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Comparing local ecological knowledge with camera trap data to study mammal occurrence in anthropogenic landscapes of the Garden Route Biosphere Reserve

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Abstract

Accurate information on wildlife occurrence in anthropogenic landscapes is essential to develop effective conservation measures. Monitoring multiple mammal species in heterogeneous landscapes can be challenging because of the diversity of land-uses and species behaviors. Therefore, different monitoring methods are better adapted to different contexts. We compared data obtained from camera traps with data gathered through an online survey to document the presence of mammal species in mosaic landscapes of the Garden Route Biosphere Reserve in South Africa. The survey detected the same 15 species as the camera traps as well as the additional common duiker. Both methods effectively detected most large mammal species, whereas results were less consistent for smaller species and carnivores. Combining the two methods enabled us to produce more robust estimates of species absence and confirm species presence reported by survey respondents. In less disturbed areas, respondents with good ecological knowledge were effective at monitoring mammal species, while extending the spatial coverage of the study. Bearing in mind the limitations of each method, camera traps and an online survey could complement each other if combined. Together, they can provide a more comprehensive understanding of mammal communities in anthropogenic landscapes, increasing both spatial coverage and the number of species sightings.

KEYWORDS

biosphere reserve, habitat transformation, mammal ecology, mixed methods, participatory research, wildlife conservation, wildlife monitoring

1 | INTRODUCTION

Humans have altered natural ecosystems and habitats through land-use change and landscape transformation (Ellis et al., 2010; Ellis et al., 2021; Kennedy et al., 2020; Williams et al., 2020). Effective conservation science and

practice require a fundamental understanding of the impact of such transformation on biodiversity (Corlett, 2015). Mammals, among other species, are facing numerous human-induced threats, such as habitat alteration, overuse of biological resources, climate change, and the introduction of invasive species (Atwood et al., 2020).

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Their movement has been considerably reduced by anthropogenic modification and fragmentation of natural habitats and an increasing human footprint (Tucker et al., 2018). A lack of connectivity between protected areas often forces animals to use human-modified landscapes for long-distance movement (Brennan et al., 2022). Therefore, the concept of “shared Earth,” developed by Obura et al. (2021), advocates for sharing the 55% of terrestrial land dominated, but not entirely transformed by humans by redesigning landscapes to make them suitable for wildlife. In this context of mitigating the transformation of terrestrial areas by human activities, it is necessary to understand which species could effectively use the diversity of anthropogenic matrices.

Several survey methods can potentially be used to monitor the presence of mammals such as camera traps (CTs) or animal tracking (Kays et al., 2015; Steenweg et al., 2017). Each monitoring method includes their own respective biases (Brittain et al., 2022; Hofmeester et al., 2019). CT surveys are a commonly used, non-invasive method of monitoring wildlife over a long period of time and often across large spatial scales (Cordier et al., 2022; Cove et al., 2021; Pardo et al., 2021; Steenweg et al., 2017). CTs are a systematic method to collect animal observation and allow to record continuously animals activity (Zwerts et al., 2021). The literature on the subject is broad and gives recommendations to set up the most appropriate protocol, thus reducing evitable detection bias (Hofmeester et al., 2019, 2021; Kays et al., 2020). CTs can provide reliable data on the frequency of detection, daily and seasonality activity of a given species (Hofmeester et al., 2019; O'Connell et al., 2011). However, detections often depend on species ecological characteristics, such as home-range size and body mass (Hofmeester et al., 2019). Environmental conditions, such as vegetation type, or seasonality, such as snow cover, can also highly influence detection; for example, detection rates are often lower in closed than open habitats (Hofmeester et al., 2019; Sollmann et al., 2013). Using CTs can become costly in terms of material investment, fieldwork, staff required for servicing, and data processing (Brittain et al., 2022).

Anthropogenic landscapes are characterized by the presence of people who could intentionally or unintentionally observe and monitor wildlife around them. Drawing on this potential, large-scale citizen science projects are becoming increasingly prolific worldwide (Dickinson et al., 2012). They were mainly developed in North America and Australia, but few were done in Africa (Stern & Humphries, 2022). Local ecological knowledge (LEK) can be mobilized using several methods, such as interviews, questionnaires, participatory mapping or diaries (Jones et al., 2008; Pédarros

et al., 2020; Stern & Humphries, 2022; White et al., 2005) and can interview different knowledge holders such as hunters or trappers, academic experts, or community members (Stern & Humphries, 2022). These methods allow one to collect a large amount of data at low cost, and often with broad coverage, to answer ecological questions (Dickinson et al., 2012). LEK can also provide information on human practices, values, and perceptions (McKinley et al., 2017; Stern & Humphries, 2022). LEK has particularly been shown to be a cost-effective tool to study the presence of mammals and realize quick biodiversity inventories (Burt et al., 2021; Turvey et al., 2013). Furthermore, involving local communities and residents in research protocols could help to promote a reconnection with their surrounding ecological systems, while allowing the researchers, and society at large, to benefit from their local knowledge (Obura et al., 2021). By improving trust in the science process, such methods could greatly benefit the efficiency and sustainability of conservation measures (McInturff et al., 2021; Stern & Humphries, 2022).

The detection of a species depends on the observer, and can vary with season and species behavior (Boakes et al., 2010; Msoffe et al., 2007; Pearse et al., 2015), and incorrect information such as mis-identifications or scale mismatches are difficult to detect (McKelvey et al., 2008; Stern & Humphries, 2022). Furthermore, the coverage of the study area is often not systematic (Stern & Humphries, 2022). Humans can detect mammals species using a range of direct observations and indirect indicators, such as tracks, scats, or hairs, which can remain after a given species has passed through a landscape, that can be used as an index to monitor mammals' presence (Zwerts et al., 2021). The reliability of LEK has been questioned (Caruso et al., 2017), despite many instances of well-designed LEK-based surveys and analyses providing trustworthy information (Jones et al., 2008; Petracca & Frair, 2017). Using LEK requires the development of standardize methods to measure and limit uncertainties as well as observers bias (Stern & Humphries, 2022).

Few studies have compared the detection outcomes of CTs with those of LEK in human-dominated landscapes (Brittain et al., 2022; Caruso et al., 2017; Schaller et al., 2012; Zwerts et al., 2021). The best choice of method depends on the target species, population metrics to be evaluated, and resources available (Zwerts et al., 2021). Therefore, it is important to conduct site-specific studies to understand which of the two methods will be the most effective (Brittain et al., 2022; Swan et al., 2014) or how to weave them. As each method collects different types of data, weaving them could provide complementary data sets, allowing one to better document and understand species occurrence (i.e., presence

and frequency of detection) in a landscape (Brittain et al., 2022; Schaller et al., 2012; Service et al., 2014). Comparing the efficiency of both methods to spatially document mammals' community composition in human-dominated landscapes will help managers to set up adequate protocols and design conservation plans for mammals monitoring.

In this study, we used two independent data sets to compare CT and LEK consistency in detecting wild mammal species in the mosaic landscapes of the Garden Route Biosphere Reserve (GRBR), located in South Africa, and tested hypotheses which could explain the discrepancies. Medium to large mammals are useful study models, as they are relatively visible and represent a diversity of sizes, diets, and ecological requirements (Santini et al., 2019). The medium to large mammal community of the GRBR is a fair representation of this diversity and comprises mammal species ranging from rare (e.g., leopard, *Panthera pardus*) to common (e.g., Chacma baboon, *Papio ursinus*) (Baard et al., 2015). We examined a set of variables, which we suspected would result in biased detection rates and explain potential discrepancies between the two methods. We included variables to account for spatial heterogeneity (human modification index [HMI], describing the level of human landscape modification), local residents' heterogeneity (ecological knowledge score of respondents) and mammal species heterogeneity by considering several traits (their diet, averaged body mass, and their daily activity pattern [diurnality]). We hypothesized that the two methods would be consistent in large mammal detection, whereas CTs would be more reliable in detecting nocturnal species and those with small body mass. We further hypothesized that LEK would be more effective in highly modified landscapes, with a greater human presence, and would provide more data for rare species, especially if respondents possessed good ecological knowledge, whereas CTs would be more efficient in detecting species in less disturbed areas,

where human observers would by definition be less present.

2 | MATERIALS AND METHODS

2.1 | Study area

The study took place in the GRBR, located in the Western Cape Province, on the southeastern coast of South Africa. Nestled between the Indian Ocean and Outeniqua Mountains (Figure 1), the region is highly attractive for local, regional and international visitors. The number of residents in the area has increased dramatically in recent years, and the population is expected to continue growing in the future (Guerbois et al., 2019; Western Cape Government, 2019). Anthropogenically transformed landscapes in the area are associated with multiple land-uses, such as forest plantations, crops, dairy farming, urban and peri-urban development, and a vast road network (Baard et al., 2015; SANParks, 2020). The Garden Route National Park (GRNP), an unfenced protected area established in 2009 and managed by South African National Parks (SANParks), is mostly accessible for visitors, who can enjoy outdoor activities, such as canyoning, mountain biking, or hiking. However, natural resource harvesting is either prohibited or strictly regulated and the GRBR is home of a diversity of medium to large mammal species (Table 1).

2.2 | Data collection

2.2.1 | Camera traps

We took advantage of a field set-up composed of a grid of 74 CTs (Figure 1), where 30 CTs were located in the

FIGURE 1 Study area in the Garden Route Biosphere Reserve represented by the green star, South Africa. The number of responses to the survey (purple gradient) and location of the camera traps (black dots) have been superimposed on the same grid of 5 km² cells. This information and the Garden Route National Park (green outline) are overlaid on a human modification index map.

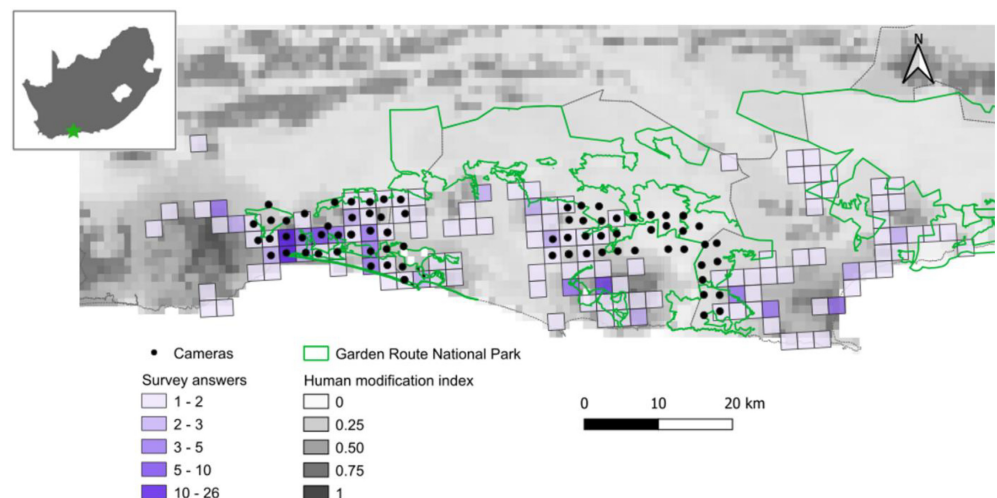


TABLE 1 Summary of the 16 species detected on 35 common cells using both camera traps and local ecological knowledge. The average body mass and diurnality were extracted from PanTHERIA database (Jones et al., 2009).

Common name	Scientific name	Diet	Body mass (kg)	Diurnality
South African large-spotted genet	<i>Genetta tigrina</i>	Strictly carnivore	2	Nocturnal
Caracal	<i>Caracal caracal</i>	Strictly carnivore	13.7	Diurnal
Leopard	<i>Panthera pardus</i>	Strictly carnivore	30	Diurnal
Cape grysbok	<i>Raphicerus melanotis</i>	Herbivore	10.2	Nocturnal
Blue duiker	<i>Philantomba monticola</i>	Herbivore	4.9	Diurnal
Common duiker	<i>Sylvicapra grimmia</i>	Herbivore	15.6	Nocturnal
Cape porcupine	<i>Hystrix africaeaustralis</i>	Herbivore	14.9	Nocturnal
Bushbuck	<i>Tragelaphus scriptus</i>	Herbivore	43.2	Diurnal
Cape gray mongoose	<i>Herpestes pulverulentus</i>	Flexible carnivore	0.8	Diurnal
Large gray mongoose	<i>Herpestes ichneumon</i>	Flexible carnivore	3.0	Diurnal
Water mongoose	<i>Atilax paludinosus</i>	Flexible carnivore	3.3	Nocturnal
Honey badger	<i>Mellivora capensis</i>	Flexible carnivore	8.5	Diurnal
African clawless otter	<i>Aonyx capensis</i>	Flexible carnivore	19	Nocturnal
Vervet monkey	<i>Cercopithecus pygerythrus</i>	Omnivore	5	Diurnal
Chacma baboon	<i>Papio ursinus ursinus</i>	Omnivore	17.7	Diurnal
Bushpig	<i>Potamochoerus larvatus</i>	Omnivore	48.8	Nocturnal

GRNP and 44 in human-dominated areas. Following the standardized protocol adopted in the Snapshot Safari project (Pardo et al., 2021), CTs were positioned as close as possible to the center of a regular 2.23×2.23 km grid cell. They were set up on game trail, at a height of 50 cm above the ground and mainly on trees, to detect medium to large mammals. We used two models of CTs: 33 Bushnell Trophy Cam HD Aggressors and 41 SPYPOINT Force Dark, which were set up similarly. When the CT detected a movement, it was programmed to take a series of three images within 1–5 s of each other. CTs were serviced every 2 to 3 months to change the batteries and SD cards. The CT monitoring period ran from February 2, 2021 to May 20, 2022 during which time four CTs were stolen.

2.2.2 | Online survey

We conducted an online survey to assess the presence of wild mammals and describe the characteristics of private properties in the GRBR (Bernard et al., 2024). This survey was designed to gain more information on species occurrence in human-dominated landscapes (Bernard et al., 2024), and targeted landowners in the GRBR. After having been piloted several times and translated, the survey was sent using mailing lists obtained from conservation agencies, such as Cape Nature, SANParks, the Wildlife and Environment Society of South Africa, and

local conservancies, as well as WhatsApp and Facebook groups. We also advertised the online survey in local newspapers (*George Herald* and *The Edge*). The survey was accessible online from November 2021 to March 2022 in both English and Afrikaans. We sent several follow-up communiques to increase our sample size. This research design was approved by Nelson Mandela University ethic committee (H20-SCI-SRU-002).

Species presence was assessed via the frequency at which respondents identified that a species was present on their properties using direct observations or indirect presence cues, such as tracks or scats. Respondents could report on the presence of multiple different species using the following sighting frequencies: not at all, not anymore, rarely (once a year), occasionally (once a month), frequently (once a week), and very frequently (every day). Respondents could also select general categories (e.g., mongooses and genets) when they could not differentiate between species. Participants were also asked to indicate their residence on the same grid of 5 km^2 cells used for the CTs; no exact GPS location had to be shared, thus preserving the anonymity of the participants (Figure 1). The ecological knowledge of the respondents was tested via four questions in the survey. Two were based on identifying species using a photo: an herbivore (blue duiker, *Philantomba monticola*) and a flexible carnivore (Cape mongoose, *Herpestes pulverulentus*), respectively. The third required respondents to identify the track of a carnivore (caracal, *Caracal caracal*), and

the fourth to identify the scat of an herbivore (Cape porcupine, *Hystrix africaeaustralis*). For each question, respondents were given a choice of three species.

2.3 | Data processing and analysis

Both methods of data collection were conducted in the same year to avoid variations due to temporal changes. We received 247 complete answers and gathered 12,002 independent photos of wild mammal species. Thirty-five cells contained data using both methods, that is, had a CT and at least one survey response. Only data sets from these 35 cells, corresponding with 35 CTs and 95 survey responses, were considered in the following analyses.

2.3.1 | Estimating wildlife presence

Camera traps

The CT photographs were manually tagged, first using digiKam (digiKam Team, 2001), then with the online software TrapTagger (Osner, 2022). Based on their content, all photographs were tagged using the following categories: mammals, birds, reptiles, humans, human vehicles, and domestic animals. For this analysis, we focused on mammal species with a body mass greater than 500 g. Empty photographs and those of birds, rodents, humans, human activities, and domestic animals were not considered. When similar species were detected, we filtered the data to keep only one photograph taken every 30 min, thereby ensuring that images represented independent events. To compare the frequency of occurrence recorded using CTs and the survey, we built a CT detection matrix to indicate the monthly and weekly presence of each species on each cell. When there was no detection by CTs, we classified the frequency of occurrence as “never”. When species were detected in less and more than three different months, we classified their occurrence frequency as “rare” and “occasional”, respectively. Species detected in more than half of the weeks surveyed were classified as “frequent”. We did not classify any species as “very frequent”, as none was photographed every day.

Online survey

We transformed the occurrence frequency of each species to obtain a presence/absence variable. When a species occurred very frequently (every day) or frequently (once a week), we assigned the variable a value of 3 (the two categories were grouped to reflect the lack of daily CT sightings), occasionally (once a month) a value of 2, and rarely (once a year) a value of 1. When a

respondent did not observe a species or reported that a species no longer occurred on their property, we assigned the variable a value of 0 (absence). As a given cell could include observations from several respondents, we averaged the occurrence frequency of each species per cell. If the average score was equal to or above 1, we considered the species present and assigned the cell a value of 1. Because mongooses and genets were difficult to identify at a species level, respondents had the option to tick the general categories “mongooses” (including the Cape gray mongoose, large gray mongoose: *Herpestes ichneumon*, and water mongoose: *Atilax paludinosus*) and “genets” (including the large-spotted genet: *Genetta tigrina* and small-spotted genet: *Genetta genetta*). Only the large-spotted genet has been recently recorded in the area and detected using CTs, although both species are listed as potentially occurring; therefore, we assumed that respondents who selected “genets” were referring to the large-spotted genet (Baard et al., 2015). In subsequent analyses, to describe the presence of mongooses, we only considered data from respondents who specified the species occurring on their properties and not those who used the general category.

2.3.2 | Candidate variables to assess detection bias

We tested a set of variables, describing characteristics of species, cells, and respondents, that we hypothesized could explain potential discrepancies in detection between the two methods. We first selected species traits that we anticipated would affect the probability of detection of mammal species in anthropogenic environments (Figure S1). As diet flexibility could influence the ability of a species to adapt to a new environment (Hulme-Beaman et al., 2016), we used the EltonTraits database (Wilman et al., 2014) to extract data on the diet of each studied species and classified them as either strict carnivores (diets comprising $\geq 70\%$ of meat), flexible herbivores, or omnivores. Further, we used the PanTHERIA database to extract data on average adult body mass and diurnality/nocturnality (Jones et al., 2009), as it could influence species detectability. To account for spatial heterogeneity at the scale of the cell and assess the level of human modification of the landscape covered by the survey, we calculated the mean HMI (1 km resolution) of each 2.23×2.23 km cell. The values ranged from 0 to 1. A value close to 0 indicated no human modification, whereas a value close to 1 indicated substantial modification of terrestrial land (Kennedy et al., 2020). Finally, to assess heterogeneity between respondents at the scale of the property, we assigned each respondent in a cell an

ecological knowledge score by averaging the scores from the species identification questions. We thus obtained a variable between 0 (all answers incorrect) and 100% (all answers correct). The variables were scaled in the subsequent analysis to ensure comparability.

2.3.3 | Data analysis

In each cell, we compared the averaged survey responses with the information obtained from the CT. When a species was detected using both methods in a given cell, we coded it as “presence” (P); when a species was not detected using either of the two methods, we coded it as “absence” (A); when a species was detected only by the CTs, we coded it as “presence CT” (PC); and when a species was detected only in the survey, we coded it as “presence survey” (PS; Figure S3). Given that we were interested in the discrepancy between the two methods, we grouped P and A under “similar detection” (S). We then compared the three categories (S, PC, and PS) by running a multinomial logistic regression (“mblogit” function, “mclout” package; Elff, 2022). We tested the five variables described in Section 2.3.2 that could potentially explain the discrepancies between the two methods as fixed effects in the same model: three numerical variables—HMI value per cell, average ecological knowledge score of respondents per cell, and the average body mass of each species—and two categorical variables—diet (flexible carnivore, strict carnivore, omnivore or herbivore), and diurnality (diurnality or nocturnality) of each species—. The cell was added as a random effect to account for potential spatial correlation. Finally, we calculated the marginal predictions (Elff, 2021). All analyses were performed using R (R Core Team, 2020).

3 | RESULTS

3.1 | Description of the raw data

The two methods detected a total of 16 species, with body mass ranging from 0.8 to 52 kg (Tables 1 and S1). Six species were strictly nocturnal. Only the common duiker was not detected by CTs and was excluded from the analysis (Tables 1 and S1). The survey detected all species, including those rarely detected by CTs (African clawless otter in 23% of the cell, Cape grysbok in 34%), on more sites than the CT method (Table S1). In contrast, large-spotted genets (97% of the cells according to CTs and 43% according to the survey) and water mongooses were detected on more sites by CTs than the survey (respectively, 49 and 17%; Table S1). The two methods were

consistent in the number of sites exhibiting bushbuck (97%) and bushpig (86% for the survey and 83% for the CTs) presence (Table S1). The number of respondents per cell ranged from 1 to 24 (Figures 1, S2 and S3). Their ecological knowledge score ranged from 0 to 100%, with an average score of 53% (Figures S1 and S3). The cells covered a gradient of human landscape modification, with an HMI value ranging from 0.07 to 0.69 (Figure S1).

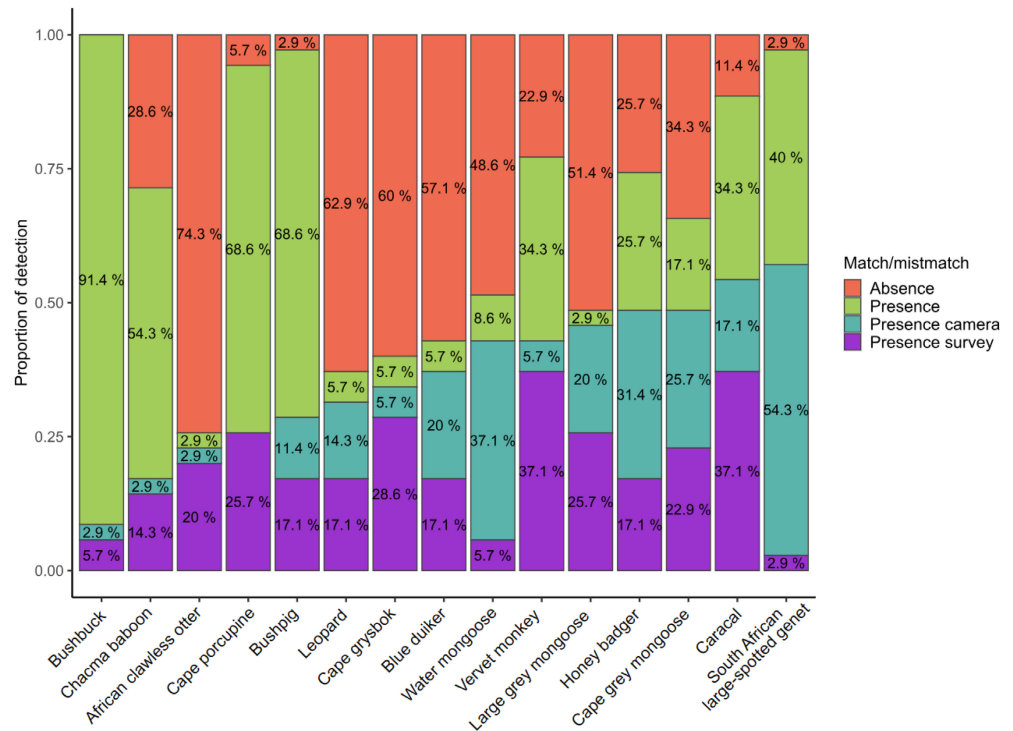
Although the consistency between the two detection methods depended on the species, 13 out of 15 species exhibited a detection overlap above 50%, and five (bushbuck, chacma baboon, African clawless otter, Cape porcupine, and bushpig) above 70% (Figure 2). Both methods detected common species (e.g., bushbuck, chacma baboon, Cape porcupine, and bushpig) as well as rare species (e.g., Cape grysbok, leopard, and African clawless otter) at similar rates (Figure 2). The two differed mostly in detecting the large-spotted genet, caracal, Cape gray and large gray mongoose, honey badger, and vervet monkey (Figure 2). Large-spotted genets, honey badgers, and water mongooses were more likely to be detected by CTs, whereas caracals and vervet monkeys were more likely to be detected by respondents (Figure 2).

Beyond the distribution of detections, the actual frequencies of detection were consistent between the two methods for common species, such as the bushbuck and chacma baboon, as well as rare species, such as the Cape grysbok and honey badger (Figure 3). The survey respondents detected carnivores more frequently than CTs (Figure 3). The same held true for the caracal, vervet monkey, and Cape Grysbok (Figure 3). In contrast, CTs detected large-spotted genets, blue duikers, and honey badgers at higher frequencies than respondents (Figure 3).

3.2 | Multinomial regression results

The probability of both methods detecting the presence of a species in a given cell was greater than that of mismatched detection, suggesting that the methods were consistent with each other most of the time (Figures 3, 4 and Table S2). Both methods were less effective at detecting smaller species than larger ones (for PC vs. P: $\beta = -0.58$ (95% CI $[-0.92$ to $-0.23]$), p -value < 0.001 ; for PS vs. P: $\beta = -0.36$ $[-0.63$ – $0.09]$, p -value = 0.008; Figure 4, Table 2). The most inconsistency between the two methods was observed in the detection of strict carnivores: the observed decrease in match between methods appears significantly influenced by the larger detection probability of these species by CTs (for PC vs. P: $\beta = 0.97$ $[0.16$ – $1.7]$, p -value = 0.02; Figure 4, Table 2). Most of the

FIGURE 2 Proportion of detections per species in 35 cells with both camera trap and survey respondent data across four match/mismatch categories: no detection using either method (absence), detection using both methods (presence), detection only by camera traps (presence camera), and detection only by detection only in the survey (presence survey).



instances of species detected exclusively by survey respondents occurred in cells where residents had higher ecological knowledge (for PC vs. PS: $\beta = -0.45$ [-0.82 to -0.07], p -value = 0.021; Figure 4, Table 2). The probability of detecting a species using the survey was lower than that using CTs in more modified contexts, with a higher HMI value (for PC vs. PS: $\beta = 0.52$ [0.13 – 0.91], p -value = 0.009; Figure 4, Table 2). The diurnality of species did not affect the probability of matching detection rates between the two methods (p -values >0.05 ; Figure 4, Table 2).

4 | DISCUSSION

Reliable information on mammal use of anthropogenic landscapes is necessary to develop appropriate conservation measures. In this study, CTs and LEK methods, respectively, detected 15 and 16 mammal species that occur in the landscape mosaic of the GRBR. As expected, the two methods were consistent in their detection of large species. However, in the case of smaller carnivores, such as mongooses and genets, LEK provided less reliable results than CTs as many respondents failed to identify the species and indicated the general categories only. LEK are often subjective as dependent on the experience of the respondent but also on their capacity to correctly identify and remember a species (Stern & Humphries, 2022; Zwerts et al., 2021). It cannot be verified whether the respondent misidentified a species and

confounded it with another (McKelvey et al., 2008). Furthermore, humans can have a detection bias toward certain species because of their own knowledge (Boakes et al., 2010; Martínez-Martí et al., 2016; Msoffe et al., 2007), and this study provides numerical evidence of the quality of the data that is possible to obtain by deploying a random and not selective online survey. We got answers of residents which had a strong interest in conservation but a relatively low level of ecological knowledge. The online survey allowed to describe the mammals' community composition of the GRBR with a relatively high level of confidence in comparison with a CTs grid. Participants with higher ecological knowledge score detected more species than other respondents as well as CTs. Therefore, to obtain reliable observational data, one must ensure that participating observers have sufficient knowledge of the relevant species (Camino et al., 2020). Scientist or managers can define a threshold required to assess which informant is considered trustworthy (e.g., only consider the responses from ecological knowledge score above 75%). This can also be adjusted depending on the objective of the study. They could also decide to focus on the interview of local ecological experts. Further, to reduce detection bias toward certain species, we only considered species detected by at least 15% of the respondents, which would surely reduce the misidentification bias by removing outliers' responses. In wide-ranging surveys, without any respondent selection criteria, it may be useful to apply this kind of threshold of acceptability based on the frequency of species citation

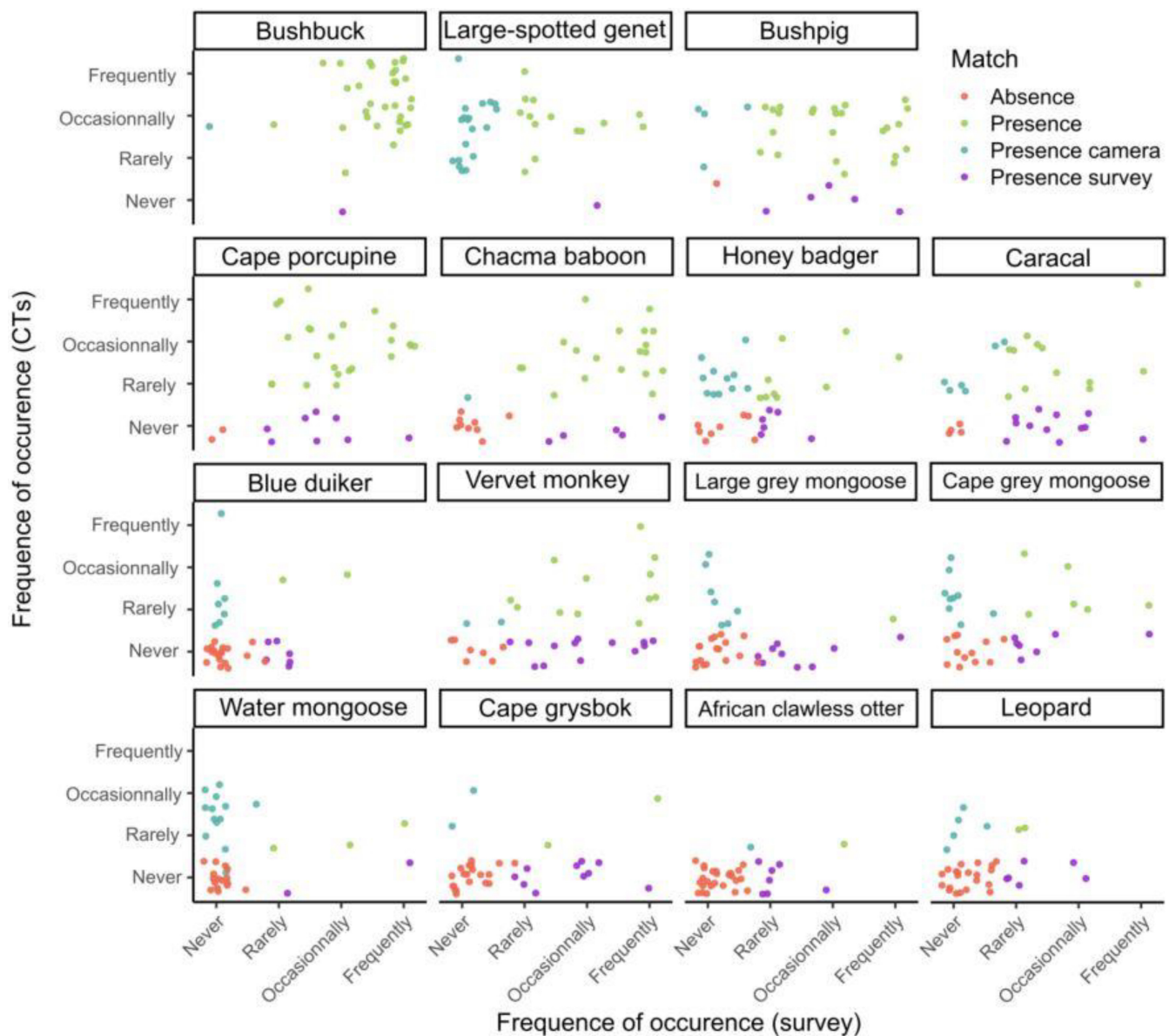


FIGURE 3 Detection frequency per species in 35 cells with both camera trap and survey respondent data across four categories representing method consistency: no detection using either method (absence), detection using both methods (presence), detection only by camera traps (presence camera), and detection only by detection only in the survey (presence survey).

(Burt et al., 2021). Hosting a preliminary course or educational program on local ecology could potentially increase the proportion of reliable respondents in a given area. In contrast, using CTs data, it is possible to verify species identification (Hofmeester et al., 2019). Although CTs also have detection biases, these can often be reduced in the field, for instance by choosing an adequate CTs placement and spacing or can be corrected via commonly used analysis methods, such as occupancy models (Hofmeester et al., 2019; MacKenzie & Bailey, 2004). Occupancy models use a temporal replicate of detection and non-detection of a species at a given site to produce a corrected estimate of the probability of occupancy

(i.e., that at least one individual of a species occupies a given site), and the probability of detection p (i.e., the probability of detecting the species at a site, given that the site is occupied; MacKenzie et al., 2017). They use hierarchical models meaning that occupancy and detection processes are modeled separately enabling to correct for detection bias by using appropriate covariates (Hofmeester et al., 2019; MacKenzie et al., 2017).

Contrary to our hypotheses, in highly modified environments, respondents were less effective than CTs at detecting mammals. A potential explanation for this could be that, in these environments, mammals were restricted to specific corridors, often located in deep

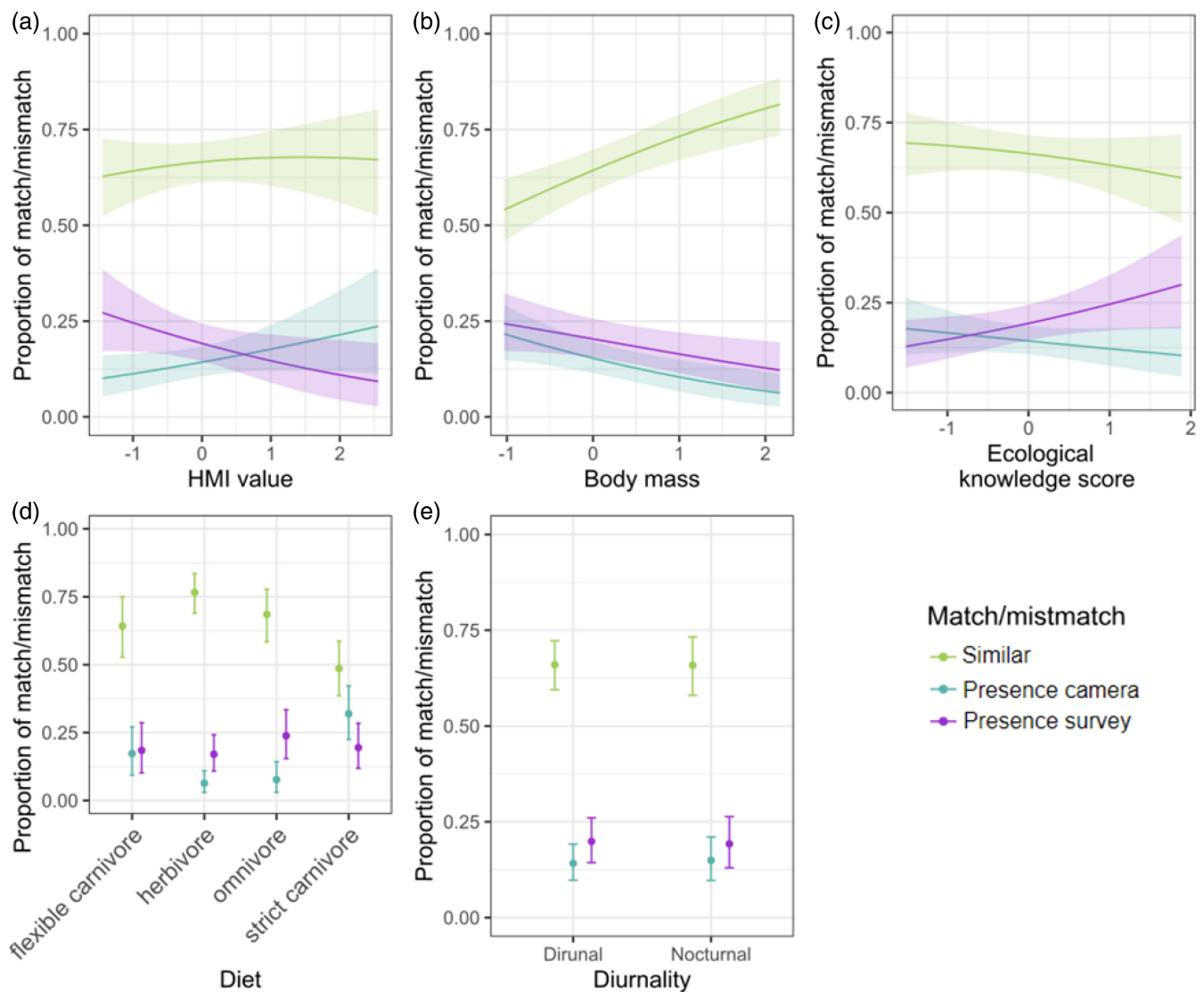


FIGURE 4 Predicted probability of a match and mismatch between camera trap and survey species detection in relation to five explanatory variables: (a) human modification index (HMI) value, (b) body mass, (c) ecological knowledge score, (d) diet, and (e) diurnality. A match represents the detection of species presence/absence by both methods (similar) and mismatch the detection of species presence by only one method (presence camera or presence survey).

valleys, with little human presence. Mammals were thus more likely to use trails rigged with CTs compared to areas where they were less constrained by human activity. Another explanation could be that in such disturbed areas, mammals could shift their peak of activity, becoming more nocturnal and thus more difficult to detect (Frey et al., 2020; Gaynor et al., 2018). They could also have a direct response to avoid encountering humans, which might reduce observers chances of a direct observation (Gilbert et al., 2022). CTs could then be a more appropriate method in such areas, as they probably have less influence on animals behavior: they often become habituated to their presence (Caravaggi et al., 2020). However, high human disturbance also implies more activities and presence and therefore more chances of CT

theft. In less anthropogenically disturbed areas, respondents with more ecological knowledge could in fact be more effective at monitoring mammal species, as shown in other contexts as well (Burt et al., 2021).

We did not observe any effect of species diurnality or nocturnality on the consistency between the two methods, probably because respondents could detect species using indirect presence such as tracks or scats (Bernard et al., 2024; Zwerts et al., 2021). Bernard et al. (2024) have shown that the percentage of sightings and indirect cues of presence varied between species; most respondents detected bushbuck via sightings (92%), whereas the blue duiker (39% of sightings) and leopard (33% of sightings) were mostly detected via indirect signs.

TABLE 2 Summary statistics of the multinomial logistic regression.

Predictors	β estimates	2.5% lower CI	97.5% upper CI	<i>p</i> -value
<i>PC vs. P</i>				
Intercept	-1.39	-2.14	-0.64	0.0003
HMI	0.23	-0.12	0.58	0.192
Diet: herbivore	-1.24	-2.17	-0.30	0.01
Diet: omnivore	-0.93	-1.98	0.12	0.083
Diet: strict carnivore	0.97	0.16	1.79	0.02
Body mass	-0.58	-0.92	-0.23	0.00096
Diurnality: nocturnal	0.07	-0.58	0.71	0.842
Ecological knowledge score	-0.14	-0.51	0.24	0.478
<i>PS vs. P</i>				
Intercept	-1.29	-2.01	-0.57	0.0005
HMI	-0.30	-0.66	0.06	0.105
Diet: herbivore	-0.27	-1.05	0.52	0.508
Diet: omnivore	0.22	-0.60	1.03	0.606
Diet: strict carnivore	0.34	-0.51	1.19	0.427
Body mass	-0.36	-0.63	-0.09	0.008
Diurnality: nocturnal	-0.03	-0.58	0.51	0.907
Ecological knowledge score	0.31	-0.01	0.63	0.060
<i>PC vs. PS^a</i>				
Intercept	-0.11	-1.00	0.78	0.810
HMI	0.52	0.13	0.91	0.009
Diet: herbivore	-0.95	-2.06	0.16	0.093
Diet: omnivore	-1.12	-2.31	0.09	0.069
Diet: strict carnivore	0.60	-0.40	1.61	0.238
Body mass	-0.21	-0.61	0.18	0.291
Diurnality: nocturnal	0.09	-0.66	0.84	0.815
Ecological knowledge score	-0.45	-0.82	-0.06	0.021

Note: Significant *p*-values are written in bold italic.

Abbreviation: CI, confidence interval.

^aAn additional multinomial logistic regression with PS as reference was performed to obtain the estimates for all the categorical values (P, PC, PS).

Discrepancies between the two methods could also be explained by the large scale of the study, which is often used in CT surveys to ensure spatial independence between sites (Pardo et al., 2021), but possibly inappropriate for monitoring species with specific localized microhabitats or with very small home ranges or territories. In cases where respondents detected a species that had not been detected by a CT, either the (1) species had not passed in front of the CT, (2) CT may have been placed in a location that the species does not use, or (3) species was not there (Hofmeester et al., 2019). Furthermore, using CTs following a standardized protocol may not be appropriate for a wide range of different species behaviors (Hofmeester et al., 2019). Potentially further explaining detection differences, humans move across landscapes, whereas CTs are fixed (Zwerts et al., 2021).

Humans might be biased toward the areas they frequent most, but their likelihood of encountering a species increases with their mobility in the landscape. Most importantly, because many people use anthropogenic landscapes, there is a high chance of species detection, especially on private properties (Pédarros et al., 2020). In our case, using a LEK online survey enabled us to cover a much larger area at a lower cost than possible with CTs (~150\$USD per camera), which have a smaller detection area (Brittain et al., 2022). Nevertheless, CTs are a more systematic method, that can be repeated and thus provides more reproducible measures (Kays et al., 2020). Echoing previous studies, we recommend combining several methods to obtain a more comprehensive understanding of species distributions and activities across large landscapes (Anadón et al., 2010; Burt et al., 2021).

It is not possible to conclude whether one method is generally better than the other. Like those of other studies, our results suggest that the reliability of the two methods depends on the species (Brittain et al., 2022; Can & Togan, 2009; Caruso et al., 2017; Zwerts et al., 2021). Inconsistencies between the two methods suggested that several species, such as the caracal, blue and common duiker, and Cape grysbok, would require specific monitoring programs. As suggested by other studies, the participatory approach produced more data for rare species (e.g. African clawless otter) than CTs, which had a lower probability of detection (Brittain et al., 2022; Burt et al., 2021). Wildlife monitoring requires a time investment and protocol adapted to specific species behavior and ecology (Hofmeester et al., 2019; Sollmann et al., 2013). Although CTs do not necessarily produce the best results for rare and cryptic species, they are useful in monitoring areas where humans are not present, as in the case of the blue duiker. Weaving monitoring methods could therefore be a promising approach to tracking these species (Brittain et al., 2022; McKelvey et al., 2008; Zwerts et al., 2021), enabling scientists to produce more robust estimates of species absence and confirm the presence of species detected by respondents (Bernard et al., n.d.).

The design, requirements, and costs (material, staff, and fieldwork) of the two methods differed meaningfully. The survey required more time in the conception phase, whereas CTs required a lot of time for data collection and processing (Zwerts et al., 2021). We piloted the survey with 20 landowners to reach a final version that was well-adapted to the context of GRBR and relevant to most participants. LEK protocols, especially online protocols, need to be piloted to ensure that they produce reliable data. The survey data were collected over 4 months, whereas CT monitoring ran for 1 year. Despite this difference, both methods required time for data cleaning and processing. For research projects with little financial support or staff, the survey is a great tool for a quick and reliable assessment the presence of mammals in a given landscape (Brittain et al., 2022). This method can also produce data on the resources used by animals on properties, thus contributing to a better understanding of coexistence between humans and wildlife (Bernard et al., 2024). Furthermore, the two methods did not provide the same level of information on ecological and anthropogenic variables (Bernard et al., n.d.; Bernard et al., 2024). For example, the survey produced more precise data on human activities than what could be gathered based on the proximity of the CTs (Bernard et al., n.d.; Bernard et al., 2024). However, CTs can more accurately capture the immediate disturbances associated with human activities, as well as the presence of cats or dogs (domestic and feral). This could

provide valuable information on their impact on wildlife (Parsons et al., 2016; Seymour et al., 2020). Overall, the use of both methods enabled us to cover larger and more diverse areas; survey respondents mostly lived in peri-urban areas, whereas CTs were mostly set up in tributaries and on farms. Therefore, human observers can represent an asset for extending the spatial coverage of mammal monitoring programs. We would recommend combining a grid of CTs and observers at a 5 km² scale, as it minimizes the risk of missing species, while extending the landscape diversity and spatial coverage of the study, both within and between grid cells. However, this requires that the methods be designed in advance to ensure compatible protocols and comparable data (Petracca & Frair, 2017). These protocols must be tailored to the target species, research objectives, and resources available for the project (Camino et al., 2020; Zwerts et al., 2021).

Transdisciplinary approaches have the potential to improve the relevance of research for conservation (Zwerts et al., 2021). Both methods have proved valuable to do a spatial biodiversity inventory of the medium-to large mammal in the GRBR. Such protocols could help monitor spatially explicit temporal changes in mammals' species distribution but also on the modification of the community composition. Quickly detecting changes is critical in a rapidly changing world to enable managers to implement targeted conservation measures to reduce biodiversity loss (Brittain et al., 2020). Further, combining methods increase the sample size and spatial coverage which could help researchers to fundamental ecological questions such as regarding the selection of specific life-history traits in human modified-habitats (Bernard et al., n.d.). Finally, combining CTs with participatory surveys allows one to target key conservation stakeholders and initiate discussions about conservation with them (Camino et al., 2020; Devictor et al., 2010). This is an important first step in co-designing conservation measures to build a shared landscape between people and wildlife (Obura et al., 2021).

AUTHOR CONTRIBUTIONS

H.F., C.G., and A.B. conceived the original idea. A.B. collected the data, performed the analyses, and wrote the initial draft. All the authors contributed to the reviewing and editing of the final version. H.F., C.G., and J.V. supervised this research.

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CONFLICT OF INTEREST STATEMENT

The authors declare no conflict of interest.

DATA AVAILABILITY STATEMENT

The data that support the findings of this study are available from the corresponding author, A.B., upon reasonable request.

ETHICS STATEMENT

The research was approved by Nelson Mandela University Human Research Ethic Committee (H20-SCI-SRU-002), SANParks (BERN-A/2020-008), and CapeNature (CN44-87-16198).

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SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

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