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Research Article

Watching a movie or going for a walk? Testing different Sun bear (*Helarctos malayanus*) occupancy monitoring schemes

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Abstract

Size and distribution of wild populations are key elements in determining their conservation status, especially for vulnerable and elusive species. Therefore, choosing the proper monitoring method is fundamental to estimate population indices and consequently address conservation actions. In this study we worked in Rakhine State, Myanmar applying and comparing two occupancy-based sampling methods to evaluate Sun bear (*Helarctos malayanus*) presence: camera traps and sign survey (line transects). Moreover, to apply occupancy models it is necessary to establish length (time or space) of sampling occasions, therefore for both methodologies we tested four different sampling intensities to explore if results are affected by different temporal or spatial replicates. Both occupancy and detectability values varied between the two methods: we found lower values from camera traps analysis with no differences between different sampling occasions/segment lengths. Sign survey showed higher values for both parameters but changes in spatial segment lengths (line transects) affect occupancy estimates. Overall camera traps represent a more appropriate tool to study Sun bears in tropical forest habitats found in our study area. Our results provide useful information to plan an efficient monitoring scheme for bears in tropical forests.

Introduction

Monitoring large carnivores is often difficult and requires great effort since they often have an elusive behaviour and live in low-density populations (Kendall et al., 1992; Linnell et al., 1998; Thompson, 2004). Thus, finding a reliable method for monitoring these species could be troublesome, not just for species' elusiveness or low densities, but also because a single population can inhabit very large areas, leading to high sampling costs (Link and Sauer, 1997; Schwarz and Seber, 1999; Zhou and Griffiths, 2007). However, density and distribution are key parameters in conservation (Wilson and Delahay, 2001) and finding a proper monitoring scheme to measure these variables is of fundamental importance.

During the last decade, researchers have made an increased use of camera trapping as an alternative technique to those that use indirect signs, and also for large carnivores many studies successfully used camera traps (Long, 2008). Camera trapping has proven successful in determining species presence (Hedwig et al., 2018; Van der Weyde et al., 2018) and in some cases also abundance of medium and large terrestrial mammals (Bengsen et al., 2012; Wearn et al., 2017; Jiménez et al., 2018; Bersacola et al., 2019), even in difficult habitats, such as the tropics, where camera traps placement is not trivial (Karanth and Nichols, 1998; Kawanishi and Sunquist, 2004). Where initial costs of camera trap usage may be high, the technique can be less expensive than alternative methods in the long term (De Bondi et al., 2010; Welbourne et al., 2015). On the other hand, their efficient use requires a set of parameters that needs to be carefully planned (Lepard et al., 2018) and sampling schemes must be adapted and tested to optimize detection of the target species in different habitats (Stokeld et al., 2016).

Here, we use the Malayan Sun bear (H*elarctos malayanus*) to test for differences in the results of occupancy modelling based on presence/(pseudo–)absence data between two commonly used methods: camera traps and signs of presence recorded along line transects. We also explored whether changing study design parameters could affect the efficiency and reliability of monitoring. Indeed, as affirmed by Gaidet-Drapier et al. (2006) the use of different methods to study medium and large mammals will always reveal different levels of accuracy and precision, as well as different cost-benefit ratios.

The Malayan Sun bear is one of the least known among Ursidae and studies on population trends are not numerous (e.g. Augeri, 2005; Ngo-

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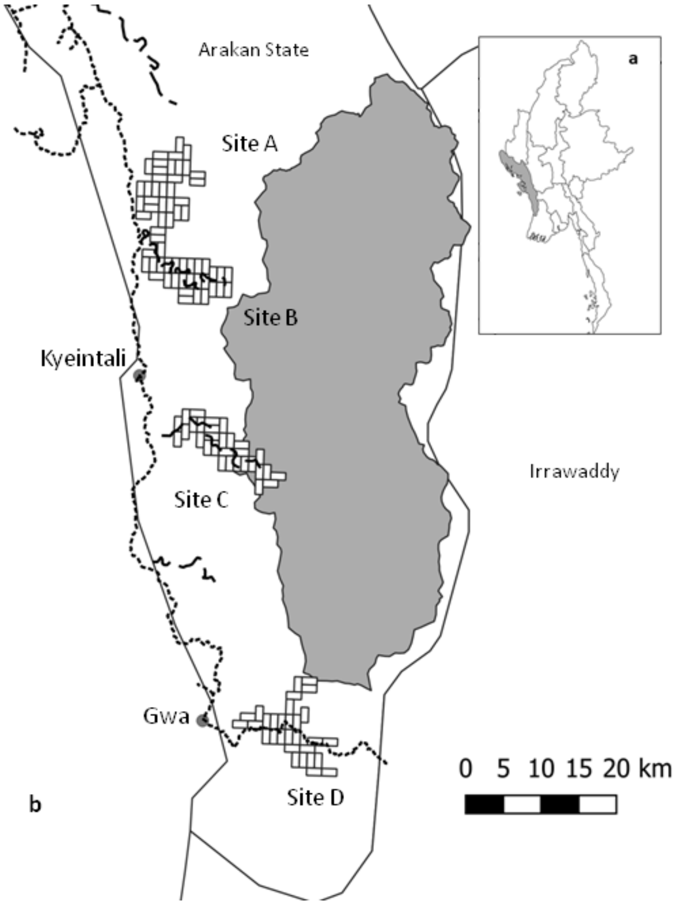

Figure 1 – **a**: Myanmar border, Rakhine state in gray; **b**: Study sites. Rakhine Yoma Elephant Range (in gray). Dotted lines: main roads. Dashed lines: main rivers.

prasert et al., 2011; Steinmetz et al., 2011, 2013; Fredriksson, 2012), or deal with populations in different conditions (such as insular environments, Linkie et al., 2007; Wong et al., 2013; Wong and Linkie, 2013; Guharajan et al., 2018). Sun bears are considered as Vulnerable by IUCN (Scotson et al., 2017), but robust data on the presence of this species across its distribution range are few, and a standardized methodology to assess species occurrence, distribution and abundance is decisively needed (Scotson et al., 2017).

Camera traps were used for single-season occupancy modelling, testing different temporal replications (MacKenzie et al., 2006). We also used bear sign survey along transects to estimates detectability and occupancy (Hidden Markov occupancy model, Hines et al., 2010) using spatial instead of temporal replicates (Srivathsa et al., 2018). Both methods have been commonly used to study bears in tropical regions (Akhtar et al., 2004; Ríos-Uzeda et al., 2007; Steinmetz et al., 2011; Ramesh et al., 2012; Sethy and Chuahan, 2016; Guharajan et al., 2018), but without a critical analysis of potential methodological biases or flaws (but see Srivathsa et al., 2018).

Materials and methods

Study area

Myanmar, the largest country in South-East Asia, is part of the Sundaic subregion of the Indo-Malayan Realm (MacKinnon and MacKinnon, 1986). Due to the combination and interaction of geography, topography, climate, pattern of seasonal rainfall, presence of high mountains and major rivers, Myanmar presents a great variety of different habitats, and ecosystems supporting a rich biodiversity with about 63% of the mainland covered by forests, but only 38% can be considered intact (Bhagwat et al., 2017). In this study, we worked in Rakhine State, on the western coast of Myanmar. The study area is located in the central part of the State, and study sites are close to the border of the Rakhine Yoma Elephant Range (17°22′0″ N, 94°36′0″ E; Fig. 1).

The orography of the area consists of a series of steep ridges running from north to south, with the main drainage lines cutting them from east to west. The area is famous for luxuriant patches of evergreen forest as well as for the presence of bamboo brakes. It was selected for its vulnerability to the loss of biodiversity due to human pressure with logging for timber, firewood or poles, forest encroachment for cultivation (both permanent and shifting) and trade-driven illegal hunting of endangered species as major threats. Project activities were carried out in particular in the areas of Gwa and Thandwe townships where four monitoring sites (Fig. 1) were selected for a total surface area of 240km^2.

Data collection: camera traps

Monitoring was carried out between November 2016 and March 2017, the dry season in Myanmar.

At each site 30 camera traps (Acorn Ltl-5210) were set according to a 2×1 km rectangular cell pattern and trap distances were checked during placement using GPS waypoints. This pattern was chosen based on average daily bear movements (1.45 km, Wong et al., 2004) and due to the fact that bear home range can overlap (average home range 14.8 ± 6.1 (SD) km^2, Wong et al., 2004). We set the majority of the cameras close to the trails to increase the probability of photographing our target species. Each camera was locked in a box to a tree at an average height from the ground of 60 cm. Camera traps were set to record short (20") videos, with a two minutes delay between subsequent recordings. The video resolution was set at 640×480 px and the PIR sensitivity level at "medium" with side PIR active. For this project, 60 camera traps were available, and they were activated initially for a minimum of 45 days simultaneously in two sites out of four and subsequently moved to the other two sites. Having no reference studies in the mainland, and having to rely on 60 available camera traps, we preferred to cover a larger area (240 km^2) and keep the cameras active for a shorter period (45 days). As suggested by MacKenzie and Royle (2005) surveying more sampling units less intensively is better than the opposite for rare species as Sun bear. While the camera traps were running automatically, we did not control them to avoid disturbance.

Data collection: sign survey (transects)

Transect were walked during camera traps deployment. Trained operators walked slowly, actively searching for bear trails or any sign of bear presence (e.g. claw marks) in a one-meter-wide strip on both the right and the left of the trail (2 m overall strip width), recording detections on a field data sheet and geo-referencing them with a GPS (Garmin GPSMAP 64s, average positioning error ±10 m). Fresh signs can be distinguished between the two bear species present in the area (Sun bear and Asiatic black bear), and we followed the procedure proposed by Steinmetz and Garshelis (2008): from measurements of hind foot claw marks on climbed trees, we decided to use the measure of 5 claws widths as the shortest straight line between toes; we classified as Sun bear marks all the sizes less than 8.2 cm wide, and as Asiatic black bear signs with width >9 cm. Since this discrimination was not possible for recent and old signs, only fresh signs were used to calculate detectability and occupancy using transects (see below). Bear signs age was assessed following Steinmetz and Garshelis (2010) based on the degree of bark regrowth, all signs aged less than 3 months were considered as "fresh" , signs classified as 3 to 9 months old were identified as "recent" and signs older than 10 months were recorded as "old".

Data analysis

All analysis were performed using the R software (R Core Team, 2018). Occupancy modelling for camera traps followed the workflow proposed by Rovero and Zimmermann (2016) and relied on the R package "`unmarked`" (Fiske and Chandler, 2011) applying single-season modelling (MacKenzie et al., 2002). The occupancy model selected for transects was the Hidden Markov occupancy model described by Hines et al. (2010) and used in a similar study also by Srivathsa et al. (2018). This model takes into account the possible pseudo-replication given by transects where animals move along linear routes, estimating two additional parameters (in addition to the detectability and occupancy): Θ^0

Table 1 – Detectability (p) and Occupancy (Ψ) with camera traps, calculated considering four different sampling occasions. SE: standard error.

p (±SE)	Ψ (±SE)	No. videos	Sampling occasions (days)
0.020 ± 0.006	0.17 ± 0.04	31	1
0.05 ± 0.01	0.17 ± 0.04	31	2
0.11 ± 0.02	0.18 ± 0.04	29	5
0.16 ± 0.03	0.17 ± 0.04	28	7

Table 2 – Detectability and Occupancy with transects calculated considering four different segment length. SE: standard error; Θ^0 and Θ^1: probability of presence conditional respectively on Sun bear absence and presence in the previous segment.

p (±SE)	Ψ (±SE)	Θ^0 (±SE)	Θ^1 (±SE)	No. signs	Segment length (m)
0.49 ± 0.04	0.56 ± 0.04	0.0060 ± 0.0001	0.30 ± 0.02	13	100
0.50 ± 0.04	0.57 ± 0.04	0.010 ± 0.005	0.32 ± 0.02	12	200
0.49 ± 0.04	0.40 ± 0.04	0.020 ± 0.001	0.31 ± 0.03	11	500
0.50 ± 0.04	0.42 ± 0.04	0.040 ± 0.002	0.35 ± 0.03	10	1000

and Θ^1, that is the probability of presence of the species conditional on the absence or presence in the previous transect segments. In order to find the most suitable survey method and to explore how occupancy and detectability estimates were affected by applying different survey parameters, we used raw detection/non-detection histories for different temporal (camera traps) or spatial (transects) replicates. Camera traps data were compared for 1, 2, 5 and 7 days sampling duration, while for transects (ranging from 2 to 7 km) data for 100, 200, 500 and 1000 m transect segments were compared.

We investigated the differences for all the sampling replicates for each method applying a type III ANOVA to a set of value obtained generating 999 random normal distributions with mean and standard deviation equal to occupancy values obtained by Mackenzie and Hidden Markov models (HMM, both for camera traps and transects). In case of significant differences between sampling occasions or transect segment lengths, a post hoc test (Tukey HSD test) was used to identify pair-wise differences between replicates.

Results

Since in the study area we did not record any relevant change (i.e. no fires, clearings, exceptional meteorological events, etc.) across the four sites, and the monitoring scheme was completed during the same season, we decided to pool data of the four sites, increasing sample size for both camera traps and transects.

Camera traps

From 116 camera traps (4 malfunctioned) we recorded 4477 videos where it was possible to identify what triggered the camera trap. More than 30 taxa were recorded, 22 of them recognisable at species level. A total of 31 not temporally correlated (not occurring in the same day) sun bear videos were recorded in the area. Table 1 reports the detectability and occupancy values for all the study sites, with the four different temporal sampling occasions. Occupancy values do not vary much ranging from a minimum of 0.17 (±0.04) with 1 day to a maximum of 0.18 (±0.04) with 5 days. Detectability increased with the number sampling occasions: from 0.02 (±0.006) of 1 day to 0.16 (±0.03) of 7 days (Tab. 1).

Sign surveys (transects)

Thirty-six transects were monitored in the study area for a total length of 313 km. In total, we recorded 112 signs of Sun bear presence, 89 of which were classified as old, 10 as recent and only 13 (14.5%) as fresh. Detectability and occupancy estimates using only fresh signs of presence with the four different segment lengths are listed in Tab. 2. Occupancy values ranged from a minimum of 0.40 (±0.04) with a 500 m sampling distance to a maximum of 0.57 (±0.04) with a 200 m sampling distance. Detectability remained almost unchanged across the different sampling distances: from 0.49 (±0.04) at 100 m to 0.50 (±0.04) at 1000 m.

Table 2 also reports the values for the parameters Θ^0 and Θ^1 used to estimate the conditional probabilities of Sun bear presence in the preceding transect segment in the HMM. These parameters could suggest evidence for possible autocorrelation between spatial replicates (Hines et al., 2010). In this study, estimates indicated no correlation and did not differ greatly with different sampling segment length: Θ^0 ranged from 0.006 (±0.0001) at 100 m to 0.04 (±0.002) at 1000 m and Θ^1 from 0.30 (±0.02) at 100 m to 0.35 (±0.03) at 1000 m.

Sampling occasions/transect segment length

Comparing the four temporal occasions of camera trap sampling, we found no differences among replicates ($F_{(3,3992)}$=0.66; p=0.58; R^2=0.0004). In contrast, there was a significant difference for transects caused by the number of replicates ($F_{(3,3992)}$=119.8; $p<0.05$; R^2=0.082). The post hoc test showed a significant difference (all $p<0.05$) at all the levels (see Tab.3) except between 100 m and 200 m sampling lengths (difference in occupancy estimates=0.013, p adjusted=0.68).

Discussion

To investigate species presence, occupancy modelling can provide reliable estimates when monitoring rare terrestrial tropical species, and can help improving their management strategies and therefore their conservation status (Linkie et al., 2007). Often studies on the same species use different methodologies with different spatial or temporal granularities, sometimes making difficult to compare results across different studies.

In the present study, we confirmed the presence of Sun bear in Rakhine State in all the four study sites, both with camera traps and presence signs along transects. We found that using two different methods could lead at different occupancy estimates, and on the methodological side, our comparisons indeed revealed that occupancy values are higher when calculated with data collected along transects than with camera traps. In addition we found that occupancy estimates changed depending on the sampling strategy used for transects, i.e. they depend on segment length. This is true for occupancy obtained from sign survey, even if detectability varied little among transect lengths. This variability in sign survey occupancy estimates further points out how difficult and possibly unreliable comparisons among different studies, with only slightly different monitoring protocols, could be. On the other hand, occupancy results obtained from camera traps did not change with different temporal sampling occasions, indicating more robust approach and suggesting that comparing results between camera trapping studies that used different temporal replicates is possible. We suggest therefore when planning monitoring schemes to carefully evaluate the most appropriate methodology for the species since minimal differences in methods could lead to different occupancy estimates.

Table 3 – Post-hoc Tukey HSD test results for all the pairwise comparisons among different transect spatial replicates.

Spatial replicates	Difference	95% confidence int. Lower	Upper	p adjusted
1000–100 m	−0.13	−0.16	−0.10	<0.05
200–100 m	0.01	−0.01	0.04	0.67
500–100 m	−0.16	−0.19	−0.13	<0.05
200–1000 m	0.14	0.11	0.17	<0.05
500–1000 m	−0.03	−0.06	−0.001	0.03
500–200 m	−0.17	−0.20	−0.14	<0.05

Both occupancy and detectability estimates varied between the two methods (Tab. 1 and 2). If we look only at occupancy values, transect-based estimates were higher than camera-trap based ones (mean occupancy: transects=0.48; camera traps=0.17) but these values are influenced by a different detectability given by the two methods (mean detectability: transects=0.49; camera traps=0.08). As affirmed also by Srivathsa et al. (2018), detection probabilities of the two methods are not directly comparable. Detectability for camera traps is referred to the probability of detecting the species in a site given its presence in the site (MacKenzie et al., 2002). For indirect sign survey calculated with the Hines et al. (2010) model instead, detectability is the probability of detecting the species in a spatial replicate given the presence of the species in the site but also in the previous replicate. In addition, the detection with the last method is also influenced by the expertise of the operators in recognizing indirect signs, and sign visibility often depends on other factors (independent of operator capacity) such as habitat structure (density of the vegetation). This source of bias is not present when using camera traps.

An interesting results from the HMM correlated detections model is that Θ^1 values seemed to be low (from 0.30±0.02 at 100 m to 0.35±0.03 at 1000 m) compared to other studies on other Asian bears (e.g. Srivathsa et al., 2018 found Θ^1=0.86±0.22 with 1000 m spatial replicates). Our results suggests that the probability of Sun bear presence in a segment (conditioned on the presence in the previous segment) is not high, primarily due to low population density in the area (as we can see also from Θ^0 values), but suggesting also that probably our target species does not move in straight lines as one would expect. Indeed, passing from a 100 m to 1000 m replicate resolution, bear signs decreased from 13 to 10. We did not find anything similar in literature but an expert opinion could be that Sun bears move randomly in the forests and not along linear paths, making the species even more difficult to detect.

Considering camera traps results, occupancy estimates are consistent and uniform when changing the duration of sampling occasions. For example, from 1 to 7 days, detection probability ranged from 0.02 (±0.006) to 0.16 (±0.03) but the occupancy estimates was almost the same: from 0.17 (±0.04) to 0.18 (±0.04). These results suggested that when dealing with Sun bear monitoring, the choice of different temporal occasion lengths for camera trapping does not affect occupancy estimates. Thus, for camera traps, comparing the four different temporal sampling occasions, we found there are no significant differences in occupancy values. This is an interesting result, since many studies used different temporal occasion lengths: from 7 days in Borneo (Guharajan et al., 2018) to 14 days in Sumatra (Linkie et al., 2007; Wong et al., 2013; Wong and Linkie, 2013).

The comparison between different spatio-temporal schemes for transects showed that estimates varied with different spatial replicate sizes; using differently sized spatial replicates could thus affect our occupancy results, and this was true for all the spatial replicates except at 100 m and 200 m segment lengths.

Camera traps were also more likely to detect the species (31 different detections) than transects (13 different fresh indirect signs). Sign survey suffered of low sample sizes because in a tropical forest the sampling effort needed to have unbiased results is very high (despite the apparent economic advantage of the method).

In conclusion, camera traps had a higher efficiency than transects to monitor sun bear in tropical forest habitat. Even if the initial cost is higher than for transects, camera traps could be more cost-effective in the long term, providing reliable results on species distribution and abundance, as well as other useful information such as activity rhythms (Ridout and Linkie, 2009; Harmsen et al., 2011; Rowcliffe et al., 2014); contribute to checklisting the faunal community (Rovero and De Luca, 2007; Albaba, 2016; Pereira et al., 2018); and monitoring the intensity of human activities in the study area (Parsons et al., 2016; Oberosler et al., 2017)). In addition in the particular case of Sun bear monitoring, the transects approach can suffer from two main problems. First, it is not always possible to distinguish between signs at the species level, given that the rapid growth of trees in tropical forests can alter the distances between claw signs and this can easily lead to errors such as confounding young Asiatic black bears and adult Sun bears. Second, considering only fresh signs, for the reasons stated above, can considerably decrease the amount of bear signs detected (in this study 13 fresh signs out of a total of 112 signs), reducing the potential sample size per survey effort. However, we suggest to consider using only fresh signs, not just for sign species attribution problems, but also to be able to monitor changes in species abundance over a 3 months time horizon, instead of several years (see also Steinmetz and Garshelis, 2010). In general, we found that dealing with detectability and occupancy estimates at small (but economically sustainable) sample sizes could lead to different results using different methods and sampling efforts.

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