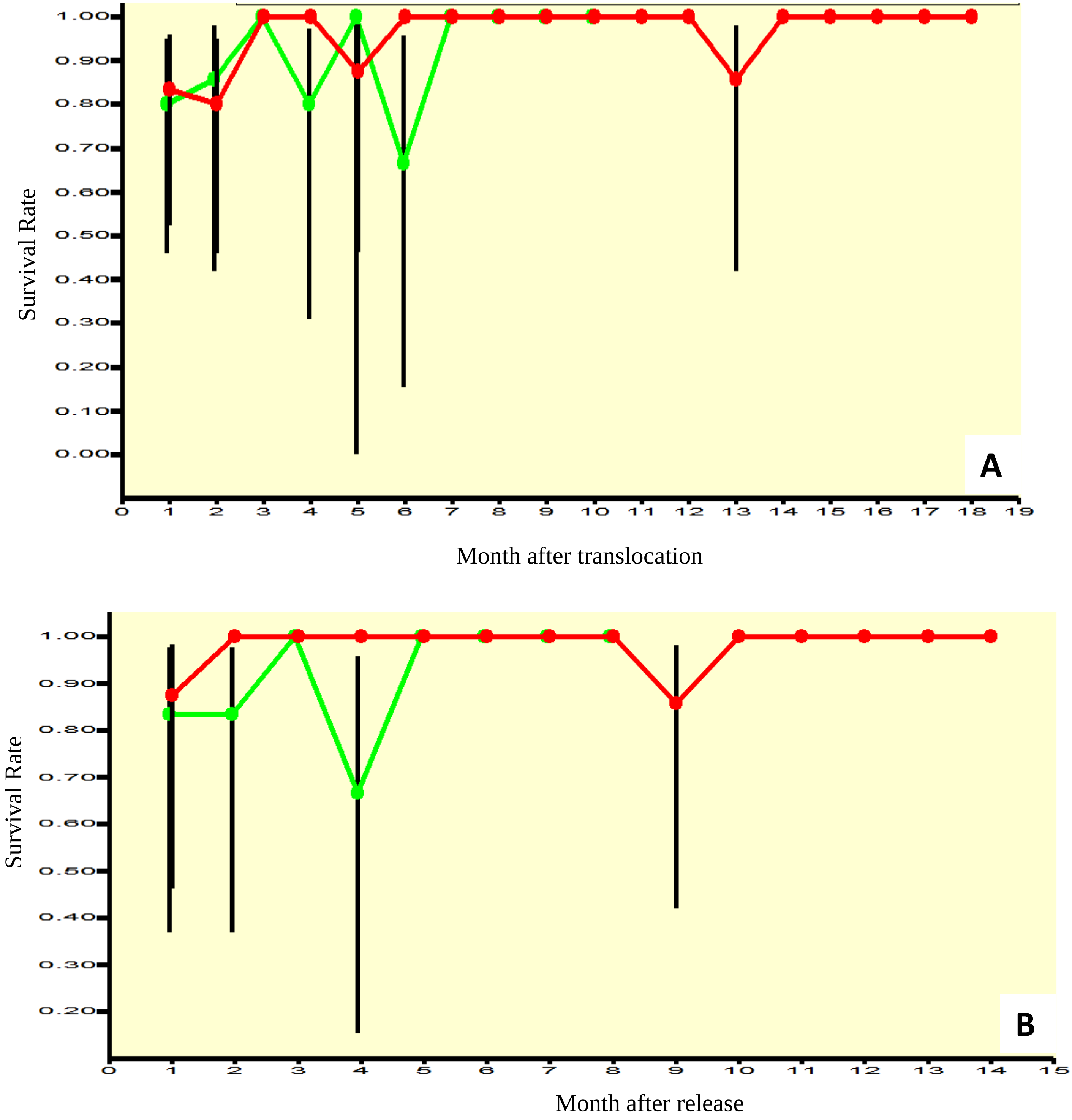


Figure 7. Site fidelity analyses showing mean square distance and linearity index for the released collared peccaries in the Ibera Natural Reserve, Corrientes, Argentina. First group in 2015 (A), 2016 (B), and 2015-2016 (C), and Male 1 (D), and Male 2 (E).

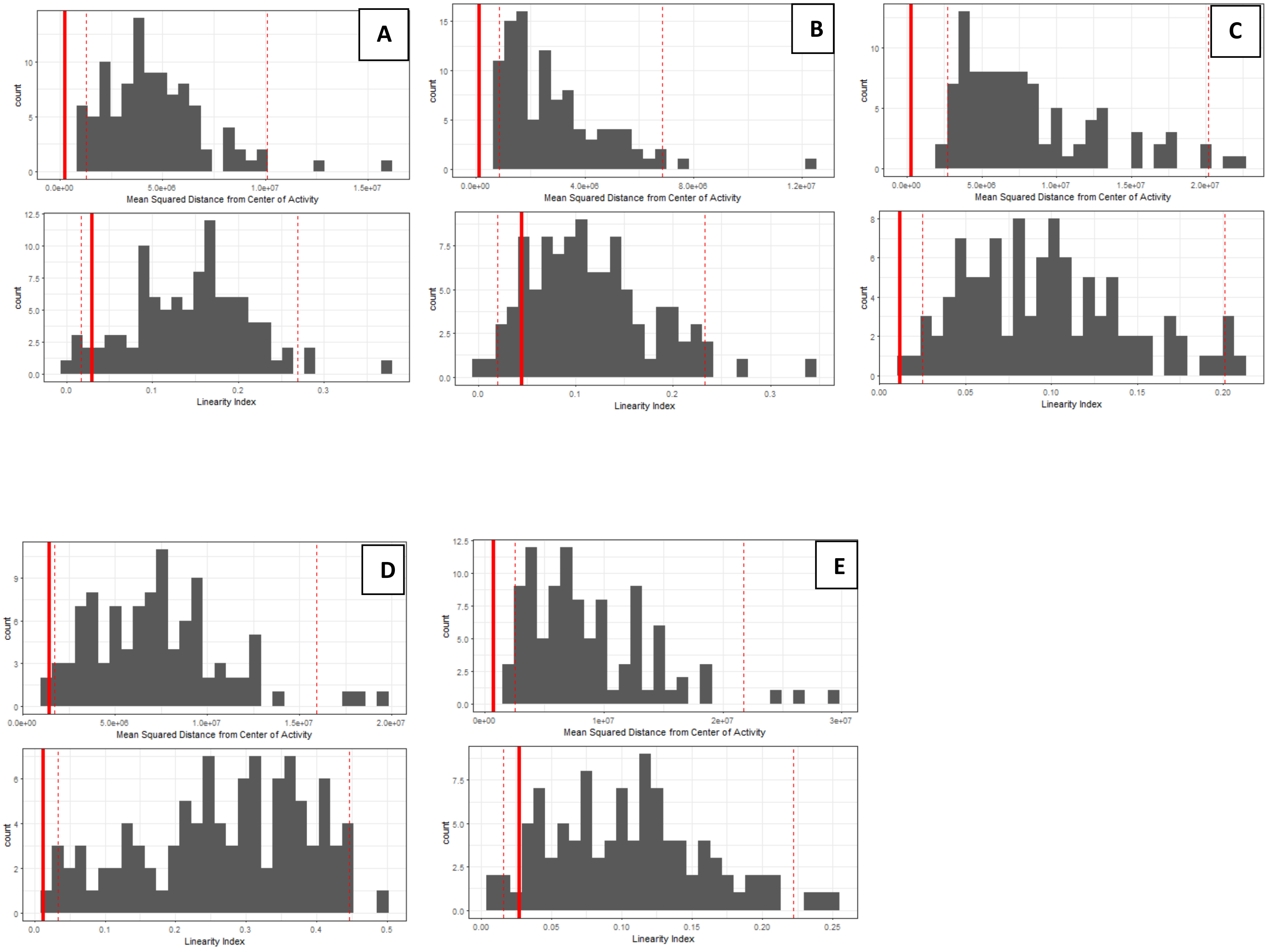


Figure 8. GPS locations from released collared peccaries in the Ibera Natural Reserve, Corrientes, Argentina. A) First group (triangle) and Male 1 (white circle), B) Male 2 (white circle), red star (release pen).

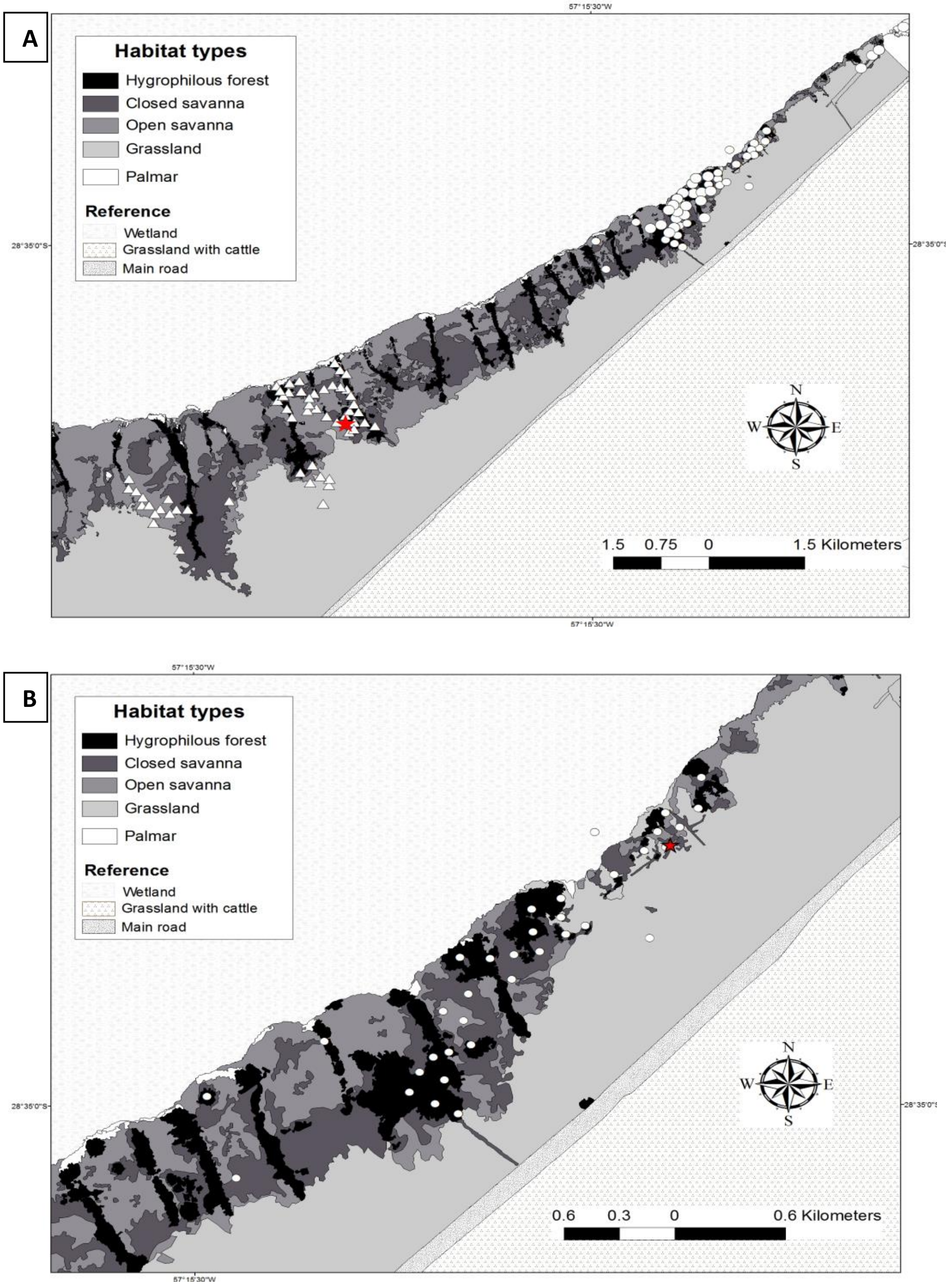


Figure 9. Post release activity budget of collared peccaries in the Ibera Natural Reserve, Corrientes, Argentina.

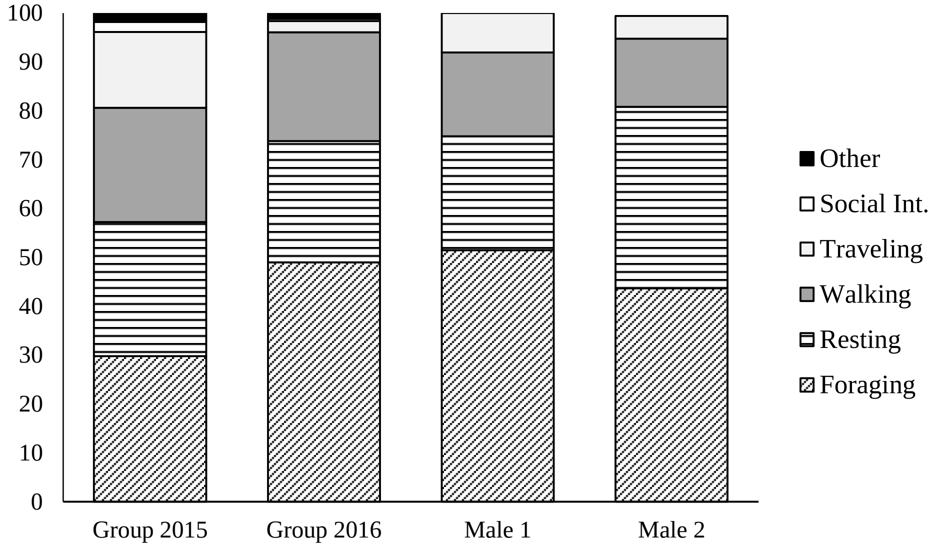


Figure 10. Comparison of collared peccary group activities' right after release. Box and whisker plots show median (horizontal line within box), 25% and 75% percentiles (box) and range (whiskers), circles indicate statistical outliers. p < 0.05 among groups indicated by a *, N = 76.

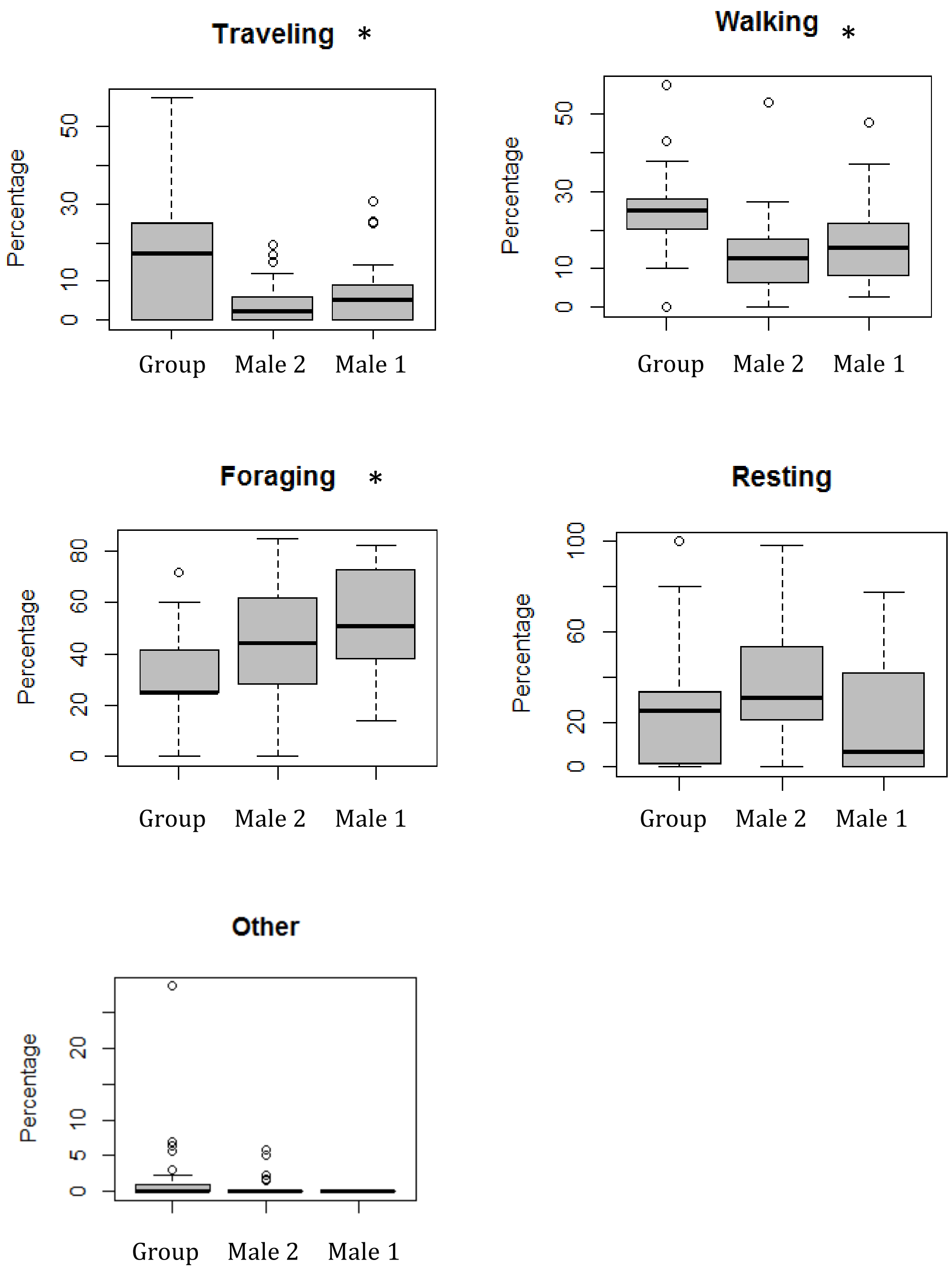


Figure 11. Collared peccaries’ behaviors for the first group in the Ibera Natural Reserve, Corrientes, Argentina, right after reintroduction (2015) and one year later (2016). Box and whisker plots show median (horizontal line within box), 25% and 75% percentiles (box) and range (whiskers), circles indicate statistical outliers, p < 0.01 indicated with * N = 101.

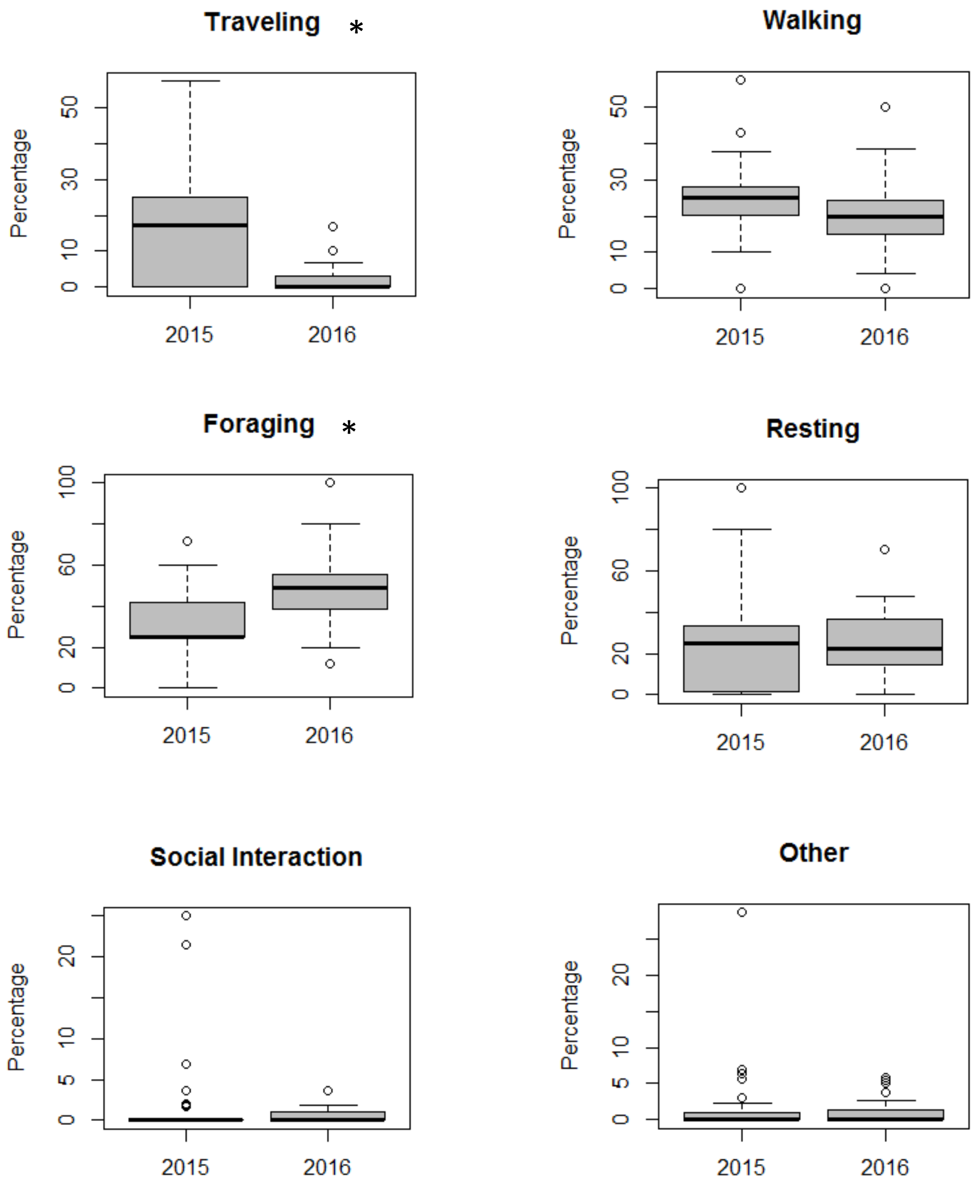


Figure 12. Daily activity pattern of released collared peccaries in the Ibera Natural Reserve, Corrientes, Argentina. 2015 (A) and 2016 (B). Black line shows kernel density estimates, confidence internals showed in dashed lines.

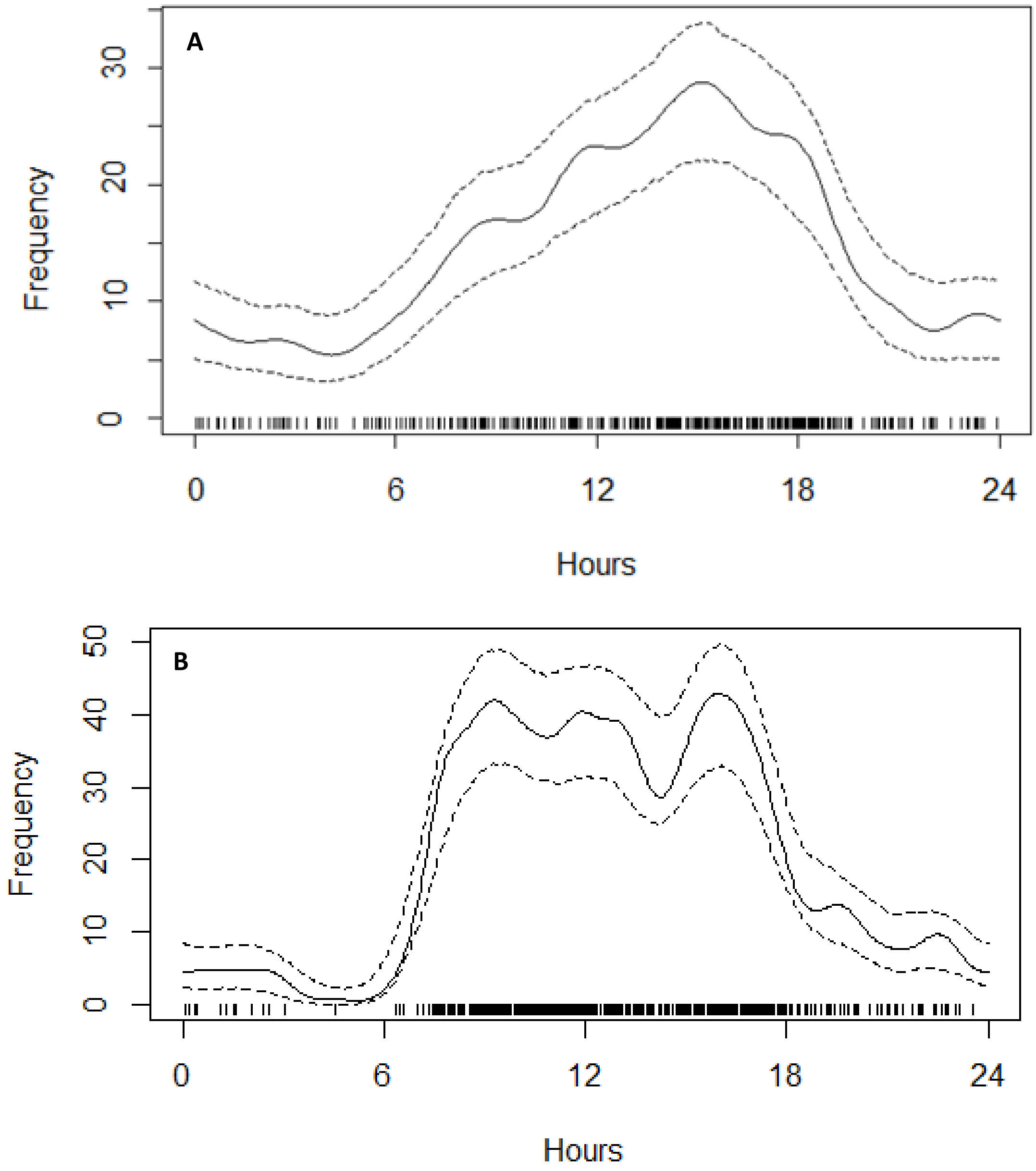


Figure 13. Comparison of daily activity pattern of the released collared peccaries and feral pigs in Ibera Natural Reserve, Corrientes, Argentina. First release group 2015 (A), and 2016 (B), male 2 (C), feral pig in 2015 (D) and 2016 (E).

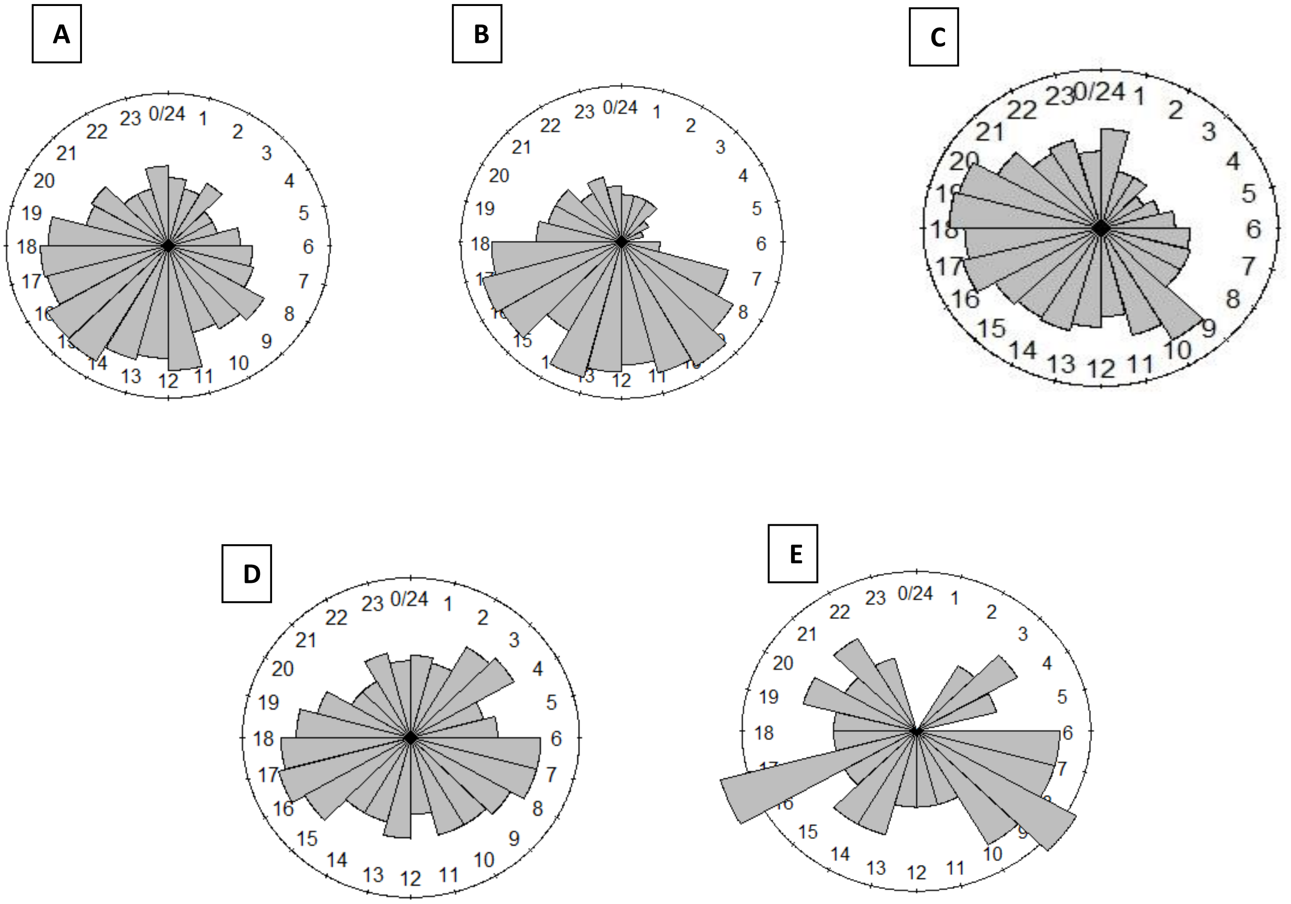
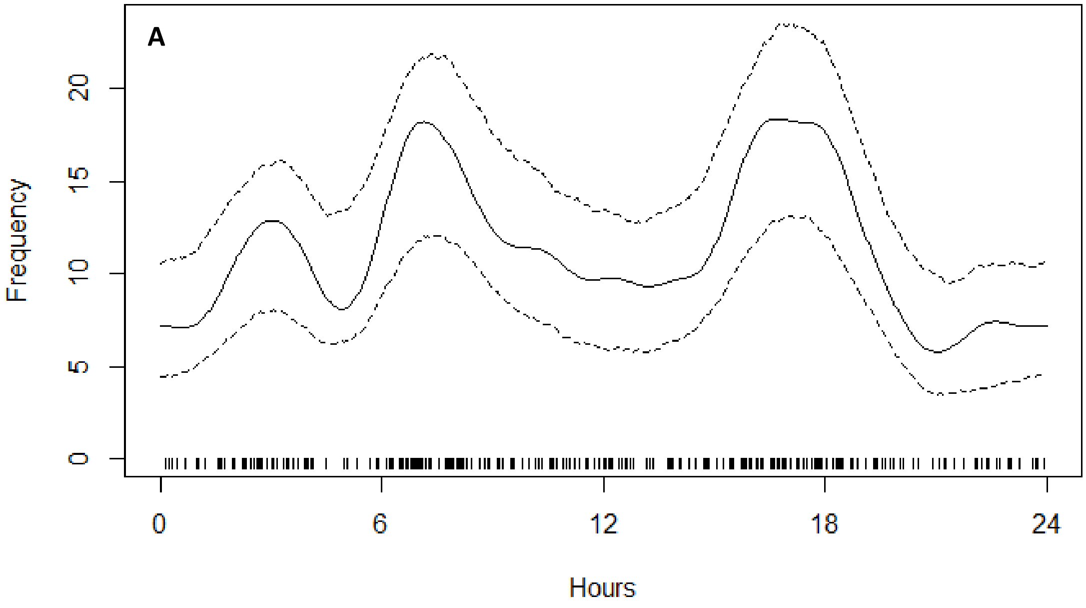


Figure 14. Feral pigs activity pattern in the Ibera Natural Reserve, Corrientes, Argentina during 2015 (A) and 2016 (B). Black line shows kernel density estimates, confidence internals showed in dashed lines.



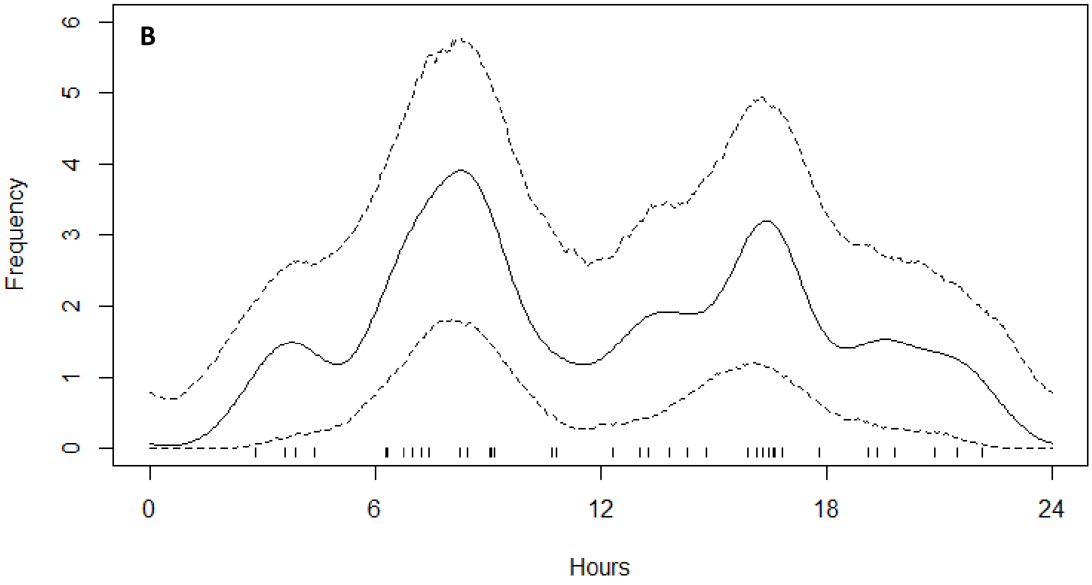


Figure 15. Home range estimation with Autocorrelated Kernel Density Estimator (AKDE) for released collared peccaries in the Ibera Natural Reserve, Corrientes, Argentina during 2015 (A), 2016 (B), and both years combined (C). Black line shows 95% kernel density isopleths, with confidence internals as gray lines. Red dots are the actual tracking locations.

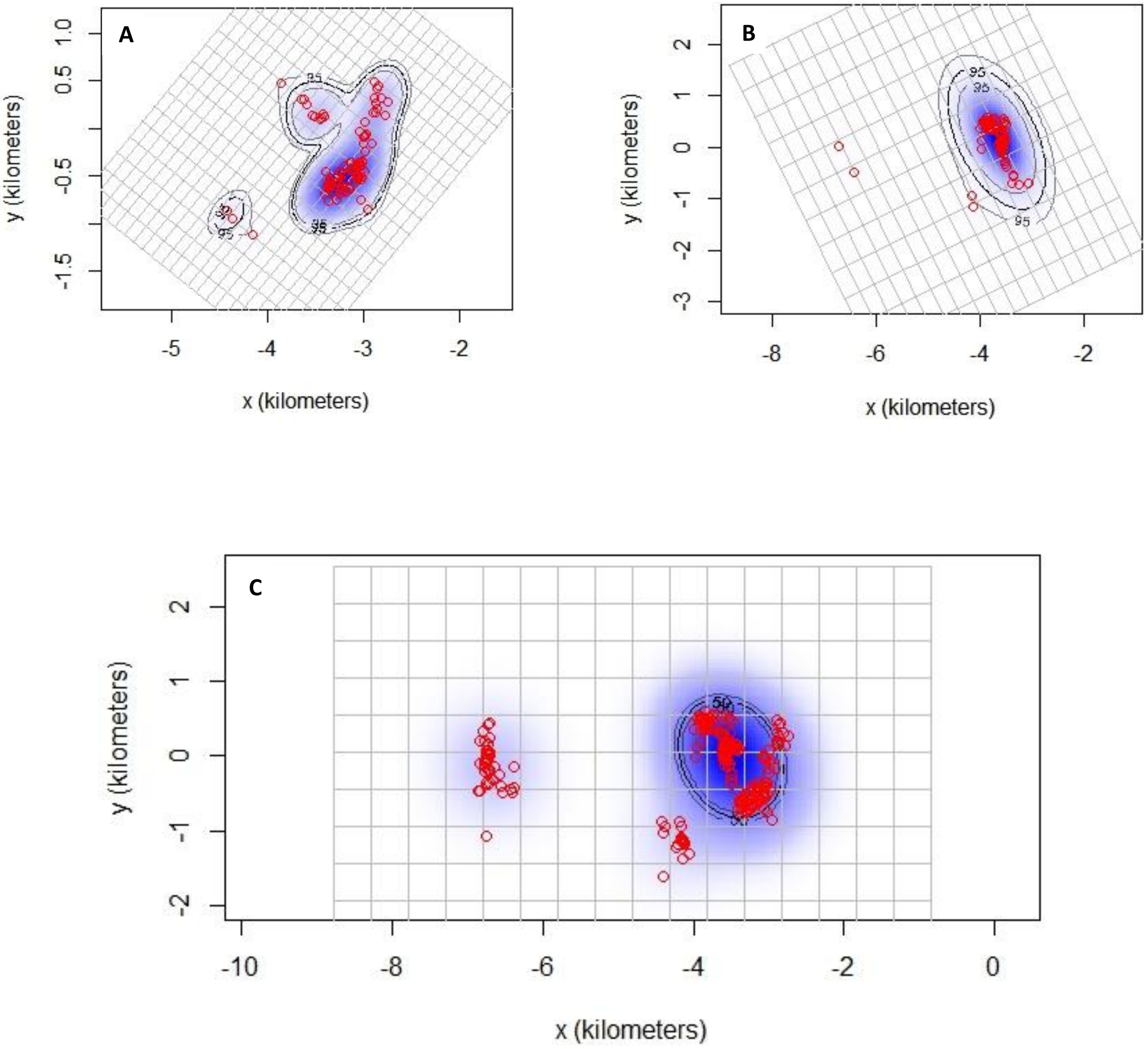
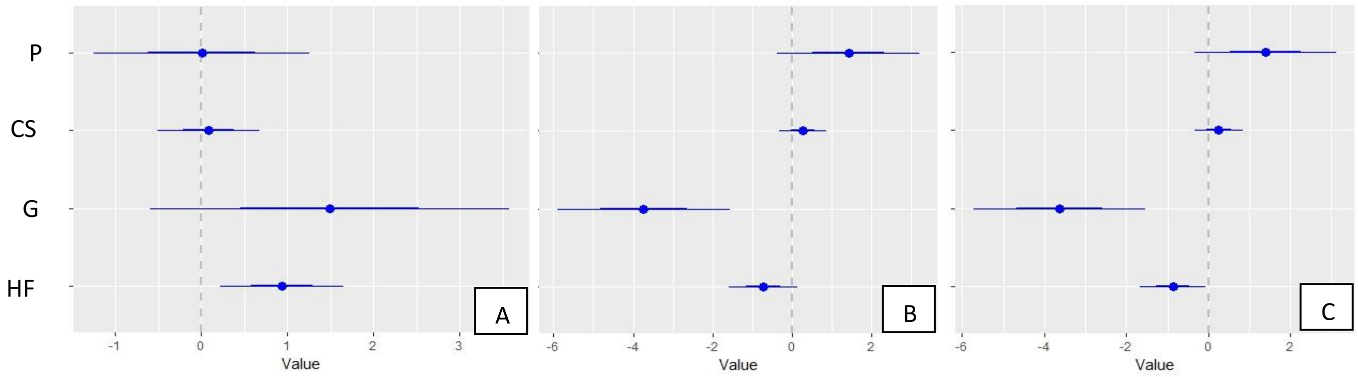


Figure 16. Coefficients ± 90% confidence intervals for different habitat types in Ibera Natural Reserve, Corrientes, Argentina. P: Palmar, CS: Closed Savanna, G: Grassland, HF: Hygrophilous forest. Open savanna was used as intercept for the model. A: Coefficients for 2nd order or selection. B: Coefficients for 3rd order of selection in the first group of peccaries. C: Coefficients for 3rd order of selection for male 1.



Literature Cited

AGOSTINELLI C. AND U. LUND. 2013. R package 'circular': Circular Statistics (version 0.4-7). URL https://r-forge.r-project.org/projects/circular/.

ALTRICHTER, M., AND ALMEIDA, R. 2002. Exploitation of white-lipped peccaries *Tayassu pecari* (Artiodactyla: Tayassuidae) on the Osa Peninsula, Costa Rica. Oryx 36:126.

ALTRICHTER, M., C. DREWS, J. C. SAENZ, AND E. CARRILLO. 2002. Presupuesto de Tiempo Del Chancho Cariblanco (*Tayassu Pecari*) En Un Bosque Humedo de Costa Rica. Biotropica 34:136–43.

ARMSTRONG, D. P., AND P. J. SEDDON. 2008. Directions in reintroduction biology. Trends in ecology & evolution 23:20-25.

BALLOU, J. D. 1993. Assessing the risks of infectious diseases in captive breeding and reintroduction programs. Journal of Zoo and Wildlife Medicine 24:327-335.

BARRIOS-GARCIA M. N. AND S. A. BALLARI. 2012. Impact of wild boar (*Sus scrofa*) in its introduced and native range: a review. Biological Invasions 14:2283-2300.

BARRETO, G. R., O. E. HERNANDEZ, AND J. OJASTI. 1997. Diet of peccaries (*Tayassu tajacu* and *T. pecari*) in a dry forest of Venezuela. Journal of Zoology 241:279-284.

BASTOS DA SILVA S., N. I. DE ALBUQUERQUE, S. P. DA COSTA, M. DA SILVA MACHADO, AND A. S. SARDINHA RIBERIRO. 2014. Reintrodução do macho reprodutor ao grupo de origem em *Pecari tajacu*: consequências da discriminação individual para o manejo de grupos em cativeiro. Suiform Soundings 12:26-28.

BATSON, W., I. GORDON, D. FLETCHER, AND A. MANNING. 2015. Translocation tactics: a framework to support the IUCN guidelines for wildlife translocations and improve the quality of applied methods. Journal of Applied Ecology :1598-1607.

BECCACECI, M. D. 1994. A census of marsh deer in Iberá Natural Reserve, its Argentine stronghold. Oryx 28:131-134.

BECK, H. 2005. Seed predation and dispersal by peccaries throughout the Neotropics and its consequences: a review and synthesis. *In* P. M. Forget; J. E. Lambert; P. E. Hulme and S. B. Vander Wall (eds.). Seed fate: predation, dispersal and seedling establishment, pp. 77-115. CABI Publishing, Wallingford, UK.

BECK, H. 2006. A review of peccary – palm interactions and their ecological ramifications across the Neotropics. Journal of Mammalogy 87:519–30.

BECK, H., J. SNODGRASS, AND P. THEBPANYA. 2013. Long-term exclosure of large terrestrial vertebrates: implications of defaunation for seedling demographics in the Amazon rainforest. Biological Conservation 163:115–121.

BEDIN, E. AND S. OSTROWSKI. 1998. Arabian oryx (*Oryx leucoryx*) reintroductions in Saudi Arabia: Update. Newsletter of the IUCN Re-introduction Specialist Group 16:13-14.

BELLANTONI, E. S. AND P. R. KRAUSMAN. 1993. Habitat use by collared peccaries in an urban environment. The Southwestern Naturalist 38:345-351.

BENDER, L. C., L. A. LOMAS, AND J. B. BROWNING. 2007. Condition, survival and cause-specific mortality of adult female mule deer in north-central New Mexico. Journal of Wildlife Management 71:1118–1124.

BERGER-TAL, O. AND D. SALTZ. 2014. Using the Movement Patterns of Reintroduced Animals to Improve Reintroduction Success. Current Zoology 60:515–26.

BIGGINS, D.E., A. VARGAS, J. L. GODBEY, AND S. H. ANDERSON. 1999. Influence of prerelease experience on reintroduced black-footed ferrets (*Mustela nigripes*). Biological Conservation 89:121-129.

BIGLER, W. J. 1974. Seasonal movements and activity patterns of the collared peccary. Journal of Mammalogy 55:851-855.

BISSONETTE, J. A. 1978. The influence of extremes of temperature on activity patterns of peccaries. The Southwestern Naturalist 1:339-346.

BISSONETTE, J. A. 1982. Ecology and social behavior of the collared peccary in Big Bend National Park, Texas.

BLAKE, J. G., D. MOSQUERA, B. A. LOISELLE, K. SWING, J. GUERRA, AND D. ROMO. 2012. Temporal activity patterns of terrestrial mammals in lowland rainforest of eastern Ecuador. Ecotropica 18:137-146.

BLUMSTEIN, D. T., T. W. WEY, AND K. TANG. 2009. A test of the social cohesion hypothesis: interactive female marmots remain at home. Proceedings of the Royal Society of London: Biological Sciences 276:3007-3012.

BODMER, R. E. 1990. Responses of ungulates to seasonal inundations in the Amazon floodplain. Journal of tropical Ecology 6:191-201.

BORGER, L., N. FRANCONI, G. DE MICHELE, A. GANTZ, F. MESCHI, A MANICA, AND T. M. COULSON. 2006. Effects of sampling regime on the mean and variance of home range size estimates. Journal of Animal Ecology 75:1393-1405.

BORGER L, B. D. DALZIEL, AND J. M. FRYXELL. 2008. Are there general mechanisms of animal home range behavior? A review and prospects for future research. Ecology Letters 11: 637–650.

BOWKETT, A.E., F. ROVERO, A. R. MARSHALL. 2007. The use of camera-trap data to model habitat use by antelope species in the Udzungwa Mountain forests, Tanzania. African Journal of Ecology 46:479-487.

BOYCE, M. S., P. R. VERNIER, S. E. NIELSEN, AND F. K. SCHMIEGELOW. 2002. Evaluating resource selection functions. Ecological modelling 157:281-300.

BRICEÑO-MÉNDEZ, M., E. J. NARANJO, S. MANDUJANO, M. ALTRICHER AND R. REYNA-HURTADO. 2016. Responses of two sympatric species of peccaries (*Tayassu pecari* and *Pecari tajacu*) to hunting in Calakmul, Mexico. Tropical Conservation Science 9:1-11.

BROWN, C., R. MCMORRAN, AND M. F. PRICE. 2011. Rewilding – A New Paradigm for Nature Conservation in Scotland ? Scottish Geographical Journal 127:288–314.

BURNHAM, K.P. AND D. R. ANDERSON. 2002. Model Selection and Multimodel Inference: A Practical Information-Theoretic Approach. Springer, New York, USA.

BURT, W. H. 1943. Territoriality and home range concepts as applied to mammals. Journal of Mammalogy 24:346-352.

BYERS, J. A. AND M. BEKOFF. 1981. Social, spacing, and cooperative behavior of the collared peccary, *Tayassu tajacu*. Journal of Mammalogy 62:767-785.

CALENGE, C., D. MAILLARD, N. INVERNIA, AND J. C. GAUDIN. 2005. Reintroduction of roe deer *Capreolus capreolus* into a Mediterranean habitat: female mortality and dispersion. Wildlife Biology 11:153-161.

CALENGE, C. 2006. The package "adehabitat" for the R software: A tool for the analysis of space and habitat use by animals. Ecological Modelling 197:516-519.

CALVETE, C. AND R. ESTRADA. 2004. Short-term survival and dispersal of translocated European wild rabbits. Improving the release protocol. Biological Conservation 120:507-516.

CANALE, G. R., C. A. PERES, C. E. GUIDORIZZI, C. A. F. GATTO, AND M. C. M. KIERULFF. 2012. Pervasive defaunation of forest remnants in a tropical biodiversity hotspot. PloS one 7:e41671.

CANZIANI, G., R. FERRATI, P. FEDERICO, D. RUIZ MORENO, F. CASTETS, AND A. CANONICA. 2003. Sistemas de información geográfica y sensores remotos. In G. Canziani, C. Rossi, S. Loiselle and R. Ferrati (eds.), Los Esteros de Iberá el manejo sustentable de los recursos de humedales en el Mercosur, pp. 77-82. Fundación Vida Silvestre Argentina, Buenos Aires.

CARRILLO, E., J. C. SAENZ AND T. K. FULLER. 2002. Movements and activities of white-lipped peccaries in Corcovado National Park, Costa Rica. Biological Conservation 108:317-324.

COOPER, J. D., R. VITALIS, P. M. WASER, D. GOPURENKO, E. C. HELLGREN, T. M. GABOR, AND J. A. DEWOODY. 2010. Quantifying male-biased dispersal among social groups in the collared peccary (*Pecari tajacu*) using analyses based on mtDNA variation. Heredity 104:79-87.

COOPER, J.D., P. M. WASER, E. C. HELLGREN, T. M. GABOR, AND J. A. DEWOODY. 2011. Is sexual monomorphism a predictor of polygynandry? Evidence from a social mammal, the collared peccary. Behavioral Ecology and Sociobiology 65:775–785.

CORN, J. L. AND R. J. WARREN. 1985. Seasonal food habits of the collared peccary in south Texas. Journal of Mammalogy 66:155-159.

CORLETT, R. T. 2016. Restoration, reintroduction, and rewilding in a changing world. Trends in ecology & evolution 31:453-462.

CUEVAS, M. F., A. NOVILLO, C. CAMPOS, M. A. DACAR, AND R. A. OJEDA. 2010. Food habits and impact of rooting behaviour of the invasive wild boar, *Sus scrofa*, in a protected area of the Monte Desert, Argentina. Journal of Arid Environments 74:1582-1585.

DANIELSON, B.J. AND R. K. SWIHART. 1987. Home range dynamics and activity patterns of *Microtus ochrogaster* and *Synaptomys cooperi* in syntopy. Journal of Mammalogy 68:160-165.

DAVIS, M. H. 1983. Post-release movements of introduced marten. Journal of Wildlife Management 47:59- 66.

DAY, G. I. 1985. Javelina: research and management in Arizona. Arizona and Game Fish Department, Phoenix, Arizona, USA

DE ALMEIDA JÁCOMO, A. T., M. M. FURTADO, C. K. KASHIVAKURA, J. MARINHO-FILHO, R. SOLLMANN, N. M. TORRES, AND L. SILVEIRA. 2013. White-lipped peccary home-range size in a protected area and farmland in the central Brazilian grasslands. Journal of Mammalogy 94:137-145.

DESBIEZ, A. L. J., S. A. SANTOS, A. KEUROGHLIAN, AND R. E. BODMER. 2009a. Niche partitioning among white-lipped peccaries (*Tayassu pecari*), collared peccaries (*Pecari tajacu*), and feral pigs (*Sus scrofa*). Journal of Mammalogy 90:119-128.

DESBIEZ, A. L. J., R. E. BODMER, AND S. A. SANTOS. 2009b. Wildlife Habitat Selection and Sustainable Resources Management in a Neotropical Wetland. International Journal of Biodiversity and Conservation 1:11–20.

DI BLANCO, Y. E., I. JIMÉNEZ PÉREZ, P. DÍAZ, AND K. SPORRING. 2012. Cinco Años de Radiomarcaje de Osos Hormigueros (*Myrmecophaga tridactyla):* Mejoras implementadas y lecciones aprendidas. Edentata 13:49–55.

DI BLANCO, Y. E., I. J. PÉREZ, AND M. S. DI BITETTI. 2015. Habitat selection in reintroduced giant anteaters: the critical role of conservation areas. Journal of Mammalogy 96:1024-1035.

DICKENS, M. J., D. J. DELEHANTY, AND L. M. ROMERO. 2010. Stress: An inevitable component of animal translocation. Biological Conservation 143:1329-1341.

DODD, C. K., AND R. A. SEIGEL. 1991. Relocation, repatriation, and translocation of amphibians and reptiles: are they conservation strategies that work? Herpetologica 336-350.

DUNBAR, R. I. M. 1992. Time: a hidden constraint on the behavioural ecology of baboons. Behavioral Ecology and Sociobiology 31:35–49.

ELLISOR, J. E. AND W. F. HARWELL. 1969. Mobility and home range of collared peccary in southern Texas. The Journal of Wildlife Management 1:425-427.

ESPINOSA, S. AND J. SALVADOR. 2017. Hunters´ landscape accessibility and daily activity of ungulates in Yasuní Biosphere Reserve, Ecuador. Therya 8:45-52.

EVERITT, J. H., C. L. GONZALEZ, M. A. ALANIZ, AND G. V. LATIGO. 1981. Food habits of the collared peccary on south Texas rangelands. Journal of Range Management 141-144.

FISCHER, J. AND D. B. LINDENMAYER. 2000. An assessment of the published results of animal relocations. Biological conservation 96:1-11.

FLANAGAN, S. E., M. B. BROWN, J. FENNESSY, AND D. T. BOLGER. 2016. Use of home range behaviour to assess establishment in translocated giraffes. African Journal of Ecology 54:365-374.

FRAGOSO, J.M.V. 1994. Large mammals and the community dynamics of an Amazonian rain forest. PhD dissertation, University of Florida, Gainsville.

FLEMING, C. H., W. F. FAGAN, T. MUELLER, K. A. OLSON, P. LEIMGRUBER, AND J. M. CALABRESE. 2015. Rigorous home range estimation with movement data: a new autocorrelated kernel density estimator. Ecology 96:1182-1188.

FRAIR, J. L., E. H. MERRILL, J. R. ALLEN, AND M. S. BOYCE. 2007. Know Thy Enemy: Experience Affects Elk Translocation Success in Risky Landscapes. Journal of Wildlife Management 71:541–54.

FRYXELL, J.M., M. HAZELL, L. BORGER, B. D. DALZIEL, D. T. HAYDON, et al. 2008. Multiple movement modes by large herbivores at multiple spatiotemporal scales. Proceedings of the National Academy of Sciences of the United States of America 105:19114–19119.

GAILLARD, J. M., M. FESTA-BIANCHET, AND N. G. YOCCOZ. 1998. Population dynamics of large herbivores: variable recruitment with constant adult survival. Trends in Ecology & Evolution 13:58-63.

GALETTI, M. AND R. DIRZO. 2013. Ecological and evolutionary consequences of living in a defaunated world. Biological Conservation 163:1–6.

GALETTI, M., H. CAMARGO, T. SIQUEIRA, A. KEUROGHLIAN, C. I. DONATTI, M. L. S. JORGE, F. PEDROSA, C. Z. KANDA, AND M. C. RIBEIRO.2015. Diet overlap and foraging activity between feral pigs and native peccaries in the pantanal. PloS one 10:0141459.

GERMANO, J.M., K. J. FIELD, R. A. GRIFFITHS, S. CLULOW, J. FOSTER, G. HARDING, AND R. R. SWAISGOOD. 2015. Mitigation-driven translocations: are we moving wildlife in the right direction? Frontiers in Ecology and the Environment 13:100-105.

GINE, G.A.F., C. R. CASSANO, S. S. DE ALMEIDA, AND D. FARIA. 2015. Activity budget, pattern and rhythm of maned sloths (*Bradypus torquatus*): Responses to variations in ambient temperature. Mammalian Biology-Zeitschrift für Säugetierkunde 80:459-467.

GIRAUDO, A. R. AND H. PROVEDANO. 2003. Threats of extinction of flagship species in the interior Atlantic Forest. *In*: I.F. Camara and C. Gallindo-Leal (eds), Atlantic Forest of the South America Biodiversity Status, Threats, and Outlook, pp.181-193. Island Press, Washington DC.

GIRAUDO, A. R., A. BORTOLUZZI, AND V. ARZAMENDIA. 2006. Fauna de vertebrados tetrápodos de la reserva y Sitio Ramsar Esteros del Iberá: Análisis de su composición y nuevos registros para especies amenazadas. Natura Neotropicalis 37:1-2.

GONGORA, J., R. REYNA-HURTADO, H. BECK, A. TABER, M. ALTRICHTER, AND A. KEUROGHLIAN. 2011. Pecari tajacu. The IUCN Red List of Threatened Species 2011: e.T41777A10562361. http://dx.doi.org/10.2305/IUCN.UK.2011-2.RLTS.T41777A10562361.en. Downloaded on 13 February 2017.

GOMEZ, H., R. B. WALLACE, G. AYALA, AND R. TEJADA. 2005. Dry season activity periods of some Amazonian mammals. Studies on Neotropical Fauna and Environment 40:91-95.

GUSSET, M., R. SLOTOW, AND M. J. SOMERS. 2006. Divided we fail: the importance of social integration for the re-introduction of endangered African wild dogs (*Lycaon pictus*). Journal of Zoology 270:502-511.

GRIFFITH, B., J. M. SCOTT, J. W. CARPENTER, AND C. REED. 1989. Translocation as a species conservation tool: status and strategy. Science 245:477-480.

GRIFFITHS, C. J., AND S. HARRIS. 2010. Prevention of Secondary Extinctions through Taxon Substitution. Conservation Biology 24:645–46.

GRIFFITHS, C. J., C. G. JONES, D. M. HANSEN, M. PUTTOO, R. V. TATAYAH, C. B.M¨ULLER, AND S. HARRIS. 2010. The use of extant non-indigenous tortoises as a restoration tool to replace extinct ecosystem engineers. Restoration Ecology 18:1-7.

GUIMARAES, P. R., M. GALETTI, AND P. JORDANO. 2008. Seed dispersal anachronisms: rethinking the fruits extinct megafauna ate. PLoS one 3:1-13.

HARDMAN, B., AND D. MORO. 2006. Optimising reintroduction success by delayed dispersal: is the release protocol important for hare-wallabies? Biological Conservation 128:403–411.

HAYDON, D. T., J. M. MORALES, A. YOTT, D. A. JENKINS, R. ROSATTE, AND J. M. FRYXELL. 2008. Socially-informed random walks: incorporating group dynamics into models of population spread and growth. Proceedings of the Royal Society 275:1101–1109.

HELLGREN, E. C., D. R SYNATZSKE, P. W. OLDENBURG, AND F. S GUTHERY. 1995. Demography of a Collared Peccary Population in South Texas. The Journal of Wildlife Management 59:153–63.

HOFMANN, G. S., I. P. COELHO, V. A. G. BASTAZINI, J. L. P. CORDEIRO., AND L. F. B. DE OLIVEIRA. 2016. Implications of climatic seasonality on activity patterns and resource use by sympatric peccaries in northern Pantanal. International journal of biometeorology 60:421-433.

ILSE, M. AND ERIC C. HELLGREN. 1995a. Spatial Use and Group Dynamics of Sympatric Collared Peccaries and Feral Hogs in Southern Texas. Journal of Mammalogy 76:993–1002.

ILSE, L. M. AND E. C. HELLGREN.1995b. Resource partitioning in sympatric populations of collared peccaries and feral hogs in southern Texas. Journal of Mammalogy 76:784-799.

IUCN. 2013. Guidelines for Reintroductions and ther Conservation Translocations. Version 1.0. Gland, Switzerland: IUCN.

JACKSON, C.L., R. SCHUSTER, AND P. ARCESE. 2016. Release date influences first-year site fidelity and survival in captive-bred Vancouver Island marmots. Ecosphere 7:1-16.

JEROZOLIMSKI, A. AND C. A. PERES. 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.

JOHNSON, D. H. 1980. The comparison of usage and availability measurements for evaluating resource preference. Ecology 61:65-71.

JOHNSON, C. J., S. E. NIELSEN, E. H. MERRILL, T. L. MCDONALD, AND M. S. BOYCE. 2006. Resource selection functions based on use–availability data: theoretical motivation and evaluation methods. Journal of wildlife Management 70:347-357.

JORGE, M.S.P., K. FERRAZ, M. C RIBEIRO AND M. GALETTI. 2013. Mammal defaunation as surrogate of trophic cascades in a biodiversity hotspot. Biological Conservation 163:49-57.

JUDAS, J., AND O. HENRY. 1999. Seasonal variation of home range of collared peccary in tropical rain forests of French Guiana. The Journal of wildlife management 1:546-552.

JULE, K. R., L. A. LEAVER, AND E. S. E. G. LEA. 2008. The Effects of Captive Experience on Reintroduction Survival in Carnivores: A Review and Analysis. Biological Conservation 141:355–63.

KERNOHAN, B.J., R. A. GITZEN, AND J. J. MILLSPAUGH. 2001: Analysis of animal space use and movements. - In: Millspaugh, J.J.and Marzluff, J.M. (Eds.);Radio Tracking and Animal Populations. Academic Press, San Diego, USA, pp. 125-166.

KEULING, O., S. NORMAN, AND R. MECHTHILD. 2009. Commuting, shifting or remaining? Different spatial utilization patterns of wild boar *Sus scrofa* in forest and field crops during summer. Mammalian Biology-Zeitschrift für Säugetierkunde 74:145-152.

KEUROGHLIAN A. AND D. P. EATON. 2008. Fruit availability and peccary frugivory in an isolated Atlantic forest fragment: effects on peccary ranging behavior and habitat use. Biotropica 40:62-70.

KEUROGHLIAN, A., D. P. EATON, AND W. S LONGLAND. 2004. Area Use by White-Lipped and Collared Peccaries (*Tayassu Pecari* and *Tayassu Tajacu*) in a Tropical Forest Fragment. Biological Conservation 120:411–25.

KIE, J. G. 2013. A rule-based ad hoc method for selecting a bandwidth in kernel home-range analyses. Animal Biotelemetry 1:1-13.

KILTIE, R. A. 1981. Stomach contents of rain forest peccaries (*Tayassu tajacu* and *T. pecari*). Biotropica 13: 234-236.

KILTIE, R. A. AND J. TERBORGH. 1983. Observations on the Behavior of Rain Forest Peccaries in Perú: Why do White-lipped Peccaries Form Herds? Zeitschrift für Tierpsychologie 62:241-255.

KING, T., C. CHAMBERLAN, AND A. COURAGE. 2012. Assessing initial reintroduction success in long-lived primates by quantifying survival, reproduction, and dispersal parameters: Western Lowland gorillas (*Gorilla gorilla gorilla*) in Congo and Gabon. International Journal of Primatology 33:134-149.

KLEIMAN, D. G. 1989. Reintroduction of captive mammals for conservation. BioScience 39:152–161.

LAPOINT, S. D., J. L. BELANT, AND R. W. KAYS. 2014. Mesopredator Release Facilitates Range Expansion in Fisher. Animal Conservation 18:50–61.

LE GOUAR, P., J. MIHOUB, AND F. SARRAZIN. 2012. Dispersal and habitat selection: Behavioral and spatial constraints for animal translocations. In: Ewen J, Armstrong DP, Parker KA and Seddon PJ (Eds.) Reintroduction Biology: Integrating Science and Management. Oxford: Wiley-Blackwell. pp. 138–164.

LESMEISTER, D. B., M. E. GOMPPER, AND J. J. MILLSPAUGH. 2009. Habitat Selection and Home Range Dynamics of Eastern Spotted Skunks in the Ouachita Mountains, Arkansas, USA. Journal of Wildlife Management 73:18–2518.

LINKLATER, W. L., K. ADCOCK, P. DU PREEZ, R. R. SWAISGOOD, P. R. LAW, M. H. KNIGHT, J.V. GEDIR, AND G. I.H. KERLEY. 2011. Guidelines for Large Herbivore Translocation Simplified: Black Rhinoceros Case Study. Journal of Applied Ecology 48:493–502.

LITONDO, F.R. 1993. Social cohesion as a factor in the successful reintroduction of collared peccaries. Doctoral dissertation, Texas A&M University.

LOCHMILLER, T. R. L., E. C. HELLGREN, AND W. E. GRANT. 1986. Reproductive Responses to Nutritional Stress in Adult Female Collared Peccaries. The Journal of Wildlife Management 50:295–300.

MANLY, N., L. MCDONALD, D. THOMAS, T. MCDONALD, AND E. WALLACE. 2002. Resource selection by animals: statistical design and analysis for field studies. Kluwer Academic Publishers, Dordrecht, Netherlands.

MARTIN, P. AND P. BATESON. 2007. Measuring behavior an introductory guide. Cambridge University Press. 176 Pp.

MARTIN, E. H., V. G. NDIBALEMA, AND F. ROVERO. 2016. Does Variation between Dry and Wet Seasons Affect Tropical Forest Mammals' Occupancy and Detectability by Camera Traps? Case Study from the Udzungwa Mountains, Tanzania. African Journal of Ecology 55:37-46.

MARZLUFF, J.M., J. J. MILLSPAUGH, P. HURVITZ, AND M. S. HANDCOCK. 2004. Relating resources to a probabilistic mea- sure of space use: forest fragments and Steller's jays. Ecology 85:1411–27.

MAZEROLLE, M. J. 2016. AICcmodavg: Model selection and multimodel inference based on (Q)AIC(c). R package version 2.1-0.

MCCOY-COLTON, M. B., C. VAUGHAN-DICKHAUT, M. A. RODRÍGUEZ-SÁENZ, AND D. KITCHEN. 1990. Seasonal movement, home range, activity and diet of collared peccaries (*Tayassu tajacu*) in a Costa Rican dry forest. Movimiento estacional, ámbito de distribución, actividad y dieta de los sainos (*Tayassu tajacu*) en un bosques seco costarricense. Vida Silvestre Neotropical 2:6-20.

MCINTOSH, T. E., R. C. ROSATTE, J. HAMR AND D. L. MURRAY. 2014. Patterns of Mortality and Factors Influencing Survival of a Recently Restored Elk Population in Ontario, Canada. Restoration ecology 22:806-814.

MESOCHINA, P., E. BEDIN, AND S. OSTROWSKI. 2003. Reintroducing antelopes into arid areas: lessons learnt from the oryx in Saudi Arabia. Comptes rendus biologies 326:158-165.

MIHOUB, J. B., K. PRINCE, O. DURIEZ, P. LÉCUYER, B. ELIOTOUT, AND F. SARRAZIN. 2013. Comparing the effects of release methods on survival of the Eurasian black vulture *Aegypius monachus* reintroduced in France. Oryx 48:106-115.

MARZLUFF, J. M., J. J. MILLSPAUGH, P. HURVITZ, AND M. S. HANDCOCK. 2004. Relating resources to a probabilistic measure of space use: forest fragments and Steller's jays. Ecology 85:1411-1427.

MILLSPAUGH, J. J., R. M. NIELSON, L. MCDONALD, J. M. MARZLUFF, R. A. GITZEN, C. D. RITTENHOUSE, M. W. HUBBARD, AND S. L. SHERIFF. 2006. Analysis of resource selection using utilization distributions. Journal Wildlife Management 70: 384–95.

MITCHELL, A. M., T. I. WELLICOME, D. BRODIE, AND K. M. CHENG. 2011. Captive-reared burrowing owls show higher site-affinity, survival, and reproductive performance when reintroduced using a soft-release. Biological Conservation 144:1382–1391.

MOEHRENSCHLAGER, A., AND D. W. MACDONALD. 2003. Movement and survival parameters of translocated and resident swift foxes *Vulpes velox*. Animal Conservation 6:199-206.

MONSARRAT, S., S. BENHAMOU, F. SARRAZIN, C. BESSA-GOMES, W. BOUTEN, AND O. DURIEZ. 2013. How predictability of feeding patches affects home range and foraging habitat selection in avian social scavengers? PloS one 8:e53077.

MOHR, C. O. 1947. Table of equivalent populations of North American mammals, American Midland Naturalist 37:223–249.

MORRONE, J. 2001. BIOGEOGRAFIA DE AMERICA LATINA Y EL CARIBE. M&T Manuales y Tesis SEA Vol 3, Zaragoza.

MOSEBY K. E., J. L. READ, D. C. PATON, P. COPLEY, B. M. HILL AND H. A. CRISP. 2011. Predation determines the outcome of 10 reintroduction attempts in arid South Australia. Biological Conservation 144:2863-2872.

MOTTA, T. C. S., G. GINÉ, S. NOGUEIRA, AND S. L. G. NOGUEIRA FILHO. 2008. Digestive seed dispersion and predation by collared peccaries in the southern Bahian Atlantic forest, Brazil. Suiform Soundings 8:45-52.

MUNGER, J. C. 1984. Home ranges of horned lizards (Phrynosoma): circumscribed or exclusive? Oecologia 62:351–360.

NEIFF, J. AND A. POI DE NEIFF. 2006. Situación ambiental en la ecorregión Iberá. In: A. Brown, U. Martinez Ortiz and A. M. Corcuera (eds.) La Situación Ambiental Argentina, pp. 177-184. Fundación Vida Silvestre Argentina, Buenos Aires.

NICHOLLS, H. 2006. Restoring Nature's Backbone. PLoS Biology 4:893-896.

NORRIS, D., F. MICHALSKI, AND C. A. PERES. 2010. Habitat patch size modulates terrestrial mammal activity patterns in Amazonian forest fragments. Journal of Mammalogy 91:551-560.

OJEDA, R. A., CHILLO V., AND G. DIAZ. 2012. Libro rojo de mamíferos amenazados de la Argentina. SAREM (Sociedad Argentina para el Estudio de los Mamíferos), Mendoza, 257 pp.

OLMOS, F. 1993. Diet of sympatric Brazilian caatinga peccaries (*Tayassu tajacu* and *T. pecari*). Journal of Tropical Ecology 9:255-258.

PERES, C. A. 2000. Effects of subsistence hunting on vertebrate community structure in Amazonian forests. Conservation Biology 14:240-253.

PEREZ, I. J., D. TOMPKINS, AND G. ALDUNCIN. 2013. Giant anteater, a homecoming to Ibera. Conservation Land Trust, Buenos Aires, Argentina, pp 90.

PEREZ CARUSI, L. C., M. S. BEADE, F. MIÑARRO, A. R. VILA, M. GIMÉNEZ-DIXON, AND D. N. BILENCA. 2009. Relaciones espaciales y numéricas entre venados de las pampas (*Ozotoceros bezoarticus celer*) y chanchos cimarrones (*Sus scrofa*) en el Refugio de Vida Silvestre Bahía Samborombón, Argentina. Ecología austral 19:63-71.

PEREZ-CORTEZ, S. AND R. REYNA-HURTADO. 2008. La dieta de los pecaries (*Pecari tajacu* y *Tayassu pecari*) en la región de Calakmul, Campeche, México. Revista Mexicana de Mastozoología 12:17-42.

POLLOCK, K. H., S. R. WINTERSTEIN, C. M. BUNCK, AND P.D CURTIS. 1989. Survival analysis in telemetry studies: the staggered entry design. The Journal of Wildlife Management 7-15.

PORTER, B. A. 2006. Evaluation of Collared Peccary Translocation in the Texas Hill Country Unpubl. Master Thesis, Texas A&M University.

POWELL, R. A., AND M. S. MITCHELL. 2012. What is a home range? Journal of Mammalogy 93: 948-958.

PRIDDEL, D., AND R. WHEELER. 1997. Efficacy of fox control in reducing the mortality of released captive-reared malleefowl, *Leipoa ocellata*. Wildlife Research, 24:469-482.

R CORE TEAM. 2016. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL http://www.R-project.org/.

REDFORD, K.H. 1992. The Empty of neotropical forest where the vegetation still appears intact. BioScience, 42:412–422.

RESENDE, L. D. S., G. L. E. NETO, P. G. D. CARVALHO, G. LANDAU-REMY, V. D. RAMOS-JÚNIOR, A. ANDRIOLO, AND G. GENARO. 2014. Time Budget and Activity Patterns of Oncilla Cats (*Leopardus tigrinus*) in Captivity. Journal of Applied Animal Welfare Science 17:73-81.

REYNA-HURTADO, R., E. ROJAS-FLORES, AND G. W. TANNER. 2009. Home range and habitat preferences of white-lipped peccaries (*Tayassu pecari*) in Calakmul, Campeche, Mexico. Journal of Mammalogy 90:1199-1209.

RICHTER, R. E. 2012. Home Range and Habitat Use of a Reintroduced Population of Collared Peccaries in the Llano Uplift Ecoregion of Texas. Unpubl. Master Thesis, Texas State University - San Marcos.

RICKETT, J., J. C. DEY, J STOTHART, C. M. O'CONNOR, J. S. QUINN, AND W. JI. 2013. The influence of supplemental feeding on survival, dispersal and competition in translocated Brown Teal, or Pateke (*Anas chlorotis*). Emu 113:62-68.

RIDOUT, M. S. AND M. LINKIE. 2009. Estimating overlap of daily activity patterns from camera trap data. Journal of Agricultural, Biological, and Environmental Statistics 14:322-337.

RIVADENEYRA, E. AND E. NARANJO. 2004. Estimación preliminar del ámbito hogareño de un grupo de pecaríes de collar (*Tayassu tajacu*) en la Reserva de la Biósfera "La Encrucijada", Chiapas, México. Memoria III Congreso de Estudiantes del Verano de la Investigación.

ROBINSON, J. G. AND J. F. EISENBERG. 1985. Group size and foraging habits of the collared peccary (*Tayassu tajacu*). Journal of Mammalogy 66:153-155.

ROSATTE, R., J. HARM, J. YOUNG, I. FILION, AND H. SMITH. 2007. The restoration of elk (*Cervus eplaphus*) in Ontario, Canada: 1998-2005. Restoration Ecology 15:34-43.

ROWCLIFFE, M. 2016. Activity: Animal Activity Statistics. R package version 1.1. https://CRAN.R-project.org/package=activity

ROWCLIFFE, J. M., R. KAYS, B. KRANSTAUBER, C. CARBONE, AND P. A. JANSEN. 2014. Quantifying levels of animal activity using camera trap data. Methods in Ecology and Evolution 5:1170-1179.

RUBENSTEIN, D. I. 1991. The greenhouse effect and changes in animal behavior: effects on social structure and life-history strategies. Pp. 180-192. In: Consequences of Global Warming for Biodiversity. R. Peters (ed.). Yale University Press.

RUMMEL, L., A. MARTÍNEZ–ABRAÍN, J. MAYOL, J. RUIZ–OLMO, F. MAÑAS, J. JIMÉNEZ, J. A. GÓMEZ, AND D. ORO. 2016. Use of wild–caught individuals as a key factor for success in vertebrate translocations. Animal Biodiversity and Conservation 39:207–219.

RYCKMAN, M. J., R. C. ROSATTE, T. MCINTOSH, J. HAMR, AND D. JENKINS. 2010. Postrelease dispersal of reintroduced elk (*Cervus elaphus*) in Ontario, Canada. Restoration Ecology 18:173-180.

SALAZAR, R. S. M. 2007. Dieta de los chanchos de monte: Taitetu (*Tayassu tajacu*) y Tropero (*Tayassu pecari*) en el territorio indigena Siriono, Beni, Bolivia. Revista Boliviana de Ecología y Conservación Ambiental 21:43-56.

SALTZ, D., M. ROWEN, AND D. I. RUBENSTEIN. 2000. The Effect of Space-Use Patterns of Reintroduced Asiatic Wild Ass on Effective Population Size. Conservation biology 14:1852-1861.

SANGUINETTI, J. AND T. KITZBERGER. 2010. Factors controlling seed predation by rodents and non-native *Sus scrofa* in *Araucaria araucana* forests: potential effects on seedling establishment. Biological Invasions 12:689-706.

SARKAR, M. S., K. RAMESH, J. A. JOHNSON, S. SEN, P. NIGAM, S. K. GUPTA, et al. 2016. Movement and home range characteristics of reintroduced tiger (*Panthera tigris*) population in Panna Tiger Reserve, central India. European Journal of Wildlife Research 62:537-547.

SARRAZIN, F. 2007. Introductory remarks – a demographic frame for reintroductions. Eco. Science 14: IV–V.

SCHAUB, M., R. ZINK, H. BEISSMANN, F. SARRAZIN, AND R. ARLETTAZ. 2009. When to end releases in reintroduction programmes: demographic rates and population viability analysis of bearded vultures in the Alps. Journal of Applied Ecology 46:92-100.

SCHULER, K. L., G. M. SCHROEDER, J. A. JENKS, AND J. G. KIE. 2014. Ad hoc smoothing parameter performance in kernel estimates of GPS-derived home ranges. Wildlife Biology 20:259-266.

SEDDON, P.J. 1999. Persistence without intervention: assessing success in wildlife reintroductions. Trends in Ecology and Evolution 14:503.

SEDDON, P. J., C. J. GRIFFITHS, P. S. SOORAE, AND D. P. ARMSTRONG. 2014. Reversing Defaunation: Restoring Species in a Changing World. Science 345:406–12.

SEIGEL R. A., AND C. K. DODD. 2002. Translocations of amphibians: proven management method or experimental technique? Conservation Biology 16:552-554.

SHEPPARD, J. K., H. D. MARSH, R. E. JONES, AND I. R. LAWLER. 2010. Dugong habitat use in relation to seagrass nutrients, tides and diel cycles. Marine Mammal Science 26:855–79.

SHIER, D. M. AND D. H. OWINGS. 2006. Effects of predator training on behavior and post-release survival of captive prairie dogs (*Cynomys ludovicianus*). Conservation Biology 132:126-135.

SHIER, D. M. AND R. R SWAISGOOD. 2012. Fitness Costs of Neighborhood Disruption in Translocations of a Solitary Mammal. Conservation Biology 26:116–23.

SIGNER, J., AND N. BALKENHOL. 2015. Reproducible home ranges (rhr): A new, user-friendly R package for analyses of wildlife telemetry data. Wildlife Society Bulletin 39:358-363.

SLOTTA-BACHMAYR, L., R. BOEGEL, P. KACZENSKY, C. STAUFER, AND C. WALZER. 2004. Use of population viability analysis to identify management priorities and success in reintroducing przewalski's horses to southwestern Mongolia. Journal of Wildlife Management 68:790-798.

SOULÉ, M. AND R. NOSS. 1998. Complementary Goals for Continental Conservation. Wild Earth 1-11.

SPENCER, S. R., G. N. CAMERON, AND R. K. SWIHART. 1990. Operationally defining home range: temporal dependence exhibited by hispid cotton rats. Ecology 71:1817-1822.

STAMPS, J. A. AND R. R. SWAISGOOD. 2007. Someplace like home: experience, habitat selection and conservation biology. Applied Animal Behaviour Science 102:392-409.

SWAISGOOD, R.R. AND B. A. SCHULTE. 2010. Applying knowledge of mammalian social organization, mating systems and communication to management. In: Kleiman DG, Thompson KV, Baer CK, editors. Wild Mammals in Captivity, 2nd edition Chicago: University of Chicago Press.

SWEIKERT, L. AND M. PHILLIPS. 2015. The effect of supplemental feeding on the known survival of reintroduced aplomado falcons: Implications for recovery. Journal of Raptor Research 49:389-399.

TABER, A. B., C.P. DONCASTER, N.N. NERIS, AND F. COLMAN. 1994. Ranging Behaviour and Activity Patterns of Two Sympatric Peccaries, *Catagonus Wagneri* and *Tayassu Tajacu*, in the Paraguayan Chaco. Mammalia 58:61–71.

TABER, A., M. ALTRICHTER, H. BECK, AND J. GONGORA. 2011. The Tayassuidae. In D. E. Wilson and R. A. Mittermeier (eds.). Handbook of the Mammals of the World: Hoofed Mammals, pp. 292-307. Lynx Edicions Barcelona, Spain.

TAYLOR, S. S., I. G. JAMIESON, AND D. P. ARMSTRONG. 2005. Successful island reintroductions of New Zealand robins and saddlebacks with small numbers of founders. Animal Conservation 8:415-420.

TEIXEIRA, C.P., C. S. DE AZEVEDO, M. MENDL, C. F. CIPRESTE, AND R.J. YOUNG. 2007. Revisiting translocation and reintroduction programmes: the importance of considering stress. Animal Behaviour 73:1–13.

TERBORGH, J., G. NUÑEZ-ITURRI, N. PITMAN, F. H. CORNEJO VALVERDE, P. ALVAREZ, V. SWAMY, E. G. PRINGLE AND C. E. TIMOTHY PAINE. 2008. Tree Recruitment in an Empty Forest. Ecology 89 (6):1757–68.

TERBORGH, J. 2013. Using Janzen–Connell to predict the consequences of defaunation and other perturbations in tropical forests. Biological Conservation. 163:7–12.

TICER, C. L., R. A. OCKENFELS, J. C. DEVOS JR, AND T. E. MORRELL. 1998. Habitat use and activity patterns of urban-dwelling javelina. Urban Ecosystems 2:141-151.

TOBLER, M.W. 2015. Camera base version 1.3. http://www.atriumbiodiversity.org/tools/camerabase/

TOBLER, M.W., S. E. CARRILLO-PERCASTEGUI, R. LEITE PITMAN, R. MARES, AND G. POWELL. 2008. An evaluation of camera traps for inventorying large-and medium-sized terrestrial rainforest mammals. Animal Conservation 11:169-178.

TOBLER, M. W., S. E. CARRILLO-PERCASTEGUI, AND G. POWELL. 2009. Habitat use, activity patterns and use of mineral licks by five species of ungulate in south-eastern Peru. Journal of Tropical Ecology 25:261-270.

TOONE, W., J. CAMPOS, AND M. WALLACE. 2003. Release of giant radio-collared Chacoan peccary in Paraguay. Reintroduction News 22:34-36.

TRESSENS, S. G., R. O. VANNI, AND M. G. LOPEZ. 2002. Las plantas terrestres. In Flora del Ibera (M. M. Arbo and S. G. Tressens, eds.). Editorial Universitaria de la Universidad Nacional del Nordeste, Corrientes, Argentina. Pp. 201–379.

TWEED, E. J., J. T. FOSTER, B. L. WOODWORTH, P. OESTERLE, C. KUEHLER, A. LIEBERMAN, et al. 2003. Survival, dispersal, and home-range establishment of reintroduced captive-bred puaiohi, *Myadestes palmeri*. Biological conservation 111:1-9.

UBEDA, B., A. DI GIACOMO, J. J. NEIFF, S. A. LOISELLE, G. POI, J. GALVEZ, S. CASCO, AND A. COZAR. 2013. Potential Effects of Climate Change on the Water Level, Flora and Macro-fauna of a Large Neotropical Wetland. PloS one 8:677-687.

VAN WINKLE, W. 1975. Comparison of several probabilistic home-range models. Journal of Wildlife Management 39:118-123.

VIGGERS, K L., D. LINDENMAYER AND D. SPRATT. 1993. The Importance of Disease in Reintroduction Programmes. Wildlife Research 20:687–98.

VENABLES, W. N. AND B. D. RIPLEY. 2002. Modern Applied Statistics with S. Fourth Edition. Springer, New York.

WANLESS, R. M., J. CUNNINGHAM, P. A. R. HOCKEY, J. WANLESS, R.W. WHITE, AND R. WISEMAN. 2002. The success of a soft-release reintroduction of the flightless Aldabra rail on Aldabra Atoll, Seychelles. Biological Conservation 107:203– 210.

WECKEL, M., W. GIULIANO, AND S. SILVER. 2006. Jaguar (*Panthera onca*) feeding ecology: distribution of predator and prey through time and space. Journal of zoology 270:25-30.

WHITE, G. C. AND K. P. BURNHAM. 1999. Program MARK: survival estimation from populations of marked animals. Bird Study 46:120–139.

WHITE, G. C. AND R. A. GARROTT. 2012. Analysis of wildlife radio-tracking data. Elsevier.

WILSON, A. C. AND M. S. PRICE. 1994. Reintroduction as a reason for captive breeding. In: Olney, P. J., G. Mace, and A Feistner. (Eds.). Creative conservation. pp. 243-264. Springer Netherlands.

WORTON, B. J. 1989. Kernel methods for estimating the utilization distribution in home-range studies. Ecology 70:164-168.

WRIGHT, S.J. 2003. The myriad consequences of hunting for vertebrates and plants in tropical forests. Perspectives in Plant Ecology, Evolution and Systematics 6:73–86.

YOTT, A., R. ROSATTE, J. A. SCHAEFER, J. HAMR, AND J. FRYXELL. 2011. Movement and spread of a founding population of reintroduced elk (*Cervus elaphus*) in Ontario, Canada. Restoration Ecology 19:70-77.

ZIDON, R., SALTZ, D., L. S. SHORE, AND MOTRO, U. 2009. Behavioral Changes, Stress, and Survival Following Reintroduction of Persian Fallow Deer from Two Breeding Facilities. Conservation Biology 23:1026–1035.

Cindy M. Hurtado Martínez

cindymeliza@gmail.com

EDUCATION

- Master of Science in Biology. Towson University, Maryland – Date to be conferred: 05/2017
- Bachelor in Biological Sciences; Zoology Minor. Universidad Nacional Mayor de San Marcos, Lima, Peru 12/2011

PUBLICATIONS

- **Hurtado C.M.,** V. Pacheco, U. Fajardo & A. Uturunco. 2016. An updated analysis of the distribution of cites Peruvian carnivores for conservation efforts. Mastozoología Neotropical, 23: 415 - 429
- **Hurtado C.M.**, J. Serrano-Villavicencio and V. Pacheco. 2016. Population density and primate conservation in the Noroeste Biosphere Reserve, Tumbes-Peru. Revista Peruana de Biología 23: 151 – 158.
- García-Olaechea A. & **C. M. Hurtado**. 2015. Distribution and conservation of the pampas cat in northwestern Peru. Mammalogy Notes 2: 21.
- Divoll T. J., A. Kumar, C. F. Flores-Negron & **C. M. Hurtado**. Maternity roost of *Eptesicus brasiliensis* in a liana in the southeast Peruvian amazon. 2015. Mastozoología Neotropical 22: 155 - 161.
- **Hurtado C.M.** & V. Pacheco. 2015. New mammalian records in the Parque Nacional Cerros de Amotape, northwestern Peru. Revista Peruana de Biología 22: 77 - 86.

ORAL PRESENTATIONS (presenter is underlined)

- Hurtado C. M, E. Galleto, I. Jimenez & H. Beck. Preliminary data on the reintroduction of collared peccaries (Pecari tajacu) in the Ibera Provintial Park in Argentina. III Latin-American Mammalogy meeting- Bogota, Colombia, 12/2015.
- García-Olaechea A. & C. M. Hurtado. Datos preliminares sobre la distribución y amenazas del gato de pajonal (*Leopardus colocolo*) en el noroeste de Perú. III Latin-American Mammalogy meeting- Bogota, Colombia, 12/2015.
- Fajardo U., V. Pacheco, C. M. Hurtado & A. Uturunco. GIS as a tool for carnivore conservation. III Congreso de la Sociedad Peruana de Mastozoología- Piura, 10/2012.

POSTER PRESENTATIONS (presenter is underlined)

- Hurtado C. M. Video-monitoring of the mammal community in wallows and streams at the Manu National Park, Peru. III Latin-American Mammalogy meeting- Bogota, Colombia, 12/2015.
- Hurtado C. M. & V. Pacheco. Population status of the black howler monkey (*Alouatta palliata aequatorialis*) in the Noroeste Biosphere Reserve. What do we really know? II Symposium of Primatology. Iquitos, Peru, 11/2013.
- Hurtado C. M. & V. Pacheco. First assessment of medium and large mammals diversity in the Pacific Tropical Rainforest of Peru using Line transect census and camera traps. 93rd American Society of Mammalogists Annual Meeting. Philadelphia, 06/2013.

- Serrano-Villavicencio J. & C. M. Hurtado. Taxonomic description *Alouatta* (PRIMATES: ATELIDAE) distributed in southern Peru. III Congreso de la Sociedad Peruana de Mastozoología - Piura, 10/2012.

GRANTS

- García-Olaechea A. & C. M. Hurtado. Spatial ecology and conservation of the pampas cat *Leopardus colocolo* in the Sechura Desert of northwestern Peru. 5/ 2016. Mohamed bin Zayed Species Conservation Fund. Awarded $10,640.
- Hurtado C. M. & H. Beck. Geospatial Analyses and Ecological Assessment of the Reintroduction of a Large Mammal Species (Peccaries, *Pecari tajacu*) into a Nature Reserve in Argentina. 1/ 2015. Phoenix Zoo. Awarded $2,900.
- Hurtado C. M. Travel grant for presenting at the 93rd American Society of Mammalogists Annual Meeting. Philadelphia, 06/2013. Consejo Nacional de Ciencia y Tecnologia (CONCYTEC). Awarded $1,100.
- Hurtado C. M. & V. Pacheco. Medium and large mammal's abundance in the Cerros de Amotape National Park, Tumbes – Peru. 07/2012. Rufford Small Grants. **Awarded** $8,500.
- Hurtado C. M. & V. Pacheco. Medium and large mammal's abundance in the Cerros de Amotape National Park, Tumbes – Peru. 07/2012. Idea Wild. **Awarded** $1,500.

PROFESSIONAL EXPERIENCE

Project Leader

- Geospatial analyses and ecological assessment of the reintroduction of a range mammal species (peccaries, *Pecari tajacu*) into a Nature Reserve in Argentina. 08/2014- 05/2017
- An updated analysis of the distribution of cites Peruvian carnivores for conservation efforts. 11/2010 – 12/2015.
- Video-monitoring of the mammal community in wallows and streams at the Manu National Park, Peru. 07-12/2012.
- Medium and large mammal's abundance in the Cerros de Amotape National Park, Tumbes 10/2011- 04/2012.

Co-instructor

- Camera trapping for wildlife studies - field course. BioS Peru. May 12th -14th 2015– Lima, Peru.

Teaching Assistant

- Biology 120L. Towson University. 08/2014 – 05/2017 Towson, USA.
- Tropical Biology-Field Course. Organization for Tropical Studies. 06-07/2014- Costa Rica.

Field and Research Assistant

- Small mammals monitoring. Walsh Environmental Company. Cuzco, 11 -12/2011
- Small mammals monitoring. Lavalin – Perú . Pampas de cobre (Mining Company), Moquegua. 09/2011.
- Small mammals monitoring. Golder Associates Company- Perú. Barrick (Mining Company), La Libertad. 11/2010.