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Combining citizen science and recreational hunters to monitor exotic ungulates and native wildlife in a protected area of northeastern Argentina

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Abstract Monitoring wildlife population trends is essential for resource management and invasive species control, but monitoring data are hard to acquire. Citizen science projects may monitor species occurrence patterns in time and space in a cost-effective way. A systematic management program of exotic wild boar (*Sus scrofa*) and axis deer (*Axis axis*) in a protected area of northeastern Argentina (El Palmar National Park) provided a framework for implementing a wildlife monitoring system based on park-affiliated hunters. We assessed the level of agreement between three indices of relative abundance: hunter sightings and camera trapping for wild boar, axis deer, capybaras (*Hydrochoerus hydrochaeris*), brown brocket deer (*Mazama guazoubira*), and crab-eating and pampas foxes combined (*Cerdocyon thous* and (*Lycalopex gymnocercus*), and catch per unit effort (CPUE) for both exotic ungulates only. Most (74%) hunting parties participated in the monitoring program and contributed to its sustainability. Bland-Altman plots displayed large levels of agreement between methods across species, with larger systematic differences between sighting and camera-

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trapping indices for native species. Restricting camera-trapping to the same time window as hunter sightings substantially increased the agreement between methods across species. Sighting and CPUE indices revealed similar temporal trends and large variations in spatial patterns between species. Comparison of the number of sighted and killed exotic ungulates indicated that, on average, 17% of wild boar and 75% of axis deer escaped hunters. The three indices were appropriate metrics for management purposes and corroborated the sustained, high-level abundance of axis deer and low numbers of wild boar in recent years.

Keywords Citizen science · Wildlife management · Method agreement · Invasive ungulates · Axis deer · Wild boar

Introduction

Information on the population trends of invasive exotic species that cause significant damage is necessary for management purposes (Blossey 1999). These species frequently require a costly early warning system to detect new invasion foci and long-term monitoring of abundance to assess its degree of expansion and the outcomes of management actions. Monitoring is crucial to make informed decisions within an adaptive management scenario with clearly defined objectives and hypothesis (Nichols and Williams, 2006). Regular monitoring systems are scarce in developing countries, probably due to its high implementation costs at appropriate spatial and temporal scales. Lack of knowledge of wildlife population trends and concomitant environmental changes hinder the fast detection and comprehension of eventual or induced changes in the abundance of targeted exotic species (Danielsen et al. 2009).

Monitoring studies designed to detect invasive species' occurrence patterns are especially compatible with a citizen science project, defined as scientific research in which all or part of the data are provided by non-scientist volunteers, and the design, analysis and interpretation of the results is carried out by researchers, although other configurations exist (Danielsen et al. 2009; McKinley et al. 2017; Chandler et al. 2017). Crall et al. (2015) showed that integrating volunteer with professionally-obtained data generated more realistic habitat suitability models for several invasive plant species at a large scale. Delaney et al. (2008) engaged volunteers to successfully generate a standardized database on distribution and abundance of invasive and native crabs. Other projects take advantage of technological platforms or equipment to gather observational data and promote sustained participation, such as Snapshot Serengeti (Swanson et al. 2015), in which volunteers classify camera-trap images online. In Europe, hunters are important sources of citizen science data, such as catch per unit effort (CPUE) indices, sightings, biometric data and biological samples (Cretois et al. 2020). In South America, hunters have been engaged in research by collecting biological samples in the Amazon region (Mayor et al. 2017), and by participating in community-based monitoring programs based on CPUE indices (El Bizri et al. 2021, Noss et al. 2005). These studies provided information on the temporal trends of wildlife populations (mainly native species) and parameters only known for captive populations, often for the sustainable management of subsistence hunting (De Mattos Vieira et al. 2015).

Citizen science projects commonly take advantage of opportunistic sightings by interested volunteers with knowledge on local biodiversity (Sullivan et al. 2009; Toms and Newson 2006; Evans et al. 2005; Cannon et al. 2005). These projects rarely control for sampling effort, observer bias and protocol, and therefore deliver unstructured or "messy data" (Dobson et al. 2020). Therefore, researchers frequently need to minimize the possibility of incorrect observations (e.g., confused species names) and observer bias (e.g., different levels of commitment over time, data aggregation to roads or highly populated areas), which determine the internal validity and utility of the data sets (Dobson et al. 2020).

One way to take advantage of "messy" data is to use an integrated approach that includes an unbiased method. Camera traps have been rapidly adopted as a non-invasive, cost-effective method to investigate wildlife distribution, abundance and behavior (Burton et al. 2015). Camera traps provide indices of relative abundance for species with non-distinguishable phenotypic traits, although several analytical frameworks have been developed to estimate absolute densities for unmarked individuals (Gilbert et al. 2021). In general, the validity of relative abundance indices has often been controversial, and its assessment challenging (Engeman 2005; Skalski et al. 2005; Anderson 2003). When population size cannot be estimated with the desired accuracy, the level of agreement between simultaneous indices can be evaluated to identify how and in which situations both methods differ, and thus provide a measure of relative validity. One relevant example of assessing agreement between two methods is Bland-Altman plot analysis (Bland and Altman 1986, 1999; Parker et al. 2016). Originally developed in the biomedical field where it gained prominence, Bland-Altman plot analysis has recently been applied to ecological research (Schaus et al. 2020).

Invasive exotic species are often targeted for control when eradication is not feasible (Hulme 2006). Mammals, in particular, have large invasive success and may cause severe environmental and economic impacts (Jeschke 2008). Wild boar *Sus scrofa* and chital or axis deer (*Axis axis*) significantly affect biodiversity, and frequently warrant management actions in their exotic range (Lowe 2004; Hone 2002; Davis 2016; Hess et al. 2015). Both exotic ungulates are present in multiple protected areas of Argentina (Merino et al. 2009, Lizarralde 2016), including El Palmar National Park (hereafter the park). The large impacts of wild boar on the park's main conservation value (the iconic yatay palm tree *Butia yatay*), followed by a large increase in axis deer numbers, prompted the continued implementation of a systematic management program of both exotic ungulates since 2006 (Gürtler et al. 2017, 2018).

This program affiliated local sport hunters to conduct still shooting from 47 elevated blinds (as of 2017) in 2–5 monthly sessions mainly conducted in the evening. The effectiveness of control efforts was mainly evaluated through the standardized CPUE of each species over a defined time window; this metric correlated reasonably well with more sparse data on rooting indices (wild boar) and deer spotlight counts (Gürtler et al. 2017, 2018). Whether CPUE indices reflect the true changes in the population size of wild boar (which apparently declined over time) and axis deer (which steadily increased over time) was deemed rather uncertain. CPUE indices have been criticized for being closely related to the details of the hunting process such as logistics, gear quality, and hunter skills (Lancia et al. 1996; Skalski et al. 2005). Therefore, other independent indices are required to establish the reliability of CPUE-based population trends. The management program of exotic ungulates at El Palmar National Park provided a perfect opportunity for a citizen science project in which the park-affiliated hunters recorded wildlife sightings from their elevated blinds deployed across the park, and thus supplied a new index of relative abundance at virtually no additional cost. We also used camera-trapping indices as a hunter-independent metric of the relative abundance of exotic ungulates and other native wildlife species: capybaras (*Hydrochoerus hydrochaeris*), brown brocket deer (*Mazama gazoubira*), and crab-eating and pampas foxes combined (*Cerdocyon thous* and (*Lycalopex gymnocercus*, respectively).

In this study we describe the design, implementation and main outcomes of a wildlife monitoring system based on hunter sightings over a six-month period. We assessed the relative validity of sighting, CPUE and camera-trapping indices for wild boar and axis deer; extended the comparison between sighting and camera-based indices to selected native wildlife of interest, and compared the temporal and spatial trends of relative abundance indices. We expected an increasing participation from hunting parties by the end of the study due to increasing motivation. We predicted that hunter sightings of both exotic ungulates will agree with CPUE and camera-trapping indices, as will sighting and camera-trapping indices for the native wildlife.

Materials and methods

Study area

El Palmar National Park is located in Entre Ríos Province (31° 55'–S, 58° 16'–W), northeastern Argentina (Fig. 1). It covers approximately 8,500 ha of *B. yatay* palm-tree savannas, grasslands, wetlands and gallery forests in a matrix of forest plantations and crops (Batista et al. 2014). The mean temperature recorded by the park's weather station between March and August 2017 was 15 °C (minimum, −2.5 °C and maximum, 31 °C); total rainfall was 960 mm. The park is limited by the Uruguay River on the east and a fast highway (route 14) on the west. This determines a humidity gradient tracked by exotic trees and shrubs (*Melia azedarach*, *Pyracantha atalantoides*, *Gleditsia triacanthos, Ligustrum lucidum* and *Ligustrum*

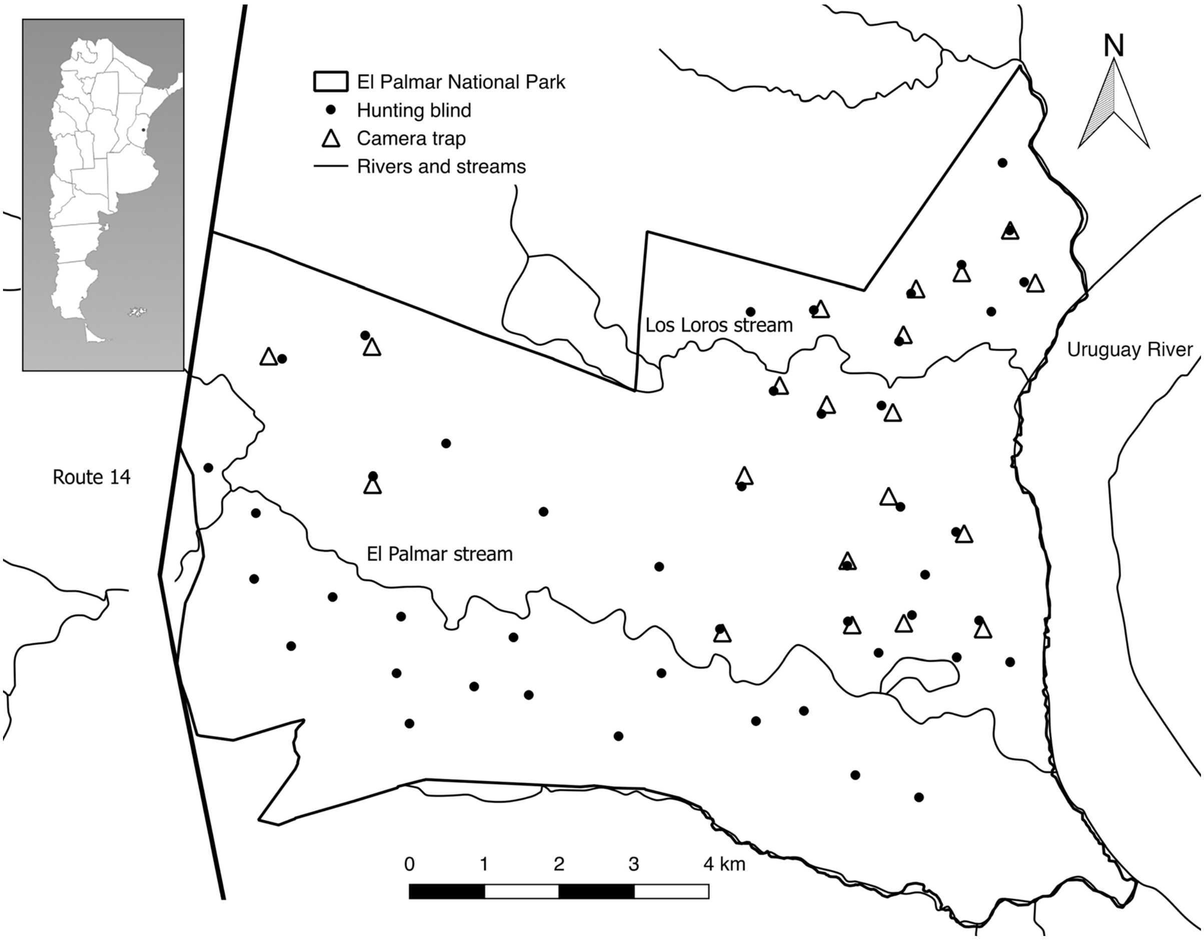

Fig. 1 Location of El Palmar National Park in northeast Argentina, and distribution of hunting blinds (filled circles) and camera traps (open triangles). El Palmar stream divides the park in two zones subject to different public-use regimes

sinense) (Ruiz Selmo et al. 2007). El Palmar stream delimitates a northern zone for public use (including 30 active blinds), and a southern zone (with 17 active blinds) of high conservation value closed to park visitors (Fig. 1). Poaching has historically been frequent despite park ranger efforts, especially in the southern zone.

Native medium-sized wildlife includes capybaras, crab-eating and pampas foxes, brown brocket deer, plains viscachas (*Lagostomus maximus*), three small felines (*Leopardus geoffroyi*, *Leopardus colocolo* and *Herpailurus yagouaroundi*), armadillos (*Dasypus novemcinctus*, *Euphractus sexcinctus*), crab-eating raccoons (*Procyon cancrivorus*), lesser grisons (*Galictis cuja*) and the nationally endangered greater rheas (*Rhea americana*) (Crespo 1982).

Management of exotic wild boar and axis deer

The program mainly consisted in shooting with rifles from elevated hunting blinds (watchtowers) in sessions occurring between 18:00 and 23:00 h every one or two weeks from February-March to December. Program goals, hunting effort and outcomes over 2006–2015 were described elsewhere (Gürtler et al. 2017, 2018). Every blind is occupied and maintained by an identified group of hunters (a hunting party, composed of 2 or 3 people), who have used the same blind over prolonged periods of time (range, 1 to 10 years), and are organized in two stable hunting associations. Hunters have a broad vision from the elevated blind and are only allowed to shoot along radial shooting lines where the vegetation has been removed and the bait (rotting maize and salt blocks) is set once or twice a week. The direction and total length (100–300 m) of shooting lines at each blind is determined by park management based on safety criteria. Each hunter party was required to carry a VHF radio to communicate with the park ranger who managed the session at a central operating post and request permission to shoot when they were sure of a clean kill or to put down a wounded animal; they were not allowed to chase specimens. No harvest quotas were set; hunters committed to conduct nonselective hunting and were allowed one annual trophy to each party. At the end of each hunting session, every hunting party brings the quarry to a central operating post for processing and measurement, and park rangers fill in a form regardless of whether the hunting party killed any specimen or not. Although in the past hunters had been asked to report wildlife sightings during sessions, this information was registered rather erratically. The CPUE data and GPS location of each blind were retrieved from the information collected by the park. Hunting of native species is not allowed.

Sighting monitoring system

Every year park management conducts a meeting with all program-affiliated hunters prior to re-initiating hunting sessions after summer holidays to review the operating procedures and issues. At the annual meeting held on early March 2017, we explained the aims of the citizen science project to hunters and invited them to participate as wildlife monitors to generate a long-term, spatially explicit database of selected wildlife species. Both hunters and park personnel provided feedback on the survey forms before starting hunting sessions. We further promoted hunter participation via informal conversations and exchange of observations before and after each hunting session. Each hunting party was provided with a survey form prior to every session held from 22 March until 30 August 2017, resulting in 16 survey occasions. We considered that a hunting party participated in a wildlife monitoring session if they returned the form regardless of whether they recorded any species. Park rangers and volunteers delivered, checked and stored the survey forms.

The survey form aimed at registering several species of interest (i.e., wild boar, axis deer and native capybara, both fox species and brown brocket deer). Other medium- or large-sized native wildlife (such as greater rhea and armadillos, among others) were recorded. Hunters recorded the identity of their blind, date, and time at which they began and finished the sighting session; these measures were used to compute a blind-specific sighting effort by session. When hunters sighted an individual, they recorded the time, species, number of individuals, and any discretionary comment. Both local fox species were difficult to distinguish at night; therefore, hunters registered them as “foxes”. Since most hunters were well acquainted with the local wildlife, species misidentifications were assumed to be rare.

Camera trapping

We selected 20 blinds located in the park’s northern zone for camera deployment in order to cover different sections as uniformly as possible while minimizing the risk of poaching (Fig. 1). We deployed 10 camera traps (Browning Strike Force HD Trail Camera BTC-5HD with infrared sensor) in two successive stages covering 20 sites. Each camera was set within a 100 m radius from each blind in natural vegetation between shooting lines, and. was active over a five- or seven-day period from August 4 to 20, 2017. Total camera-trap effort was 2802 h. At each blind, site selection took into account the presence of signs (tracks, fecal pellets, trails) of native and exotic wildlife. The cameras were set on trees 30–100 cm above the ground and protected from direct sunlight, and unbaited. We chose a time delay of 20 sec and a rapid fire mode (4 pictures per detection event) following established guidelines (Meek et al. 2014). Detection events were taken as independent when consecutive photos of the same species were separated by at least one minute.

Data analysis

We calculated the proportion of hunting sessions in which a hunting party was both present and delivered a survey form (i.e., participated), and scored their degree of participation over the study period as excellent (0.75–1), good (0.5-0.74) and regular (<0.5). We checked for temporal and spatial (blinds in southern versus northern section) patterns in the frequency of survey forms handed in.

The current report focuses on wild boar, axis deer, capybaras, foxes, and brown brocket deer. We calculated indices of relative abundance for each species and method (sighting, hunting and camera trapping) as the sum of specimens recorded by each method per 100 h of effort. These results were expressed as sighting indices per unit effort (SPUE, by hunter sighting), catch per unit effort (CPUE, by hunting, only for exotic ungulates) and photographic catch per unit effort (PhCPUE, by camera trapping) by hunting blind and session.

We evaluated the agreement between indices of relative abundance using Bland-Altman plots (Bland and Altman 1986), also known as Tukey mean difference plots. The former consists in plotting the difference between a pair of quantitative measures (e.g., SPUE–CPUE) of a single individual or survey unit (i.e., a hunting blind) against the measurement average ([SPUE + CPUE]/2) over a defined time period. If there was agreement between methods, these differences would be distributed normally, 95% of them between the mean difference ($\overline{d}$, average bias) and two standard deviations. This range constitutes the limits of agreement: $LoA = \overline{d} \pm 2 * SD$ (Bland and Altman 1986, 1999). The average bias indicates by how much one method over- or under-estimates the other; when both methods return the same measurements the average bias is zero. A small range of LoA suggests better agreement. Since in our study the differences between measurements were linearly related to their mean, we transformed the indices by adding 1 to the sum of specimens recorded (sighted, killed, or photographed) for each species and then taking their logarithm (to the base 10), and plotted the log-ratio (i.e., [log SPUE–log PhCUE]) against the log-mean [(log SPUE–log CPUE)/2] (Bland and Altman 1999).

The three sampling methods varied in (a) the number of species monitored: CPUE was restricted to axis deer and wild boar; (b) spatial coverage: PhCPUE was available for 20 hunting blinds; CPUE covered all active blinds, and SPUE only the participating hunting parties; and (c) temporal scale: camera trapping covered a 24 h-period over several days in winter, whereas CPUE and SPUE covered an average of 5 h per session over several months. Therefore, in order to refine comparisons of the level of agreement between methods, we restricted hunting and sighting data to the 20 selected blinds in which the cameras had been deployed, and to concurrent hunting sessions that coincided with or occurred within 10 days of camera-trapping efforts. The four hunting sessions included occurred on July 26, and August 5, 9 and 30, 2017. We assumed no significant temporal correlation. For Bland-Altman analyses, camera-trapping data were restricted to the interval between 6 and 11 p.m. (i.e., PhCPUE_res) to match the timing and approximate duration of hunting sessions. Non-restricted camera-trapping data (PhCPUE) were also included to assess the importance of having the same the time frame for agreement analysis.

We report the mean indices of relative abundance by species and survey method for winter season (including mean PhCPUE for restricted and unrestricted datasets), and two relevant outcomes based on sighting and hunting data: temporal trends in mean logSPUE and logCPUE (only for exotic ungulates) throughout the study period, and the spatial structure of logSPUE for axis deer, wild boar, capybara and brown brocket deer at parkwide scale. We did not restrict the SPUE and CPUE data (spatially or temporally) for these analyses. We refrained from mapping out the outcomes of camera traps because they only covered 20 sites in the northern zone, whereas hunter sightings and CPUE data covered 47 sites in both zones.

All analyses were performed in R-software v. 3.6.3 (R Development Core Team 2020) using the suit of 'Tidyverse' v.0.3.2 (Wickham 2019) and 'Rmisc' v.0.3.2 (Hope, 2013) for data management and analysis; 'ggplot2' v.0.3.2 (Wickham 2016) for plotting; 'sf' v.0.3.2 (Pebesma 2018), and 'tmap' v.0.3.2 (Tennekes 2018) and QGIS software (QGIS.org 2021) for map making.

Results

Hunter participation

In total, we collected 492 survey forms that amounted to 2283 h of sighting effort over 16 hunting sessions. On average, 30.8 (range, 24–42) hunting parties returned a survey form per session, and 40.7 (range, 38–44) were present. Overall participation by session averaged 76% and displayed no temporal trend (Fig. 2a). Hunting parties participated in wildlife monitoring in 74% of the sessions they attended to regardless of park zone (Fig. 2b). Only one party (present in 4 of 16 sessions) did not contribute to

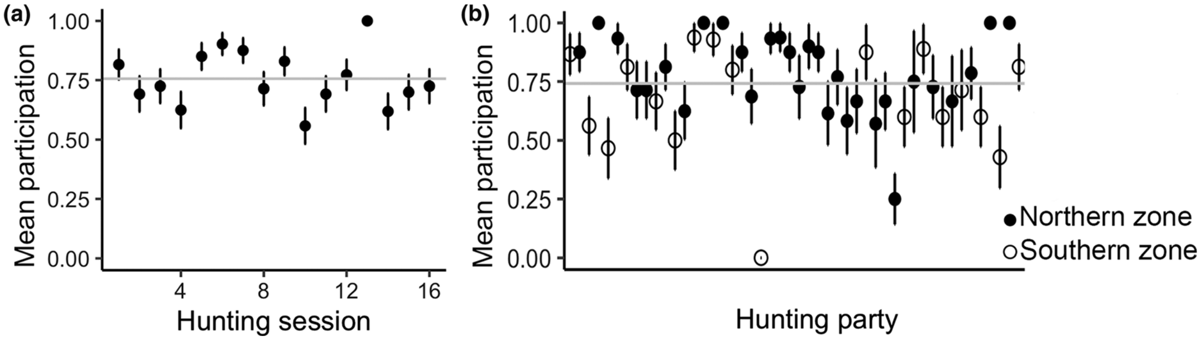

Fig. 2 Mean participation of hunting parties over each hunting session conducted between March and August 2017 (**a**), and per hunting party in the northern (filled circles) and southern (open circles) zones (**b**). The solid lines represent the mean proportion of participating hunting parties

wildlife monitoring. Participation was scored excellent in 51% of the groups (i.e., delivered a form in 75–100% of sessions); good in 38% (50–74%), and regular in the remaining 11% (<50%).

Method agreement

All methods showed good agreement since 95% of the log-ratios were within the LoA. The only exception was for the SPUE-CPUE comparison for wild boar, which showed 90% of the log-ratios included within the LoA even though all other points were evenly distributed around the geometric mean ratio (Fig. 3b). The distribution of log-ratios was not homogenous, and showed the largest dispersion in comparisons including camera-trapping indices (PhCPUE_res, from 9 to 140 units of difference in LoA range, and PhCPUE, from 36 to 346 units of difference in LoA range) (Table 1). For both axis deer and wild boar, SPUE and CPUE displayed narrower LoA, with 2–4 units of difference. For all comparisons and species, the SPUE index displayed significantly higher values than any other index except for wild boar.

The mean log-ratio between SPUE and CPUE for axis deer was 0.24 (SE, 0.08). Taking its antilogarithm, we obtain the geometric mean ratio of SPUE to CPUE, 1.75 (95% CI, 1.46 to 2.10); thus, on average, the SPUE index was 75% higher than the CPUE index. Two hunting parties removed more axis deer than they reported seeing, which are represented as negative LoA values (Fig. 3a). For wild boar, the mean log-ratio was 0.07 (SE, 0.09), and the geometric mean ratio was 1.17 (95% CI, 0.96 to 1.43); hence, on average, the sighting index was 17% higher than the CPUE, but this difference was not statistically significant. All values were closely distributed around the mean log-ratio, except for the two data points that fell outside the LoA (Fig. 3b).

Geometric mean ratios between CPUE or SPUE and PhCPUE_res were 3.37–5.89 for axis deer and 1.99–2.42 for wild boar (Table 1). The agreement between PhCPUE_res and other methods was good though weaker than that recorded between SPUE and CPUE, as reflected in a broader range of LoA ($\sim$10–20 versus 2–4 units). The SPUE-PhCPUE_res comparisons for all native species were in good agreement, with 95% of log-ratios within the LoA. The mean difference between methods was maximum for capybaras (16.68; 95% CI, 11.14 to 24.97) and minimum for brown brocket deer (2.19; 95% CI, 1.68 to 2.86) (Table 1). Comparisons between CPUE or SPUE and PhCPUE showed less agreement than with the time-restricted PhCPUE_res index for all species (Table 1). There was no discernible spatial structure of the log-ratios between PhCPUE_res and SPUE for any species (Supplementary material 1).

Relative abundance indices

The relative abundance of capybaras, foxes and axis deer greatly exceeded that of wild boar and brown brocket deer as determined by SPUE and PhCPUE, except for PhCPUE_res, computed for the 6–11 p.m. time band (Table 2). The mean PhCPUE_res indices for wild boar and axis deer were the same ($\sim$1.7 animals per 100 h) whereas comparisons of both species by any other two indices indicated that axis deer were approximately 2.7–4 times more abundant

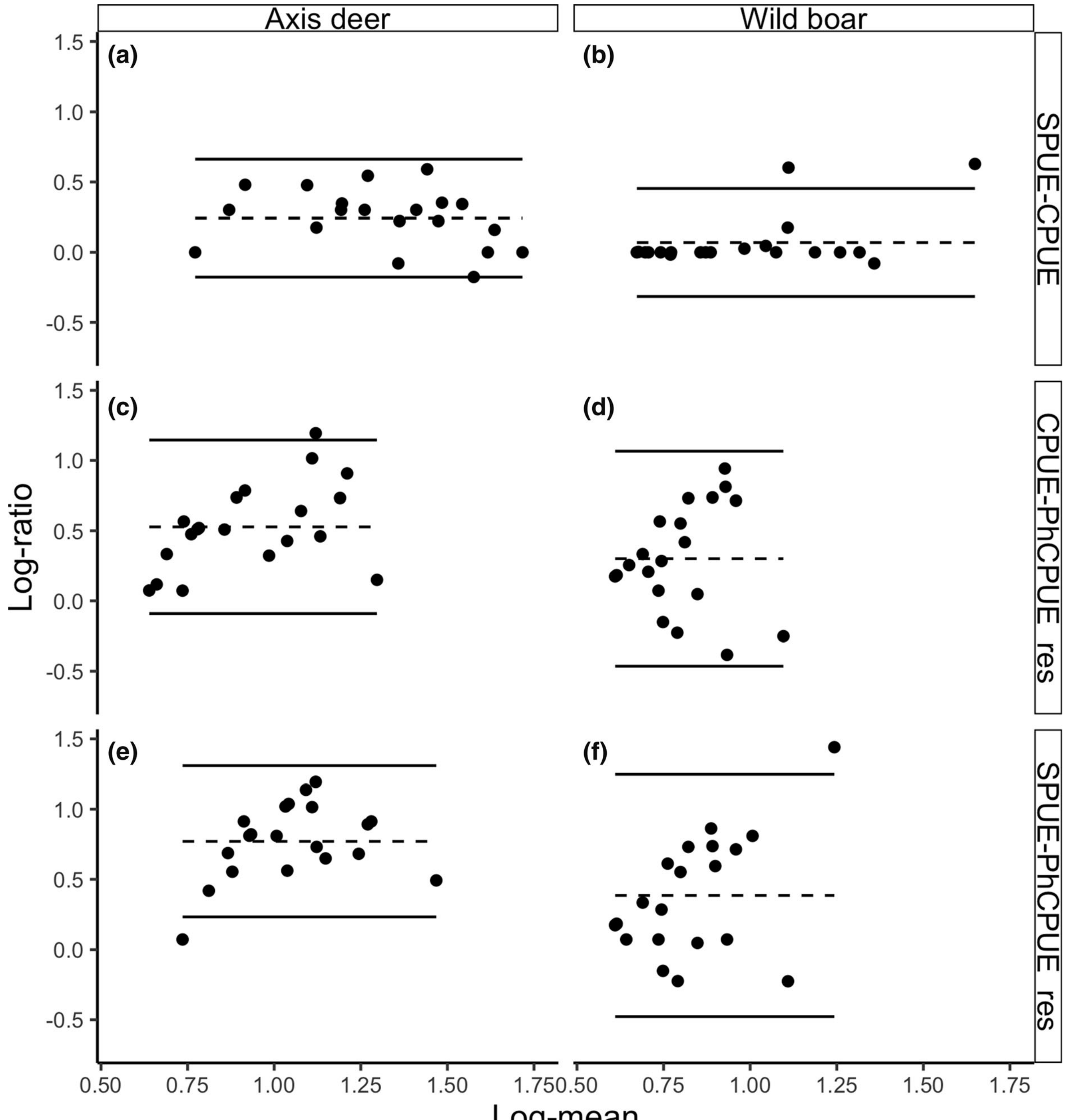

Fig. 3 Bland-Altman plots of the log-ratio between indices of relative abundance of axis deer and wild boar versus their log-mean by hunting blind, for comparisons between SPUE and CPUE (**a**,**b**), CPUE and PhCPUE_res (**c**,**d**), and SPUE and PhCPUE_res (**e**,**f**). Dashed lines represent the geometric mean ratio ($\bar{d}$) and solid lines the limits of agreement

than wild boar (Table 2). The mean PhCPUE_res index for wild boar more than doubled the unrestricted mean PhCPUE (1.7 versus 0.7) whereas for axis deer this ratio was nearly halved (1.7 versus 3.3), implying that wild boar were more active during the usual time band of hunting sessions while axis deer were substantially less active. The only apparent change among native species was an almost 35% decrease in mean fox abundance as determined by the restricted PhCPUE (Fig. 4).

Table 1 Geometric mean ratios and 95% confidence intervals for every pair of methods and species, and the range of values of the limits of agreement (LoA)

Species	Comparison	Geometric mean ratio [95% CI]	LoA range
Axis deer	SPUE–CPUE	1.75 [1.46–2.10]	3.94
	CPUE–PhCPUE_res	3.37 [2.58–4.40]	13.16
	SPUE–PhCPUE_res	5.89 [4.67–7.45]	18.67
	CPUE–PhCPUE	5.58 [3.80–8.20]	43.27
	SPUE–PhCPUE	9.77 [7.06–13.52]	53.23
Wild boar	SPUE–CPUE	1.17 [0.96–1.43]	2.36
	CPUE–PhCPUE_res	1.99 [1.44–2.78]	11.29
	SPUE–PhCPUE_res	2.42 [1.67–3.52]	17.33
	CPUE–PhCPUE	8.02 [5.25–12.27]	52.33
	SPUE–PhCPUE	9.41 [5.69–15.56]	88.04
Capybara	SPUE–PhCPUE_res	16.68 [11.14–24.97]	140.41
	SPUE–PhCPUE	35.24 [20.55–60.43]	349.46
Foxes	SPUE–PhCPUE_res	8.06 [5.55–11.68]	57.02
	SPUE–PhCPUE	10.91 [6.01–19.79]	138.22
Brown brocket deer	SPUE–PhCPUE_res	2.19 [1.68–2.86]	8.53
	SPUE–PhCPUE	9.06 [6.46–12.73]	36.5

All values were back-transformed to the scale of the index (number of specimens per 100 h of detection effort). All geometric mean ratios significantly differed between methods except for SPUE-CPUE of wild boar

Table 2 Indices of relative abundance (specimens per 100 h of detection effort) for wild boar, axis deer, capybara, foxes and brown brocket deer by sighting, hunting, and unrestricted or restricted camera trapping

Sampling method	Metric (number of sites)	Effort (h)	Wild boar	Axis deer	Capybaras	Foxes	Brown brocket deer
Sighting	SPUE (47)	598	9.7 ± 5.5	26.2 ± 5.7	85.7 ± 17.6	47.3 ± 7.9	1.6 ± 1.2
Hunting	CPUE (47)	600	3.7 ± 1.9	12.9 ± 3.8	–	–	–
Unrestricted camera trapping	PhCPUE (20)	2802	0.7 ± 0.3	3.3 ± 0.8	2.7 ± 0.7	8.9 ± 2.6	0.3 ± 0.2
Restricted camera trapping	PhCPUE_res (20)	570	1.7 ± 0.8	1.7 ± 0.7	2.7 ± 1.1	5.8 ± 2.0	0.3 ± 0.3

Temporal and spatial patterns

Axis deer displayed no definite time trend over the study period and were much more abundant than wild boar, as determined by SPUE and CPUE (Fig. 5a,b). Wild boar numbers displayed a slightly negative time trend, with minima over winter, in coincidence with the reproductive season (Fig. 5b). The relative abundance of capybaras determined by SPUE varied more widely than for all other species, with minimum values by late July (winter), whereas brown brocket deer was the least abundant and least variable of all the investigated species (Fig. 5c).

The relative abundance maps showed different spatial structure between exotic ungulate species (Fig. 6). Axis deer numbers peaked to the east along a broad band close to the Uruguay River (Fig. 6a,b). Wild boar abundance was lowest to the east, especially by CPUE, and peaked in the south-western zone (Fig. 6c,d). The spatial patterns described by SPUE and CPUE for each species were similar. Brown brocket deer had the sparsest distribution (Fig. 7). Axis deer and capybaras were more widespread across the park than the other species, with lower abundance in western and northwestern sections.

Hunters additionally sighted 316 armadillos, 33 felines, 14 lesser grisons, 6 crab-eating raccoons, 6 greater rhea, 1 yacaré caiman (Caiman latirostris) and a few individuals of unidentified species (passeriforms, ducks and raptors).

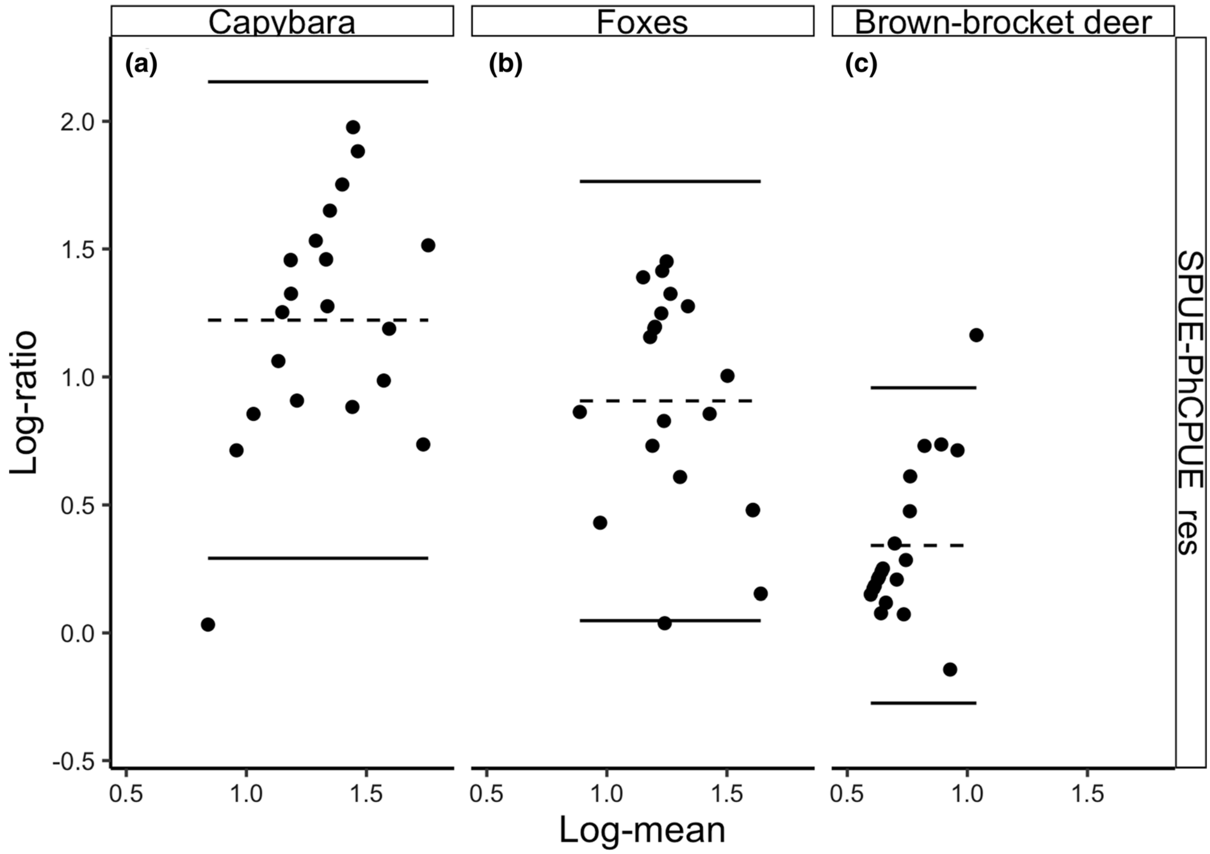

Fig. 4 Bland-Altman plots of the log-ratio between indices of relative abundance SPUE and PhCPUE_res for capybaras (**a**), foxes (**b**) and brown brocket deer (**c**) versus their log-mean per hunting blind. Dashed lines represent the geometric mean ratio ($\bar{d}$) and solid lines the limits of agreement

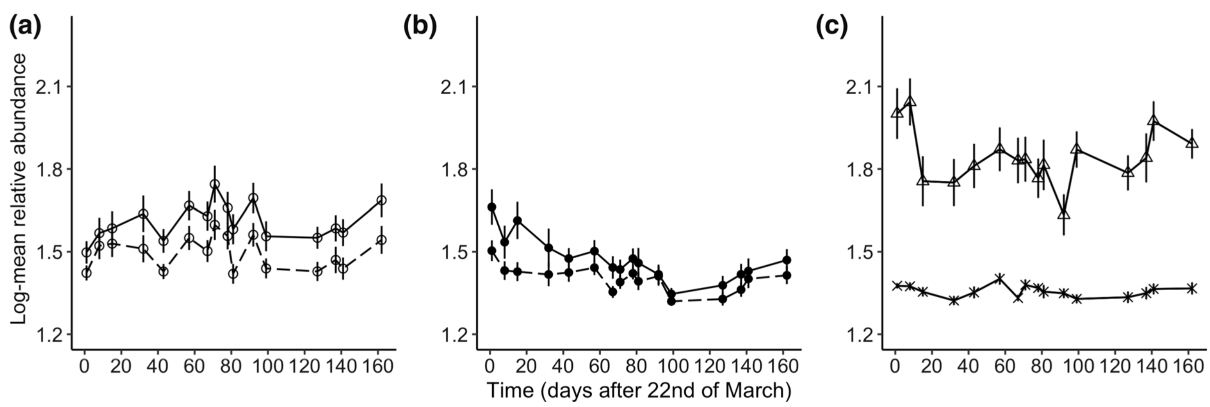

Fig. 5 Temporal trends in the log-mean relative abundance per hunting session for axis deer (**a**), wild boar (**b**), capybaras (triangles) and brown brocket deer (asterisks)(**c**) by sighting (solid lines) and hunting (dashed lines, only for boar and axis deer). Error bars are standard errors

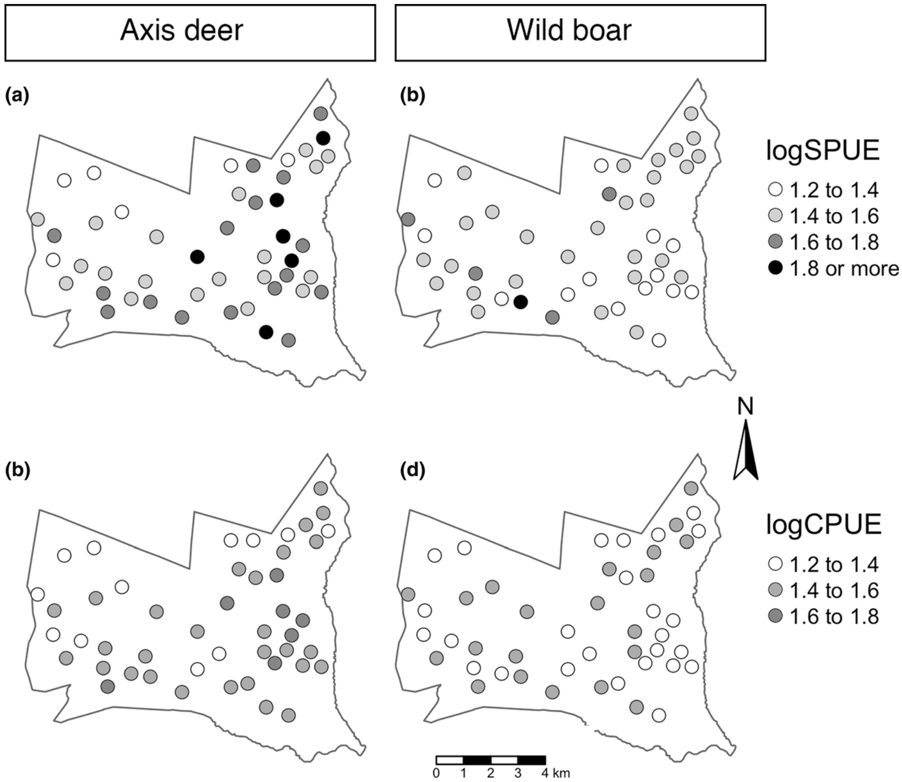

Fig. 6 Spatial distribution of the log-mean relative abundance of axis deer (**a**,**b**) and wild boar (**c**,**d**) as determined by hunter sightings (**a**,**c**) and catch per unit effort (**b**,**d**) in El Palmar National Park. Intervals of index values are non-overlapping and left-closed

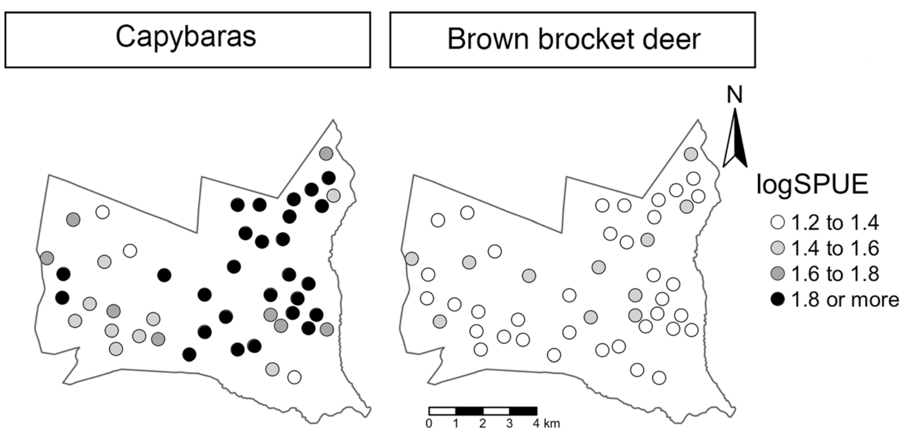

Fig. 7 Spatial distribution of the log-mean relative abundance of capybaras and brown brocket deer by hunter sightings in El Palmar National Park. Intervals of index values are non-overlapping and left-closed

Discussion

Our study shows a high level of agreement between survey methods, especially between sighting and hunting-based indices for exotic ungulates, which also reflected in qualitatively similar temporal and spatial trends for each species. Sighting indices also provided simultaneous, spatially explicit information on several native wildlife species, and the first assessment of their distributions across the park. Of prime relevance is that the hunter-based wildlife monitoring system proved sustainable.

Method agreement

The triple comparison between CPUE, sighting and camera-trap indices suggest that both the SPUE and CPUE are cost-effective, appropriate metrics of axis deer and wild boar abundance for trend analysis and identification of high- and low-density habitats. Our results confirm that axis deer were at much larger abundance than wild boar: the ratio of axis deer to boar was 2.5:1 by standardized CPUE in 2015 (Gürtler et al. 2017, 2018); and here for 2017, 2.7:1 (26.2/9.7 in Table 2) by hunter sightings and 3.5 (12.9/3.7) by crude CPUE. These results further corroborate the relative status and population trends of both invasive ungulates over recent years: sustained, high-level abundance of axis deer and a wild boar population kept at low numbers.

It has often been unclear whether CPUE indices are proportionally related to deer and boar abundance across a broad range of values (implying a constant catchability coefficient), or whether the relationship is affected by hyperstability or hyperdepletion (Walters 2003). Both deviations are well known in the fisheries literature (Harley et al. 2001), but in some cases the CPUE was a valid index of wildlife density for management of deer and game birds (e.g., Lancia et al. 1996; Cattadori et al. 2003; Rist et al. 2010). Recent studies focusing on the relationship between CPUE and surrogate indices of moose or bobcat abundance reported hyperstability and a low level of agreement between metrics (DeCesare et al. 2016, Allen et al. 2020). Hunter self-monitoring information positively correlated with camera-trapping indices in the Republic of Congo, with hunter selectivity apparently being a large source of bias (Marrocoli et al. 2019). These relationships were likely affected by hunter selectivity (for male trophies); poor or unreliable estimates of hunting effort, and how abundance was estimated. The first two issues are virtually irrelevant to the park's management program in which there is no restriction on the number, sex and stage of exotic ungulates harvested (though each party can only claim one trophy a year), and park rangers register individual hunting effort in every session. This overview highlights a key difference between park hunters, who complied with a strict protocol for management of exotic ungulates in a protected area, and hunters elsewhere acting with little supervision under other frameworks, with or without harvest quotas.

The geometric mean ratio in SPUE-CPUE comparisons for exotic ungulates returned an estimate of the percentage of individuals that apparently escaped control efforts: on average, 75% of the sighted axis deer were not removed. This large fraction may be linked to the enhanced vigilance of axis deer derived from long-standing hunting pressure and their gregarious nature (Schaller 1967). As the number of shooters was restricted to one or two per blind, the chances of dispatching several individuals at once were severely limited. Estimates of wild boar abundance by SPUE and CPUE were similar overall, with 90% of the values within the LoA. All values were closely distributed around the mean log-ratio. The two extreme data points that fell outside the LoA were provided by hunting parties with excellent or good participation, who killed and sighted 3 of 16 and none of 3 wild boar, respectively. Comparison of the average number of wild boar sighted (1.5) and removed (0.3) across participating hunting parties suggests that sighting a group of wild boar and harvesting more than one individual on a session occurred rarely. This is consistent with the low-density status of wild boar within park premises after more than a decade of continued management operations (Gürtler et al. 2017).

Both hunter-based indices were in good agreement with camera-trapping indices (either time-restricted or unrestricted), although the LoA between them were an order of magnitude greater than between sighting and hunting indices. Variations in wildlife activity throughout the day may affect the detectability and agreement between survey methods. We examined this hypothesis a posteriori by restricting camera-trapping detections to the same time window as hunter sightings. Controlling for time window effects substantially increased the agreement between methods and returned narrower LoA relative to the unrestricted indices for all the species investigated (Table 1). The differences between time-restricted and unrestricted camera-trapping indices were most evident for wild boar (positive) and axis deer (negative), suggesting a potential mismatch between deer activity patterns and the timing of hunting sessions (Table 2). Whether these patterns are inherent to axis deer or derive from their evasive responses to long-term hunting pressure has important implications for the management program and is the focus of an ongoing study.

The sighting indices of axis deer, wild boar, capybaras and brown brocket deer differed largely between species over space. Axis deer appeared to be spatially aggregated on the eastern section of the park along the Uruguay River, which is heavily invaded by exotic trees and shrubs, whereas wild boar were somehow aggregated in the northwestern and southeastern sections. Both pieces of information are relevant for targeted management efforts directed at areas with a larger concentration of specimens, and need to be further investigated. Capybaras were at high abundance across the park, even at great distances from the Uruguay River, whereas the rare brown brocket deer showed no distinct pattern. This is the first map of the distribution of native species at a parkwide scale. The sighting indices over the six-month period varied little for both deer species and fluctuated more widely for capybaras and wild boar, with minimum values over the winter.

For capybaras, foxes and brown brocket deer, SPUE indices agreed with and substantially exceeded PhCPUE indices, suggesting that the effort necessary to detect an individual of these species by camera trapping was much higher than that needed to sight them from the hunting blinds. Such large differences were unexpected because camera-trapping detection efforts were substantially greater than those implicit in hunter-derived indices (i.e., 5–7 days per camera versus 1–4 sessions per hunting party, leading to average total efforts of 580 versus h, respectively).

The large systematic differences between camera trapping and both hunter-based indices may be related to their characteristic detection areas and microhabitat placement sites nearby hunting blinds. Hunter sightings were made from approximately 4–7 m high and covered a 180° radius, with sections of greater visibility (i.e., shooting lines) where wildlife could be spotted while crossing or feeding on bait. In contrast, camera traps were set 50 cm above the ground, generally within woodlots between shooting lines, and lacked bait. Detection areas are not typically accounted for in relative abundance estimates obtained through camera trapping (e.g., Jenks et al. 2011; Palmer et al. 2018), but they are essential to estimate the density of unmarked animal species (Rowcliffe et al. 2008; Hofmeester et al. 2017; Howe et al. 2017). Differential habitat use between wildlife trails (where cameras were set up) and shooting lines (with bait stations) might favor sighting to camera-trap detections, in an analogous way as the differential outcomes recorded by random and trail-oriented placement of camera traps (Cusack et al. 2015).

Method strengths and limitations

The three survey methods provided spatially explicit measures of abundance and presence-absence. Both SPUE and CPUE can be estimated throughout the year, and virtually require no extra resources or fieldwork provided that the management program continues. However, both methods depend on hunter gear, skills and participation; weather conditions affect visibility and animal activity (e.g., rain, fog). Gear use was heterogeneous among local hunting parties, and the type of night-vision equipment likely affected sightability during a fraction of the hunting session. Camera-trapping indices are hunter-independent and operate under all weather conditions, but they require an initial investment that proves cost-efficient for extended monitoring. On the flip side, camera trapping requires long exposure periods to detect rare species, risking camera loss and battery depletion. Both SPUE and PhCPUE indices may double-count the same individuals since the species we investigated lack distinctive phenotypic traits. The number and individual area covered by each shooting line varied among hunting blinds, and the surrounding vegetation may affect detectability (i.e., greater in grassland than in dense shrubland or forest). Ancillary information on the relevant factors listed above may be used to construct improved, more precise indices of relative abundance.

Monitoring system

The wildlife monitoring system based on hunter sightings functioned consistently during six months, averaging 30 participant groups per session. Hunter participation remained stable, with nearly half of the parties scoring excellent participation and only 11% showing regular participation. The latter is quite lower than the "dabblers" group in other citizen science projects, which can reach up to 80% of participants (August et al. 2020). Hunter participation was approximately similar between park zones, and thus afforded a fair spatial coverage of monitoring effort over a large area.

Sustained commitment of hunters and park personnel to wildlife monitoring and cooperation from park volunteers were essential for logistics and data collection. Even though the hunters' main driver was to harvest wild boar and axis deer, they volunteered to report sightings on native, non-game wildlife and frequently reported on wildlife behavior and other issues (e.g., poaching, bait consumption by native species). Foxes and capybaras were reported to stay nearby throughout a hunting session, and apparently were not disturbed by shooting noise. The shooting lines may have attracted wildlife as feeding hubs with a regular supply of bait, short grass, near-by carrion and eventual water sources. This set of factors may explain the high frequency of wildlife sightings despite the disturbance associated with hunting activity. The ranking of relative abundance among wildlife species was consistent across survey methods except for time-restricted camera-trapping indices. The wildlife monitoring system was sustainable and provided baseline information on selected exotic and native wildlife species in a systematic, cost-effective way.

Future research initiatives may take advantage of the multiple data sources we collected to generate integrated species distribution models that keep track of population trends and can be used to model habitat use and species co-occurrence patterns in protected areas (Warton et al. 2015, Pacifici et al. 2019). Do exotic ungulates (or native versus exotic species) co-occur at random or avoid each other? A thorough spatial analysis of the patterns returned by sighting and hunting-based indices at a parkwide scale deserves special attention. Identification of the underlying drivers of species' spatio-temporal patterns will require additional efforts.

One of the main advantages of hunter sightings is that it provides a standardized metric and protocol for monitoring both invasive and native wildlife species, and therefore allows a direct comparison of data between species. Both aspects contribute to improved monitoring protocols for the impacts and dynamics of biological invasions. Our study highlights how hunters may be engaged both in exotic management and biodiversity conservation programs as a group of interested citizens who regularly spend time in natural areas, have detailed knowledge on local wildlife, and are willing and able to follow protocol. The concerted efforts of park personnel, hunters and researchers contributed to the long-term sustainability of a monitoring system for exotic ungulates and other native wildlife species. Government-sponsored actions at other parks or scales may provide similar high-quality data to improve monitoring efforts by engaging social participation and knowledge. This is of paramount importance for alien invasive species management since reinvasion risk and other ecosystem consequences may occur after control actions (Simberloff et al. 2013).

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Authors' contributions GN, LIRP and REG designed the study, GN, LIRP, AM and AAM acquired the data, GN, LIRP and REG analyzed the data, GN, LIRP and REG drafted the document, GN, LIRP, AM, AAM and REG edited the document.

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Data availability The data that support the findings of this study are available from the corresponding author upon reasonable request.

Code availability No custom codes were developed during the study. All image processing and analysis were conducted using pre-built tools. Please refer to the methods section for further details.

Declarations

Conflict of interest The authors declare no conflict of interest.

Human or animal rights All applicable international, national, and/or institutional guidelines for the care and use of animals were followed. All procedures performed in studies involving animals were in accordance with the ethical standards of the institution or practice at which the studies were conducted.

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