ETHNOBIOLOGY AND CONSERVATION

RESEARCH ARTICLE

Ethnobiology and Conservation, 12:19 (25 October 2023)
doi:10.15451/ec2023-09-12.19-1-14
ISSN 2238-4782 ethnobioconservation.com

Comparison of local ecological knowledge versus camera trapping to establish terrestrial wildlife baselines in community hunting territories within the Yangambi landscape in the Democratic Republic of Congo

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ABSTRACT

Baseline population data are fundamental to the development of wildlife management plans and are usually generated based on field surveys using sampling tools such as camera traps (CT). However, this method can be costly and ineffective with rare species or in wildlife-depleted areas. An alternative is to complement baseline wildlife population data with Local Ecological Knowledge (LEK)-based methods. We compared LEK and CT surveys in terms of their capacity to assess the status of terrestrial mammal species (richness, abundance, distribution) in the Yangambi landscape of the Democratic Republic of Congo. This region is heavily hunted and wildlife population densities are low. Species not captured by CT included naturally rare and endangered species that were instead recorded by interviewed hunters. LEK and CT abundance metrics were positively related for all species. For all medium- and large-sized species, the number of positive sites from LEK outnumbered the number of positive sites from the CT survey, indicating that hunters detected species over larger areas. Overall, our comparison suggests that LEK and CT methods can be used interchangeably to provide reliable information on relative abundance. Nevertheless, LEK appears as a more cost- effective alternative to camera trapping, particularly for hunted and depleted tropical forests.

Keywords: Wildlife surveys; community occupancy; Africa; Local Knowledge; Camera traps; tropical forests.

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SIGNIFICANCE STATEMENT

In this study, we compare information from Local Ecological Knowledge (LEK)-based methods with camera trap (CT) surveys in terms of their capacity to assess the status of terrestrial mammal species (richness, abundance, distribution) in the Yangambi landscape of the Democratic Republic of Congo. We show that naturally rare species recorded by LEK were not always captured by camera traps. Nevertheless, LEK and CT abundance metrics were positively related suggesting that they can interchangeably provide reliable information on relative abundance. Overall, our comparison shows that LEK is an efficient method for wildlife baselines and appears as a more cost-effective alternative to camera trapping, particularly for hunted and depleted tropical forests.

INTRODUCTION

At the landscape level, baseline population data (e.g., status, trends, habitat requirements and distribution) are fundamental to the development of wildlife management plans, whether for conservation interventions, restoration activities or for setting sustainable use guidelines (Sutherland et al. 2004; MacKenzie et al. 2006; Clare et al. 2017). Such information is usually generated through wildlife surveys (e.g., line transects, camera traps, etc.) carried out by biologists and often with support from local experts (Camino et al. 2020). Deploying these methodologies in robust and systematic ways can be challenging when funds are limited and information is required over large areas with limited accessibility, as their implementation requires considerable investments in human resources, time and equipment (Parry and Peres 2015). In dense tropical forests, moreover, the probabilities of detecting wildlife can be extremely low for rare, quiet and nocturnal species, and in areas of low visibility (MacKenzie et al. 2006; Carvalho et al. 2016), hence species misidentifications can be particularly high, particularly for sympatric species. As such, chances of obtaining false-absences, i.e., considering a species absent when it is present (Fragoso et al. 2016) or false-presences, i.e., registering the presence of a species that is absent (Clare et al. 2017) can significantly bias the information. In this context, camera trapping is a passive and non-invasive monitoring technique that has become increasingly popular (Burton et al. 2015), but still requires regular monitoring over long periods of observation to yield robust results (Tobler et al. 2008), limiting its application to large spatial scales (Burton et al. 2015).

Another approach which is gaining recognition among conservation practitioners consists in assessing wildlife status through interviews that elicit Local Ecological Knowledge (LEK)-based methods from people living in close vicinity to wildlife, often reliant on wildlife resources for their livelihoods and thus knowledgeable about the presence of animals in surrounding forests, both in the present and past (Braga-Pereira et al. 2020; Thompson et al. 2020). Over the past decade, LEK surveys have been successfully used to assess the status of wildlife presence and faunal depletion in the Neotropics (Parry and Peres 2015; Benchimol et al. 2017; Camino et al. 2020; Coomes et al. 2020; Zayonc and Coomes 2022; Braga-Pereira et al. 2022) and in African contexts (Gandiwa 2012; McPherson et al. 2016; Fopa et al. 2020; Madsen et al. 2020; Brittain et al. 2022). To account for possible bias, some studies have combined LEK with occupancy analysis to gather data on rare or wide-ranging species at large-scales (Martínez-Martí et al. 2016; Brittain et al. 2018, 2022). LEK techniques can be used to identify trends in wildlife presence (Zayonc and Coomes 2022) and their efficiency often exceeds that of more conventional survey techniques such as distance sampling or GPS telemetry (Parry and Peres 2015; McPherson et al. 2016). In a study conducted in the Brazilian and Peruvian Amazon, Braga-Pereira et al. (2022) compared LEK surveys with transect surveys and found that distance sampling on line transects was not as effective at surveying nocturnal and rare species; there was, however, significant agreement of population abundance indices for diurnal and game species, regardless of species social behaviour, body size, locomotion mode (terrestrial and arboreal) and habitat type. Madsen et al. (2020) compared LEK surveys to GPS collar data for African lion, cheetah and African wild dog in Kenya and concluded that LEK surveys could be used as a rapid and cost-efficient tool for assessing threatened species. However, comparisons of results from LEK based interviews and camera traps remain scant. Camino et al. (2020) compared LEK with camera trapping surveys in the South American Chaco and found that data derived from LEK increased detection probabilities while providing accurate information for three peccary species. In a recent study conducted in Cameroon, Brittain et al. (2022) found that LEK was particularly useful where camera trap detection rates were too low to produce robust occupancy model estimates, notably for rare or cryptic species. Thus, and notwithstanding the usefulness of camera trapping, these studies suggest that LEK surveys have a clear potential to generate accurate wildlife baselines.

Here, we aimed at comparing LEK surveys and camera trap surveys in community hunting territories located in the Yangambi landscape, Democratic Republic of Congo. The target region is known to be heavily hunted and withstands low wildlife population densities compared to other conservation zones in the Congo Basin forests (van Vliet et al. 2018; van Vliet et al. 2023). We compared LEK and camera trapping in terms of their capacity to deliver richness and abundance indicators, geographical distribution, and indicators of the status of wildlife populations. This study brings novel insights that contribute to the growing literature that develops or validates cost-effective wildlife assessment methods in community-based hunting management contexts.

MATERIAL AND METHODS

Study area

The Yangambi landscape is located in the North-East of the Democratic Republic of Congo (DRC), about 100 km West of Kisangani City in the Tshopo Province (Figure 1). Our focus within this landscape, are three Turumbo hunting territories (Weko, Yaselia/Bosukulu/Yaliboto and Lokeli). As it is typically observed in the Congo Basin forests, the landscape is characterized by a superposition of land tenures where customary hunting territories are overlapped as follows: The Yaselia/Bosukulu/Yaliboto and Lokeli hunting grounds are overlapped by the Yangambi Man and Biosphere Reserve (YBR) created in 1979 and the Weko hunting ground is overlapped by a logging concession established in 2003 and the Ngazi Reserve, which legal status is unclear (van Vliet et al. 2018). The logging concession has a validated forest management plan but official agreements have not yet been made with Weko with regards to hunting within their hunting grounds. As such, hunting is, in practice, carried out based on customary practice and, in theory, in accordance to national regulations.

The climate in this region is marked by two dry seasons (from December to mid-March and from June to July) that alternate with two rainy seasons (from April to May and from August to November). The landscape is covered by semi-deciduous dense forests and dense evergreen forests, with agricultural fields, young secondary forests and old secondary forests dominating around the villages (van Vliet et al. 2019).

The Yangambi landscape counts with an estimated population of 141,643 inhabitants distributed in the medium-sized town of Yangambi (10 districts) and villages in the Turumbo sector and Bamanga sectors. The population living in the three hunting territories which are the focus of our study belongs to the Turumbu ethnical group, which are specialized in hunting and traditional farming. Fishing is also an important, but secondary, alternative livelihood. Forest products (particularly wildmeat) significantly contribute to household food security (van Vliet et al. 2017). As such, this medium sized town drives a vibrant wildmeat trade from neighboring forests equivalent to about 145 tons of smoked wildmeat sold annually. The main species sold are small monkeys (38% of the carcasses) and red duikers (31%), followed by blue duikers, bush pigs and brush tailed porcupines (van Vliet et al. 2017). Hunters in the Yangambi landscape specialize in commercial hunting (more than 80% of the biomass is sold). Most of the meat sold in Yangambi (66% of the biomass) originates from Weko, where forests have remained more intact, with very low deforestation rates (Kyale Koy et al., 2019), as compared with forests around Yangambi.

Data collection

Local Ecological Knowledge

From March to May 2018, we conducted semi-structured interviews with a sample of 234 hunters from 14 individual villages (average 17, SD 16) belonging to 5 village groups (Weko, Yaselia/Bosukulu/Yaliboto and Lokeli), and sampling approximately 30% of the total number of active hunters (Table S1). These surveys were conducted as part of a wider ethnozoological study described in van Vliet et al. (2018). In each location, a discussion group was initially organized to produce a check list of mammal species present or supposed to exist in the landscape. Hunters were selected based on their availability and willingness to participate in the study. Then, semi-structured interviews were conducted with each hunter separately to inquire about the last observation made by each hunter for each mammal species listed in the group discussions. Each observation was geolocated on a printed map developed during a previous participatory mapping exercise (which methodology is explained in van Vliet et al. 2018) covered by a 4.2 * 4.2 km grid, where each cell could be identified by a number (Y axis) and a letter (X axis) (Figure 1). The size of the grid cells matches the average size of home ranges for most medium-sized species of mammals in our study area (duikers, rodents, bush pig, small monkeys, small carnivores, okapi).

The questionnaire was administered using KoBoCollect® on Android and consisted in two main sections: 1. a general section about the hunter: age, ethnicity, hunting frequency, number of years of hunting experience, most used hunting technique; 2. a section concerning the last time a species was observed by the hunter for a pre-determined list of mammal species developed based on species known to occur in the area (van Vliet et al., 2019). For each observation we indicated species, location (based on the grid),

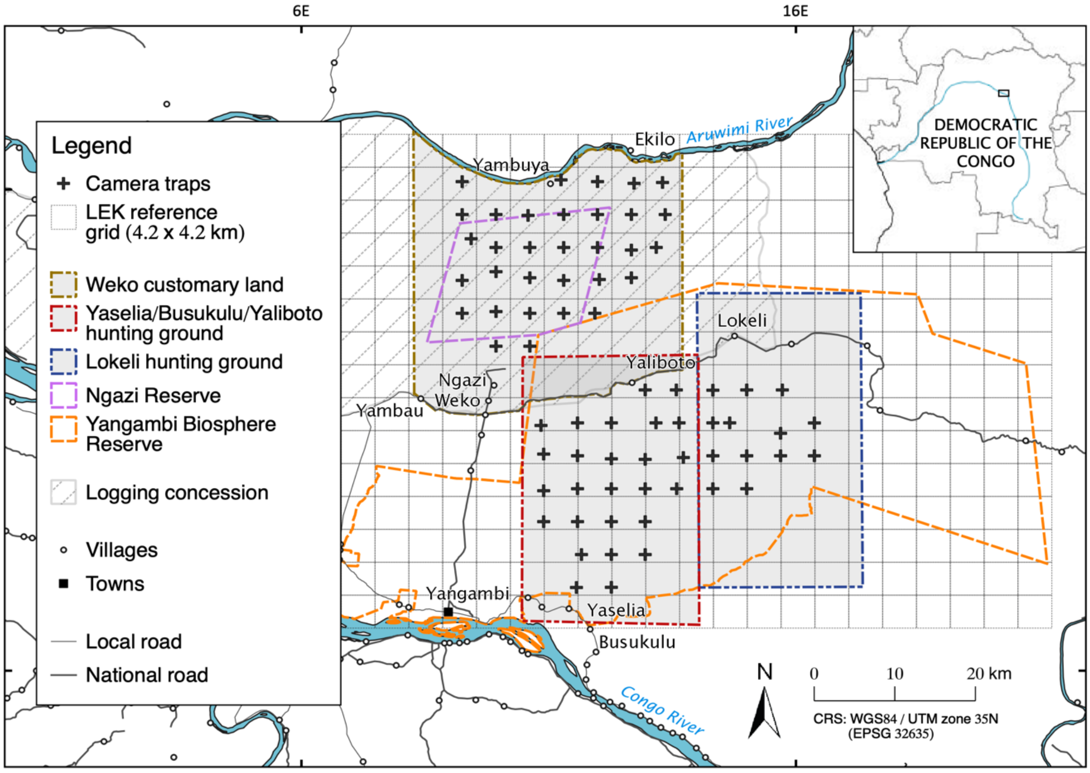

Figura 1. Map of the Yangambi landscape in the Democratic Republic of Congo showing the camera trap sites (black crosses) and the grid covering the hunting territories of the hunters interviewed.

date and type of observation (heard, seen, footprints, feces, nests, carcass, other).

Camera trapping

The objective of the camera trap survey was to assess species composition, and distribution, to derive metrics of abundance and wildlife status indicators comparable to those obtained from the LEK survey. Occupancy was modelled with account for imperfect detection (see below) as a proxy of distribution and abundance (MacKenzie and Nichols 2004). We used a systematic randomized sampling over a grid covering the YBR and Weko customary land. To ensure spatial independence between detections, we chose a 4.2 x 4.2 km grid, that approximates the map used for LEK interviews (Figure 1).

The sampling design included a total of 71 sampling positions divided into two blocks: 32 camera stations in Weko customary land and 39 camera stations in the YBR, within the Yaselia/Bosukulu/Yaliboto and Lokeli hunting grounds. Each block remained active for 47.6 days on average per camera from September to November 2018, representing an adequate effort in tropical forests (Kays et al. 2020). Due to malfunctioning or theft of the SD cards, only 59 camera traps worked effectively (29 in Weko customary land and 30 in YBR, Figure 1) for a realized sampling effort of 2,806 camera days. This survey served as a baseline for a subsequent multi-year camera trap study (van Vliet et al. 2023). The location of each camera station was chosen to maximize probability of detecting wildlife by targeting sites with animal signs. At each site, the camera field of view was cleaned to remove large leaves and other major obstacles in front of the camera. The cameras were secured to a tree (30-40 cm from the ground) and locked to prevent theft. No bait was used.

Image annotation was done manually, and the identification of mammals following Kingdon and Hoffmann (2013) and Kingdon (2014). When species identification was uncertain, records were excluded from analysis. As per LEK data, we pooled all images of genets and considered them as *Genetta* spp. (Gaubert et al. 2006). *Cephalophus nigrifrons* and *Cephalophus callipygus* were all observed in the camera trap pictures at least once and their existence in the study areas was confirmed with local hunters who name them Afoli and Mungala in Turumbo language, respectively. However, it was not possible to identify them to species level in all pictures where they were detected. As such, these species were also treated as one under the name of diurnal red duikers. Only species above 0.5 kg were considered in this study. Among the smallest species, only one Macroscelidea species (the elephant shrew (*Rhynchocyon cirnei*)) and a shrew (the forest giant squirrel (*Protoxerus stangeri*)) were distinctive and large enough to be reliably identified in the camera trap images. All smaller murid species and shrew species were discarded from the analysis.

All data collection routines were non-invasive in that they did not require any trapping or handling of animals, and therefore fully adhered to international ethical standards.

Data analyses

Analysis of LEK data

From the LEK survey, we pooled all observations of wildlife that occurred in 2018 (January to April) and 2017 (the whole year) to align LEK and camera trapping data temporally, as the latter were collected in 2018. Similarly, we used the subset of LEK data that were spatially related to the camera trap survey area. Records on last observations of a given species is a common indicator used to assess the date at which a species became extinct. In this study we also used records of "last observation"as an indicator for the abundance of non extirpated species based on the assumption that abundant species were more likely to be observed recently. We therefore assumed that the higher the number of hunters that had last observed a given species in 2017/2018, the higher the relative abundance of the species at a particular 4.2 x 4.2 cell. This metric was then related to species' abundance and occupancy metrics derived from camera trapping data, described hereafter.

Analysis of camera trapping data

From the camera trapping data, we first compiled the checklist of the mammal species detected and computed for each species the relative abundance index (RAI). This is the number of detection events separated by 15 min, divided by the sampling effort and multiplied by 100 (Rovero and Spitale 2016). We then used a multi-species occupancy model (MSOM; Dorazio et al. 2006) to estimate species' occupancy as the second metric, in addition to RAI, for comparison with the interview score from LEK data. Occupancy is the probability of site use by a species, estimated from repeated detections/non-detections of the species at a collection of sites, and well suited to camera trapping data (MacKenzie et al. 2002). A key value of occupancy is that it accounts for the probability that a species is detected when present at a site, hence addressing the pervasive issue of imperfect detection (MacKenzie et al. 2002). Here, we opted to use a multi-species model, rather than several single-species ones, as a more coherent approach and to fit a consistent suite of covariates across all species modelled in the community. We used occupancy in addition to RAI for the comparison with the interview data because while RAI measures raw site use intensity, occupancy is unbiased by detectability and describes species distribution. Therefore, we considered the two metrics highly complementary to properly assess how camera trapping data compared with LEK data.

We fitted the MSOM model with a suite of habitat covariates on occupancy and detection. Site occupancy covariates were the following: (1) distance to villages, as the distance to the nearest human settlement; (2) percentage of forest cover, as the proportion of 30x30 m pixels with a 90% of forest cover within a 4 km buffer and based on Hansen et al. (2013) Global Forest Change dataset; (3) distance to nearest trail; (4) distance to nearest road; and (5) distance to nearest river. We used the variable distance to nearest trail also as a covariate of detectability, on the assumption that trail may represent a source of disturbance to animals and hence their detection by camera traps may vary.

To run the MSOM, we first organized the camera trapping detections of all species in a 2-dimensional array Y, with elements y_{ki} where $k = 1$, ..., n being the species sampled at sites i, so that y_{ki} $\geq$1 if the species k was detected at site i, and $y_{ki} = 0$ if it was not detected. The state process (i.e., the presence of species across sites) was modelled as z_{ki}/w_k Bernoulli($w_k \psi k_{il}$), where $z_{ki} = 1$ for an occupied site and $z_{ki} = 0$ for an unoccupied site by species k, with ψ_{ki} representing the species-specific occupancy probability. We described the observation process (i.e., detection) as y_{ki}/z_{ki} Bernoulli($z_{ki}p_{kr}$), with p_{ki} representing the detection probability. To investigate the response of species occupancy to the set of covariates listed above, we regressed occupancy and detection probabilities of species across sites on site covariates using a logit-link function, and allowed the α and β coefficients to vary across species. Following an established approach (e.g., Zipkin et al. 2010; Rich et al. 2016), we fitted a single model that includes all covariates for which we hypothesised ecologically meaningful potential effects as follows:

$$logit(\psi_{ki}) = \beta0_k + \beta1_k \cdot dVill_i + \beta2_k \cdot forCover_i + \beta3_k \cdot dTrail_i + \beta4_k \cdot dRoad_i + \beta6_k \cdot dRiv_i$$

$$logit(p_{ki}) = \alpha0_k + \alpha1_k * dTrail_i$$

Where 'dVill' the distance to the nearest settlement, 'forCover' the % of forest cover, 'dTrail' the distance to nearest trail, 'dRoad' the distance to nearest road, 'dRiv' the distance to nearest river.

We implemented the models in a Bayesian framework using JAGS (version 4.3.0; Plummer 2003) via R (version 4.1.2; R Core Team 2017) with the R2jags package (version 0.7-1; Su and Masana 2015). We generated three parallel chains of 100,000 iterations with a burn-in of 20,000 iterations and thinning by 20 to derive summaries of parameter posterior distribution. Convergence of the Markov chains was satisfactory based on the Gelman-Rubin statistic, which was always ≤ 1.04 (Gelman and Rubin 1992).

Comparison of the two data sources

We compared results from the two data sources as follows:

(1) Checklist: we compared the number and composition of species detected by the two methods, considering only the predominantly ground-dwelling species, i.e., excluding arboreal and fossorial species as we deployed camera traps on the ground; hence, these cameras generally did not detect arboreal and fossorial species, or detected them unreliably. For the comparison we also considered a suite of species' attributes which may explain potential differences in detection. For example, hunters' observation may be influenced by the size of the species, the dietary guild or diel activity pattern that favor their catchability. These attributes were: body mass, dietary guild (sourced in Wilman et al. 2014), IUCN red list category (IUCN 2023), and diel activity pattern. For species that were camera trapped, the diel activity patterns were derived directly from the frequency distribution of the time stamp in the images, while for the few species detected only by LEK, the information on diel activity was sourced from Wilman et al. (2014).

(2) Abundance and occupancy: for species detected by both methods (LEK and camera trapping) we compared the abundance metrics by means of Spearman's correlation, given the small sample size and skewness of the data. Thus, we assessed the correlation between number of hunters that last observed the species (hereafter 'interview score') to both RAI and estimated species occupancy, hence assessing how the interview score calibrates to abundance metrics from camera trapping.

(3) Distribution: to compare the species distributions between methods, we produced species maps with the distribution of LEK observations across the sub-area where we deployed camera traps. For this analysis we targeted the 14 predominantly ground-dwelling species for which we had both LEK and camera trapping data, grouping *Cephalophus callipygus* and *C. nigrifrons* in diurnal red duikers. We then scored for each species the number of positive CT sites and LEK cells that were shared between datasets and computed the proportion of shared sites. A buffer of 100 m was considered around CT sites, to determine overlap with LEK positive cells. The first proportion quantifies how shared sites overlapped with successful LEK cells, while the second quantifies how shared sites overlapped with successful CT sites.

(4) Wildlife status indicators: we compared three indicators derived from the respective checklists: (1) the number of threatened species (including near threatened, vulnerable, endangered or critically endangered according to the IUCN 2023 red list), (2) the ratio between rodents and ungulates (Rowcliffe et al. 2003), and (3) the ratio between blue duikers and red duikers (Yasuoka et al. 2015).

RESULTS

Combining both methods, we detected 24 species of medium to large-sized terrestrial mammals (above 0.5 kg of body mass; Table 1). Hunters and camera traps also reported non-terrestrial species which we did not consider in this analysis (10 arboreal (primarily primates), one fossorial (*Orycteropus afer*), and one aquatic species (*Aonyx capensis*); Table S2). LEK interviews yielded 1,536 records of species that were last observed in 2017/2018, 1,434 (93%) of which were represented by direct observation of the animal or carcasses (animals seen or killed by the hunter/found dead in the forest), while the remaining 7% were from indirect records such as tracks, feces and vocalizations. Of the 24 terrestrial species, 22 were referred by LEK and 16 detected by the 59 camera traps; 8 were exclusive to LEK, while 2 to camera trap survey (*Crossarchus alexandri*, and *Civettictis civetta*; Table 1). Species not captured by camera traps included naturally rare and endangered or vulnerable species such as *Okapia johnstoni, Smutsia gigantea* and *Panthera pardus*. In addition, cameras did not capture two large diurnal ungulates, *Syncerus caffer nanus* and *Tragelaphus scriptus*, which like moving along streams or in marshy areas, two common medium sized nocturnal carnivores (*Atilax paludinosus* and *Bdeogale nigripes*) and one rodent common in deforested and agricultural areas (*Thryonomys swinderianus*).

Abundance and occupancy

In general, all species were detected by few camera traps, representing low naïve occupancies (range

Tabela 1. Checklist of ground-dwelling, small-, medium- and large-sized mammals in the Yangambi landscape (> 0.5 kg) detected by LEK and/or camera trapping. Two functional traits and the IUCN threat status are also indicated.

Order	Species	LEK	CT	Mass (kg)	Guild1	IUCN	Diel activity2
Macroscelidae	*Rhynchocyon cirnei*	x	x	0.5	insectiv	LC	C
Rodentia	*Protoxerus stangeri*	x	x	0.6	omniv	LC	D
Rodentia	*Cricetomys emini*	x	x	1.3	omniv	LC	N
Rodentia	*Atherurus africanus*	x	x	2.8	herb	LC	N
Rodentia	*Thryonomys swinderianus*	x		6.0	herb	LC	N
Pholidota	*Smutsia gigantea*	x		30	insectiv	EN	N
Cetartiodactyla	*Philantomba monticola*	x	x	5.0	herb	LC	C
Cetartiodactyla	*Hyemoschus aquaticus*	x	x	11.5	herb	LC	N
Cetartiodactyla	*Cephalophus dorsalis*	x	x	12	herb	NT	N
Cetartiodactyla	*Cephalophus nigrifrons*	x	x	18	herb	LC	C
Cetartiodactyla	*Cephalophus callipygus*	x	x	20.1	herb	LC	C
Cetartiodactyla	*Tragelaphus scriptus*	x		43	herb	LC	NCD
Cetartiodactyla	*Tragelaphus spekii*	x	x	100	herb	LC	D
Cetartiodactyla	*Potamochoerus porcus*	x	x	70	omniv	LC	DC
Cetartiodactyla	*Okapia johnstoni*	x		200	herb	EN	N
Artiodactyla	*Syncerus caffer nanus*	x		600	herb	NT	NC
Primates	*Papio anubis*	x	x	24	frug	LC	C
Primates	*Pan troglodytes*	x	x	50	omniv	EN	C
Carnivora	*Crossarchus alexandri*		x	1.5	insectiv	LC	D
Carnivora	*Bdeogale nigripes*	x		2.0	carn	LC	N
Carnivora	*Atilax paludinosus*	x		2.6	carn	LC	NC
Carnivora	*Civettictis civetta*		x	15	omniv	LC	NC
Carnivora	*Panthera pardus*	x		47.5	carn	VU	N
Carnivora	*Genetta* spp.	x	x	1.1 - 2.5	carn	LC	N

1 Dietary guild: carnivores (carn), herbivores (herb), insectivores (insectiv), omnivores (omniv)
2 Diel activity: D= diurnal, C=crepuscular, N=nocturnal.

2-31% of positive sites on total sites) and low RAI estimates (0.07-2.07 detection events every 100 camera days). The community occupancy model estimated an average occupancy for the pool of species of 0.154 (95% BCI 0.057-0.332) and an average detectability of 0.014 (95% BCI 0.005-0.033). Average occupancy across all species increased with the distance from roads (β = 0,319; -0,017-0,697; Table S3) and tended to increase with distance from trails, while the other covariates did not show marked effects. Average detectability was significantly higher near to the trails (α = -4.249; -5.218 - -3.432). As expected, estimated species' occupancy was higher than naïve occupancy but still tended to be small, as it ranged from 0.05 to 0.50 (Table S3).

The comparison of the interview score from LEK surveys and the camera trap metrics was possible for 14 species with certain correspondence of identification. LEK interview score and camera trapping abundance and occupancy metrics had a positive corre-

Tabela 2. Comparison of distributions between CT and LEK survey results in the Yangambi landscape, quantified by the number of positive CT sites and LEK cells, respectively, for the 13 species for which data from both methods were available. The matching (shared) considers a 1-km buffer around CT sites. An overlap index normalized to CT and LEK, respectively, is also shown (see Methods for more details). Species ordered as in Table 1.

Species	Positive CT sites	Positive LEK cells	Shared	Prop. shared (CT)	Prop. shared (LEK)
Rhynchocyon cirnei	6	23	2	333	87
Protoxerus stangeri	27	28	10	370	357
Cricetomys emini	21	39	19	905	487
Atherurus africanus	9	40	6	667	150
Philantomba monticola	17	36	11	647	306
Hyemoschus aquaticus	1	25	1	1000	40
Cephalophus dorsalis	18	37	11	611	297
Diurnal red duikers	5	26	4	800	154
Tragelaphus spekii	1	30	1	1000	33
Potamochoerus porcus	9	37	7	778	189
Papio anubis	2	16	0	0	0
Pan troglodytes	3	16	0	0	0
Genetta spp	5	44	5	1000	114

lation. This was significant between LEK interview score and RAI (Spearman's $rho = 0.615$, $p < 0.02$) and near significant between LEK interview score and estimated occupancy ($rho = 0.481$, $p = 0.08$; Figure 2).

Distribution

Within the area sampled by both methods, the number of positive cells from LEK surveys outnumbered the number of positive sites from camera trapping surveys, indicating that hunters detected species over a larger area than camera traps. This applied to all species except two small-sized species for which the number of positive CT sites was slightly larger (*Protoxerus stangeri*) and equal (*Rhynchocyon cirnei*) than the number of positive LEK cells (Table 2). Moreover, the number of positive LEK cells tended to be high for most species including many that were poorly detected by camera traps, indicating strong hunters' abilities to detect species. For example, the bushpig (*Potamochoerus porcus*) was observed by hunters across 37 cells out of 58 but detected by only 9 camera traps. In consequence, the proportion of shared sites related to positive LEK cells was generally small (mean 0.16, range 0-0.49). Instead, the proportion of shared sites related to positive CT sites was larger (mean 0.65, range 0-1), indicating that camera traps tended to detect species within the area where hunters also recorded them, with limited extent of records outside this area. Exceptions applied to two primates, *Papio anubis* and *Pan troglodytes*, that were recorded only at two and three CT sites, respectively, and did not overlap with the species distribution referred by hunters, and for the two small mammal species ($<$ 1kg) for which only nearly 35% of the positive CT sites coincide with positive LEK cells. For the rest of the other species, a high proportion ($>$ 60%) of shared sites in relation to positive CT sites was estimated, regardless of how commonly detected by camera traps the species were (Figure 1).

Wildlife status indicators

Both the proportion of threatened species and the ratio of rodents on ungulates were higher for the pool of species detected by LEK than by CT, while the proportions were identical between datasets for ratio

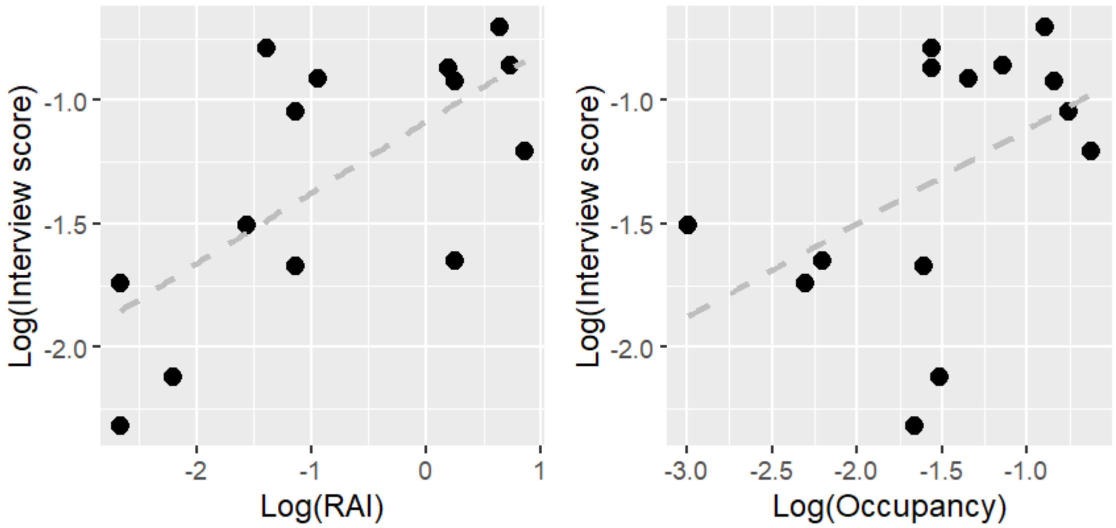

Figura 2. Bivariate plots of interview score from LEK surveys (i.e., the proportion of hunters that last saw a species) and two metrics from camera trapping, RAI (left chart) and estimated occupancy (right), both expressed in logarithmic scale. The dashed grey line shows the trend from a linear regression model.

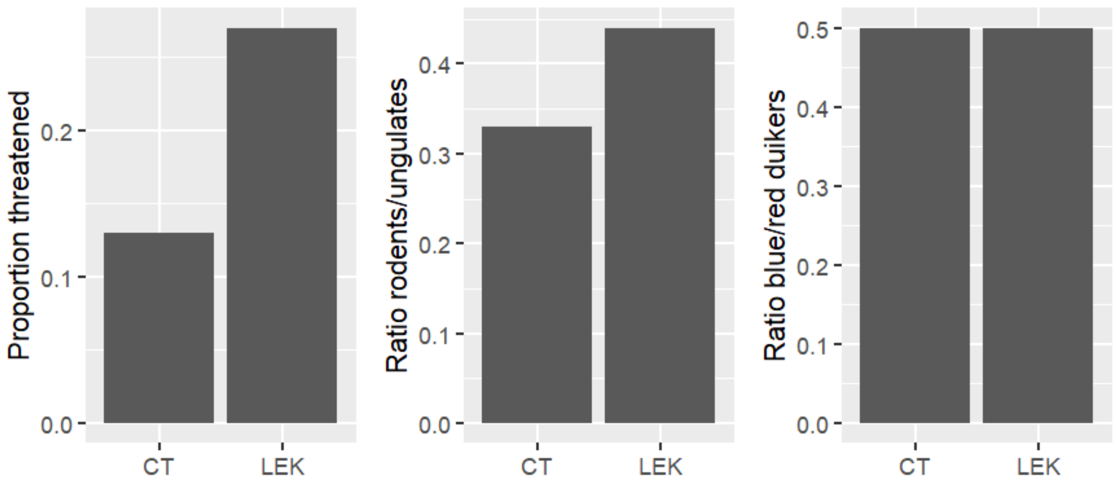

Figura 3. Results of the comparison of wildlife status indicators derived from camera trapping (CT) and Local Ecological Knowledge (LEK)-based methods.

between blue on red duikers (Figure 3).

DISCUSSION

Our study provides new insights into the use of LEK and camera traps for establishing terrestrial baselines in central African tropical hunted systems. We found that LEK outperforms CT in terms of checklist with the later failing to detect 7 terrestrial wildlife species that were instead detected by LEK. Thus, LEK appeared more efficient to confirm the presence of naturally rare and threatened species such as *Okapia johnsoni* and *Smutsia gigantea,* as well as species with specialized habitats that were not adequately sampled by the point sampling nature and inherently limited number of camera trap sites, such as *Thryonomys swinderianus*, which likes agricultural lands, or *Atilax paludinosus, Syncerus caffer* and *Tragelophus scriptus* which are more likely found in proximity to marshy areas or streams. Remarkably, the repeated use of camera trapping in subsequent years in the same sites did allow to confirm the presence of four of those species not initially detected by camera traps in 2018, but recorded by LEK in this study (van Vliet et al. in 2023). This suggests that for rare and cryptic animals, camera trap surveys should significantly increase the number of camera trap nights to provide a more accurate check list in line with hunter's knowledge. Nevertheless, the fact that the forest buffalo (*Syncerus caffer nanus*), okapi (*Okapia johnsoni*) and leopard (*Panthera pardus*) remained undetected by consecutive years of camera trap surveys could either suggest false positives in LEK responses or the failure of camera traps to detect extremely rare species. Taking into account the hunter‘s expertise in species recognition, the distinctiveness of the above-mentioned species, and based on our personal observation of a hunting event of two forest buffalos in Weko customary area, we deduced that is due to the failure of camera traps to detect rare species rather than false positives coming from LEK. It is very likely that these extremely rare species don't have established territories within the studied landscape and may incidentally cross from the other side of the Arwimi River, which could explain why they were never recorded in camera traps.

Our results are in line with previous studies that have also found estimates of wildlife population trends from local inhabitants to be particularly robust for rare species (van der Hoeven et al. 2004; Brittain et al. 2022). Indeed, in forests where the number of hunters is high, "more eyes on the ground" increases the likelihood of detection, which has also been shown to be especially useful where species densities are low (Turvey et al. 2015; Martínez-Martí et al. 2016). Our personal observation of hunters' behaviour in Weko reveals that hunters actively and capillary search for species across the area, hence resulting in a wider coverage of space, including in poorly represented micro-habitat types preferably used by some species. That most of the species detected only by LEK are nocturnal and/or crepuscular is interesting, as it contrasts with the expectation that camera traps would better detect nocturnal species while hunters would be limited in their ability to detect species at night. While this hypothesis might have been true for gun hunters, the use of traps designed to cover a large variety of species may increase detection by trap hunters for nocturnal species. In addition, hunting at night is known to have increased in the area (van Vliet et al. 2018) therefore suggesting that hunter encounters with nocturnal species is high.

Our results are particularly relevant with regards to the use of LEK and CT in contexts where determining wildlife baselines can be used to monitor the sustainability of hunting. This is because the good performance of LEK at determining species checklist, distribution and relative abundance as validated by camera trapping means that LEK surveys can reliably inform on the status of wildlife, to establish baselines and, potentially, for replication over time to assess changes. Towards this end the significant correlation of relative abundance metrics that we found between methods represents an important finding. In spite of the relatively small sample size ($N = 14$ species), it suggests that the hunters' observation score can be used as a proxy of relative abundance and occupancy. That RAI from camera trapping data calibrated better to LEK observation score than estimated occupancy may be because RAI more readily describes the gradient of intensity of site use while occupancy saturates at 1, and hence RAI and LEK observation score may reflect more similar detection processes. These findings in turn support our hypothesis that LEK can generate accurate terrestrial wildlife baselines and can be used to monitor changes in population over time, particularly where access to specialized equipment such as camera traps, resources and expertise to carry out surveys may be lacking (e.g., in community-led hunting management systems). As expected, the correlation of relative abundance metrics between the two methods was higher for common species (e.g., *Cricetomys emini*), suggesting that for establishing baselines camera traps may be better suited in areas where species are relatively more abundant than in areas where they are depleted by hunting, such as in protected areas or areas with low hunting pressure, while LEK methods may be particularly appropriate in areas with medium to high levels of hunting, where hunters thoroughly search for prey, therefore increasing detection probabilities of LEK. Nevertheless, our results also show that CT estimates are more conservative than LEK with regards to the representativeness of threatened species in the community and the ratio between rodents and ungulates, making it a safer method for conservative estimates of the status of wildlife populations.

The comparison between methods should consider the inherent biases of each. Camera trap data may be positively biased towards trap-curious and larger species (Wegge et al. 2004) and relies upon training and experience of field technicians (Kolowski and Forrester 2017). On the other hand, several studies have warned against the use of LEK data due to concerns over species misidentification (McKelvey et al. 2008; Molinari-Jobin et al. 2012), inaccurate knowledge of respondents (Ruddle and Davis 2013), social desirability bias (Leggett et al. 2003), or dishonest behaviour as a result of mistrust with the research team (Jacobsen et al. 2018). Furthermore, the two methods are based on inherently different sampling efforts and detection processes: LEK data were related to observations made by hundreds of hunters over 16 months, and usually they spent 5 to 10 days per month in the forest concentrating their searches at dawn and dusk when hunting is more profitable. In contrast, camera trap data were derived from 59 sites continuously monitoring for nearly 50 days. In spite of these differences, the comparison of distribution patterns highlighted the ability of hunters to detect species across most of the area, whereas camera traps generally detected species within a smaller area, with several species found only at a few sites but still widely recorded by hunters.

Besides the different sampling biases, efforts required (time, personnel, costs) and feasibility also need consideration. Camera trapping requires an initial capital investment for the purchase (and importation in areas such as ours) of cameras, GPS, batteries and SD cards, in addition to expenditures related to setting and collecting the cameras. Camera traps may malfunction or be stolen (Larrucea et al. 2007; Burton et al. 2012), especially when deployed outside protected areas, reducing the data available and potentially preventing robust analyses. On the contrary, as used in this study, only requires a one-time survey per year, with low costs basically related to the transportation to the villages for data collection. Results of the LEK surveys collected with Kobocollect® (with automatically generated reports) are readily available locally, while camera trapping requires time-consuming post-processing of images on a computer prior to analyses. While these steps begin to be greatly facilitated by AI-aided platform such as Wildlife Insights (www.wildlifeinsights.org), expert knowledge and Internet connections are still required to process and analyze the raw data.

In conclusion, this study suggests that the combination of LEK and camera trapping is efficient to establish terrestrial wildlife assemblages. Nevertheless, the use of LEK is particularly well suited in areas that are hunted on a regular basis, and therefore well known from local hunters. Data derived from LEK is more cost efficient, particularly for rare species. This is notwithstanding the well-established potential of systematic camera trapping for estimating and modelling population and community metrics that cannot be otherwise obtained, and for robust, multi-annual monitoring (reviewed in Rovero and Kays 2021). Indeed, while camera trapping is well suitable to standardization and hence replication across space and time, further research is needed to optimize LEK methods, fine tune methodologies and hence maximize their robustness and standardization. Currently available studies using LEK vary largely with regards to the sampling, the profile of the interviewees (gender, occupation, level of experience of the forest), and the questions asked (last observation; last kill; all observations in a given time period, etc.), greatly reducing the comparability between studies. With these issues in mind, our study shows that LEK is an efficient approach to establish wildlife baselines particularly in hunted and depleted tropical forests.

ACKNOWLEDGMENT

We are very thankful to the village leaders and hunters who participated in this study. We would also like to thank SODEFOR for their constructive comments on this study. This work would not have been possible without the support from FORETS project. This research received funding from USAID and the European Union. It is part of the Bushmeat Research Initiative from CIFOR under the framework of the CGIAR Research Programme on Forests, Trees, and Agroforestry. We thank the reviewers and editors of the journal for the very constructive comments.

DATA AVAILABILITY

The data used to support the findings of this study are available from the corresponding author upon reasonable request.

CONFLICT OF INTEREST

The authors have no conflicts of interest to declare.

CONTRIBUTION STATEMENT

Conceptualization and writing the first draft of the manuscript: NvV and FR.
Study design and data collection: NvV, JN and JN.
Data management and analysis: FR and SQ.
Review and final writing of the manuscript: SQ, PC and RN.

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Received: 01 June 2022
Accepted: 29 July 2022
Published: 17 October 2023

Editor: Rômulo Alves