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Source: Wildlife Biology, 2021(1)

Published By: Nordic Board for Wildlife Research

URL: https://doi.org/10.2981/wlb.00771

Wildlife Biology 2021: wlb.00771
doi: 10.2981/wlb.00771

Subject Editor: Maria Johansson. Editor-in-Chief: Ilse Storch. Accepted 8 February 2021

Leveraging local knowledge to estimate wildlife densities in bornean tropical rainforests

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Wildlife managers and researchers need to understand the status of the wildlife populations they are trying to conserve. Though various methods have been developed to monitor wildlife in their natural habitats, the complexity and accessibility of most techniques often limit their usability. Such techniques often take significant resources and time to deliver results, and methodological noncompliance may lower the reliability of results. Given this need to study wildlife populations reliably, quickly and within financial and human resource constraints faced by wildlife managers, we assessed the reliability and effort required to carry the pooling local expert opinion (PLEO) method as articulated by Hoeven et al. (2004). We did this by comparing density estimates of several wildlife species derived by following the PLEO method with those estimated using a conventional method along with results from the literature on wildlife monitoring studies from Bornean rainforests. Our analysis shows that the PLEO methodology provides an effective and complementary tool to estimate wildlife densities in tropical rainforests. We suggest that by incorporating the PLEO methodology into regular monitoring activity, conservation NGOs can create a platform that allows for participatory wildlife monitoring and create the platform to involve local communities in biodiversity conservation.

Keywords: density estimates, indigenous knowledge, local communities, PLEO, wildlife monitoring

Monitoring wildlife populations is an essential part of managing biodiversity (Stem et al. 2005, Nichols and Williams 2006, Gardner et al. 2008). Although various methods are used to study wildlife populations, each method varies from others in terms of advantages and limitations in their applications (Wilson and Delahay 2001, Davison et al. 2002, Hoeven 2007, Boitani and Powell 2012, Mathai et al. 2013, Mills 2013, Buckland et al. 2015). Conventional methods predominantly used to study different aspects of wildlife populations such as population size, abundance and densities include transect and point count survey based on distance sampling techniques, and camera trap surveys following the SECR method (Arendt et al. 1999, Simon et al. 2002, Riley 2003, Lee and Marsden 2008, Sastramidjaja et al. 2015, Paddock et al. 2020). As with any wildlife survey methodology, these methods require researchers to follow standard protocols and assumptions, which if violated leads to misleading measurements that further results in erroneous inferences about population under study (Pollock et al. 2002, Lee and Marsden 2008, Foster and Harmsen 2012, Mathai et al. 2013).

In particular, distance sampling techniques require careful consideration of four key assumptions, failure of which requires additional modifications of the standard method in order to provide robust density estimates (Mathai et al. 2013, Buckland et al. 2015). Moreover, distance sampling techniques combine multiple estimates to ascertain uncertainty (encounter rate, detectability, cluster size) in study designs, and therefore researchers need to minimize the sources of such uncertainties (Buckland et al. 2004, 2015). For example, one way to reduce uncertainties in detectability (variance of detection functions) is to increase detections through longer transects (Plumptre 2000). However, difficult terrain of the sampling unit often hinders ability of observers to carry out detections efficiently over long distances that undermines estimating the variance of detection function reliably (Gale and Thongaree 2006, Mathai et al. 2013). In order to avoid biases, transect surveys need to be carried out

multiple times a year to account for seasonal variations in animal behavior, employ highly trained observers able to collect high quality data, and collect large enough data sets that allow calculating detection probabilities (Fashing and Cords 2000, Buckland et al. 2004, 2015, Meijaard and Sheil 2007). Therefore, factors such as costs and expertise in terms of finance, technology and trained human resource required to gather sufficient data and carry out robust analysis often acts as practical constraints for researchers studying wildlife populations (Marsden 1999, Stem et al. 2005, Foster and Harmsen 2012).

Though some of these limitations of traditional methods are circumvented using camera-trap surveys, it still requires careful consideration of methodological and logistical factors to allow for reliable extrapolation of inferences from the sample area to the study area (Thomas et al. 2007, Foster and Harmsen 2012, Mathai et al. 2013, Miller et al. 2019). In particular, standardizing camera-trap data over time and space creates difficulties to determine population densities using such data (O'Brien and Kinnaird 2008, Foster and Harmsen 2012, Sastramidjaja et al. 2015). Moreover, camera trap data are often incomparable as researchers can choose among a variety of modelling approaches for data analysis that best fit their data (Brodie and Giordano 2012, Foster and Harmsen 2012). Due to high costs and logistical issues, camera traps are seldom used to study wide-ranging species as it requires spacing camera traps over large areas that go beyond the financial scope of most studies (Foster and Harmsen 2012). Though successfully used in detecting inconspicuous and rare bird and arboreal species in tropical forests, camera-trap data were found to consistently record fewer species when compared to point-transects due to limitations arising from their static observation points, differences in animal's activity budget and inability to calculate density estimates (Whitworth et al. 2016, Fontúrbel et al. 2020).

It is clear that conventional methods of estimating wildlife densities often require collecting data over long time periods, using significant resources in terms of equipment, finance and personnel, without which robust analysis and reliable estimates are not possible. Especially for transect and point transect surveys this inevitably requires carrying out numerous repetitive transects over long periods just to gather enough observations that allow analysis. Even in cases where sufficient data is available, grassroots conservation NGOs may still face the challenge of carrying out data analysis as there is a steep learning curve associated with using specialist software (Johnson et al. 2018). Recent developments in hierarchal modeling methods and machine learning software, such as R and Python coding, have exacerbated the learning curve and have created human resource restrictions for grassroots organizations wanting to apply such advanced data analysis techniques (Pimm et al. 2015, Johnson et al. 2018). Despite such challenges, conservation NGOs are required by government agencies and donors to provide viable evidence of effective and efficient use of funds (Salafsky et al. 2001, McKinnon et al. 2015). Therefore, conventional methods of estimating wildlife densities not only require satisfying the nuanced assumptions of the analytical approach that ensure robust and reliable estimates, have strong study designs that meet aspects of replication and randomization, and need to be made within the constraints of financial and technical resources available to researchers (Foster and Harmsen 2012, Mathai et al. 2013).

In order to overcome such difficulties inherent in using conventional and contemporary wildlife survey methods, conservationists and wildlife managers have used social surveys as a complimentary tool to study wildlife populations (White et al. 2005). In addition, researchers have called for incorporating local knowledge of communities living close to wildlife in tropical biodiversity conservation (Sheil and Lawrence 2004, Gilchrist et al. 2005, Bawa et al. 2006, Danielsen et al. 2014). Though once considered anecdotal, leveraging locality specific knowledge in resource management and ecosystem conservation are now used to guide management of various conservation programs (Drew 2005, Gilchrist et al. 2005, White et al. 2005, Rodas-trejo et al. 2008, Sandbrook et al. 2013, Peñaherrera-Palma et al. 2018, Needham et al. 2020). Rodas-trejo et al. (2008) combined systematic boat transect surveys and underwater sonar scan recordings with interviews with local fishermen to piece together a comprehensive picture of the distribution of manatees in the Catazajá wetland system in Mexico. Fishermen were shown maps to indicate where they sighted manatees and asked supplementary questions that were recorded using a semi-structured interview guide. Conversely, Meijaard et al. (2011) used a structured questionnaire survey across 687 villages to assess the underlying causes and the distribution of threats to Bornean orangutans and suggest that social factors related to human orangutan conflicts could only be identified using social surveys. A more hands off social survey method known as 'citizen science' was used by Datta et al. (2018) to collect information on hornbill species by engaging citizens across India to provide data through an interactive web interface. Despite concerns about data quality and reliability, citizen science methods allows for a financially viable way for big data sets to be generated that span over a long time and a large geographical area (Galloway et al. 2006, Silvertown 2009, Lukyanenko et al. 2016, Schewe et al. 2020). Equally potential benefits of citizens include educational value, the potential to raise awareness in conservation related issues, and facilitate stakeholder engagement in natural resource management through 'creation of trust and mutual respect' between resource users and resource managers enabling co-management (Goffredo et al. 2010, Schewe et al. 2020). Social surveys are also used to gather information on wildlife species from indigenous communities (Gilchrist et al. 2005). By using structured interview guides, Gilchrist et al. (2005) gathered data on less studied Arctic bird species by leveraging the local ecological knowledge (LEK) of Inuit communities. Gilchrist et al. (2005) uses the definition of LEK provided by Berkes et al. (2000) which is 'a cumulative body of knowledge, practice and belief, evolving by adaptive processes and handed down through generations by cultural transmission. It deals with the relationship between living beings (including humans) with one another and with their environment'. According to Gilchrist et al. (2005), in order to reduce variability and increase confidence in the collected information, an adequate sample of respondents need to be interviewed and data interpretation need to be made in light of communities relationship with the animal species. Gilchrist et al. (2005) were able to detect dramatic population

declines that went unreported through conventional methods of wildlife monitoring. Needham et al. (2020) combined tacit local knowledge with data driven wildlife movement pathway models to develop detailed plan for wildlife corridor between Nova Scotia and New Brunswick, Canada. They used a combination of individual interviews and focus group mapping workshops with local knowledge holders such as 'subsistence harvesters, woodlot owners, farmers, naturalists and recreation users' to gather their experiential knowledge on wildlife species and movement patterns. The authors concluded that findings from their study not only enriched formal data models by providing an explanatory context, but also helped to build local support for conservation measures. These examples indicate that harnessing the knowledge of local communities about their surrounding natural resources allow researchers to gather information that would otherwise be difficult to obtain using conventional methods (Drew 2005, Gilchrist et al. 2005, Hoeven 2007, Schewe et al. 2020).

Involving local communities and combining alternative knowledge systems with scientific methodologies has the potential to provide success in conservation and restoration efforts (Bawa et al. 2006, Needham et al. 2020, Schewe et al. 2020). Since empirical knowledge about the environment is often incomplete due to logistical limitations in data collection throughout the year, observations and knowledge of local communities about their surrounding natural resources provides an underutilized opportunity to identify species and areas that need management attention (Gilchrist et al. 2005, Danielsen et al. 2014). Collaboration with local communities also creates the scope to engage such communities in the conservation process and promote their commitment to continued action (Sheil and Lawrence 2004, Bawa et al. 2006, Needham et al. 2020, Schewe et al. 2020). Participation has been identified as one of the key aspects that can lead to the success or failure of community based conservation initiatives (Berkes 2007). Involving communities in every step of the process from design to evaluation can greatly improve conservation outcomes. It also has the potential to encourage engagement of local communities in the management of their natural resources and provides new opportunities for effective conservation (Sheil et al. 2005). However, before engaging community members into action related to resource conservation, aspects such as knowledge of individuals about the natural resource, cost effectiveness and methodology of assessing local knowledge, and monitoring of the effectiveness of the participatory approach need to be carefully considered to achieve success (Castello et al. 2010). Nevertheless, involving local community members in conservation research opens up avenues of discussion that moves beyond mere data collection and engages research participants into thinking about changes in the state of their surrounding natural resource, and encourages them to become involved in conservation action (Gilchrist et al. 2005, Rodas-trejo et al. 2008).

Among the various methods used to study wildlife with support from local communities, Hoeven et al. (2004) used the 'pooling local expert opinion' (PLEO) method to estimate wildlife densities of 33 species of animals within the area of the Campo Ma'an Management and Biodiversity Conservation Project, located in the southwest corner of Cameroon, Africa. The PLEO method was developed based on the Delphi method that combines the 'consensus of opinions of a group of experts' on a specific subject matter where precision is lacking (Dalkey and Helmer 1963, Prins and Wind 1993, Hoeven 2007). Hoeven et al. (2004) used the PLEO method to pool together subjective opinions of local hunters and use group judgements to estimate wildlife population densities for their project site. This allowed them to tap into the collective knowledge of hunters on the abundance and distribution of specific local wildlife, which hunters amassed through their hunting trips. Hoeven et al. (2004) compared their estimates with other studies carried out in and around their study area and found that, out of the 85 species surveyed using the PLEO method, 42 species had a slightly lower density estimate and 43 species a slightly higher density estimate compared to results from the 'traditional' line transect methods. Additionally, all of their PLEO based densities fell within the 95% confidence interval of estimates from the 'traditional' line transect methods. Hoeven et al. (2004) also tested their PLEO results by correlating PLEO based density estimates with body size of the selected species, showing a linear negative correlation between body size and densities, i.e. large body size correspond with low densities. Another study used the PLEO method to determine the patterns in the sighting of live and dead manatees and substantiated hypotheses that informed manatee conservation efforts in Lower Sanga Basin, Cameroon (Mayaka et al. 2015). Both studies concluded that compared to conventional methods of estimating wildlife parameters, the relatively novel PLEO approach 'reduces costs, time constraints and dependence of outside expertise' as well as allows researchers to gather sufficient data enabling opportunity for extrapolation of results over larger landscapes (Hoeven 2007, Mayaka et al. 2015).

Given the need to assess wildlife population densities reliably, quickly and cost effectively, supplementing transects and other conventional methods with local indigenous expert knowledge and observations presents a unique opportunity. Taking advantage of our unique position of working with Dayak communities at our project site, we implemented the PLEO methodology to determine the densities of various wildlife species that inhabit our project sites. By leveraging the collective knowledge of selected individuals from Dayak communities inhabiting our project site, this paper examines the reliability and effort of applying the PLEO method, as described by Hoeven et al. (2004), to estimate wildlife population densities in one of the last major forested areas outside the 'Heart of Borneo' central forest spine, in West Kalimantan, Indonesia. In doing so, we carried out a survey based on the PLEO methodology and compared results for selected species with population density estimates derived using our point count surveys at the same location, and with comparison studies carried out in other sites on the island of Borneo.

Material and methods

Study site

This study was carried out in the Gunung Niut Nature Reserve (GNNR), situated on the northwestern part of West Kalimantan, Indonesia (Fig. 1). GNNR covers an area of 91

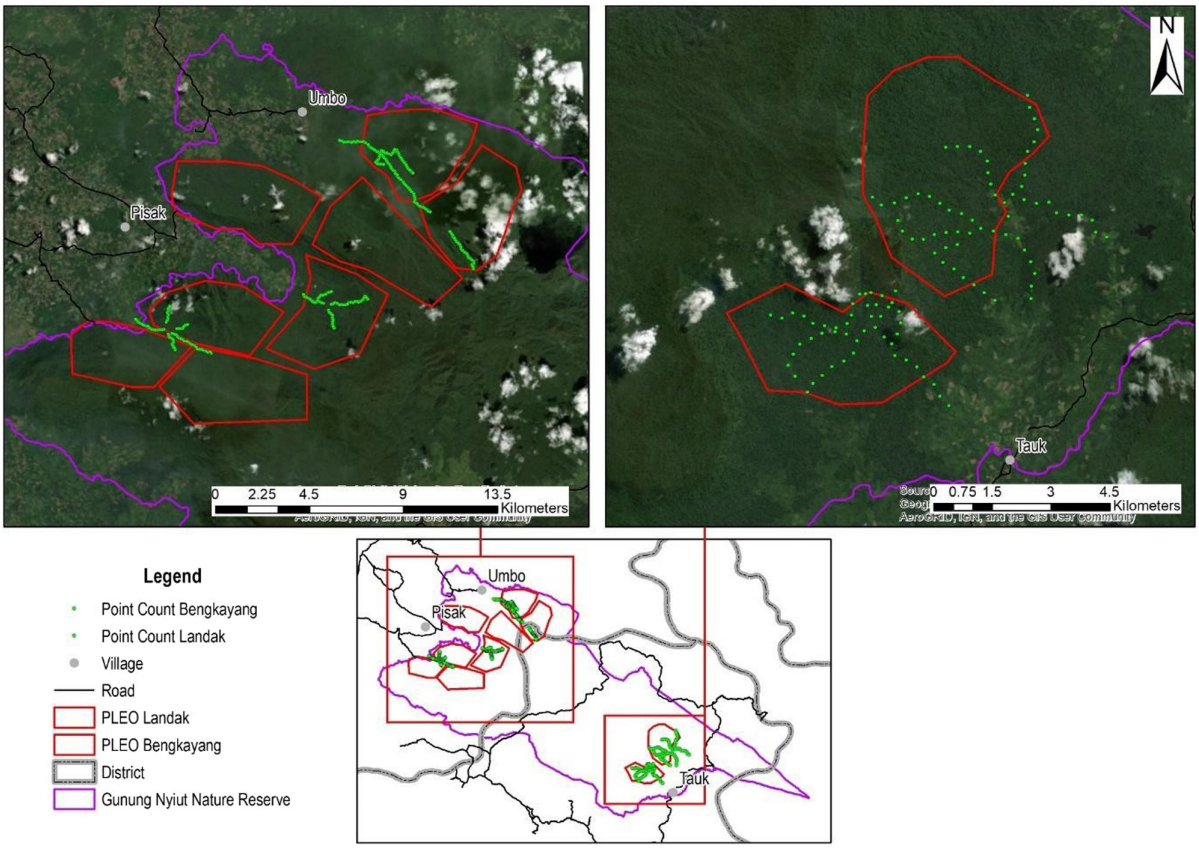

Figure 1. Map of the study area depicting area covered by PLEO survey and corresponding locations of point transect survey in the Gunung Niut Nature Reserve, West Kalimantan, Indonesia.

759 ha (917 km^2) of reserve forest and 60 815 ha (608 km^2) of 'protection' forests. The study area is comprised of sub montane (upland) and some lowland rainforest at roughly 100 m a.s.l. The tallest peak within the reserve, Mt Niut, is at 1700 m a.s.l. The six mountains with peaks above 1000 m and differences in terrain give rise to high levels of variation in forest type from lowland to cloud forests in GNNR. Though the overall management of GNNR is under the responsibility of the Balai Konservasi Sumber Daya Alam (BKSDA) of West Kalimantan, indigenous communities living around GNNR are allowed to harvest non-timber forest products under Indonesian law. In 1984, the area was gazetted to protect three of West Kalimantan's most important watersheds (Landak, Sanggau and Sambas rivers) that originate in the reserve.

Sample units selected for the PLEO survey were derived from the spatial, monitoring and reporting (SMART) tool used by Yayasan Planet Indonesia (YPI) to carry out field patrols at the project site. Sample units ranged between 5 km^2 and 20 km^2 and were defined by well-known features of the landscape such as waterfalls, mountain peaks, rivers, logging roads and hunting trails to help local hunters familiarize with the sample area during data collection. In total, the ten (10) sampling units were selected that covered a total of 167 km^2 or 18% of the total area of GNNR. Participants selected for the PLEO survey were from 3 Dayak villages representing the study area shown in Fig. 1.

Dayak communities, a blanket term used to describe over 300 indigenous groups living in Borneo, are the primary inhabitants of our project site. They are forest dwelling people whose lives, culture and survival are interwoven with their surrounding forests (Crevello 2004). Due to being forest inhabitants, Dayak communities are known to have a rich understanding of their natural surroundings that has allowed them to harvest forest resources sustainably over generations (Joshi et al. 2004, Mulyoutami et al. 2009). As a result, hunting of wildlife is a long standing Dayak tradition carried out to meet various needs including food, medicine, sport and 'pest' control (Wadley et al. 1997, Struebig et al. 2007).

Point transect surveys were originally created using randomized transects in the eastern part of the reserve. However, due to extremely difficult terrain spanning from rock walls, large rivers, to unsurpassable cliffs randomly selected transects and points were not able to be used. Field teams then worked with local communities to identify and map out old hunting trails. Trails and subsequent points were then verified to cover variation in elevation and forest types and each point transect survey route contained between 8 and 10 points 300 m apart.

Data collection

PLEO survey

Fifty-eight persons across the four villages participated in the survey between April and May 2019. Both community members (particularly hunters) and local community SMART patrol unit members were included. Village lead-

ers were consulted to gain access to an initial set of participants from each village who were hunters and involved in resource extraction for more than five years in a given sample area. Adhering to the ethical principles of informed consent, issues related to confidentiality of the study participant, use of the data, objective of and the time requirement to carry out the survey was explained to village leaders during selection of study participants as well as to each study participant before commencing data collection (Hammersley and Atkinson 2007). After data was collected from the initially identified participants, assistants of the YPI monitoring team used the chain-referral method to identify additional suitable participants in the village until no new participant names emerged (Newing et al. 2011). On average, the individual meetings lasted between 15 and 20 min, which helped us to keep participant dropout rates to zero. Additionally, the fact that study participants were selected using a chain referral technique and as assistants facilitated the surveys, we were able to reach villagers who closely resembled the target population as well as achieve high response rates for the survey.

After verbal agreement, one of the two assistants collected data by personally meeting and talking with each participant following a set procedure using a semi-structured questionnaire. The first step in the procedure required each assistant to use a map to delineate the sample area by showing features of the landscape (e.g. waterfalls, logging roads, etc.) to the participant. Once a participant approved his familiarity with the sample area, the assistant then proceeded with showing pictures of each species with their local names from a prepared booklet to avoid misidentification of species. Once a participant was able to identify a species correctly, the assistant then asked the participant to provide an estimate for number of individuals of that species that inhabit the sample area. For example, once a participant was able to identify the Sunda clouded leopard, the participant was asked if they think there were more than 100 or less than 100 Sunda clouded leopards inhabiting the given sample area. If they stated less than 100, then a follow up question was asked whether this number was more than 50 or less than 50 individuals in the given area, and so on. Following Hoeven et al. (2004), when participants provided ranges e.g. between 20 and 40, we used the mid-points to carry out the analysis. In cases where participants were not able to identify a species from the booklet, we estimated the population to be '0' in that area based on the assumption that the particular species was not seen within the given area. This procedure was carried out for each of the 36 species (Table 1) for all the 58 study participants' sample areas.

Following the method used by Hoeven et al. (2004), density estimates were calculated by dividing each participant's estimate range for number of individual animals per species by the corresponding surface area of that given sample unit. This resulted in 58 density estimates for each species (from 58 individual meetings), which was then pooled into one density estimate per species for the entire study area. As densities are indices, geometric means were calculated for each species for the entire study, which were then root-transformed to account for overestimates and provide a normal distribution (Hoeven et al. 2004). Finally, density estimates for each species for the entire study area (Table 1) was calculated after re-transforming the mean density estimates of each species (Hoeven et al. 2004). Therefore, in order to derive density estimates for the 36 species (95% confidence intervals) for our study, we followed step-by-step process enunciated by Hoeven et al. (2004). The scientific nomenclature and taxonomic order are based on the IUCN Red list classification.

Point transect survey and conventional distance sampling

As YPI carries out point transect surveys as part of a regular monitoring activity in the GNNR, we used data from point transect surveys conducted between October 2018 and February 2019 for density estimate comparisons for this study. In case of the point transect survey, two teams of two trained observers, one local guide, and sometimes an accompanying BKSDA employee, carried out the point transect survey along 14 established trails totaling approximately 39 km. These surveys were conducted for 12–16 days each month, where each team collected data between 05:30 a.m. and 10:00 a.m. at every 300 m intervals with 10 min count periods. Once arriving at an interval, observers in the survey team recorded the time, date and weather condition after which they sat silently for 5 min before beginning data collection. Both visual and auditory detection methods were used and all point transect data were recorded digitally with detected individuals being logged into predetermined distance bins of 0–25, 26–50, 51–75, 76–100, >100 m. Species that were not identifiable were recorded as unknown individuals and only the time relative to the start of the point count was recorded.

Conventional distance sampling (CDS) methods described by Buckland et al. (2004, 2015) were followed to analyze the point transect data. Analysis was split into two parts: 1) fitting a detection function for $g(x)$, where 'x' is the observed distance in meters (m) from the center of the point to the sighting used to estimate average probability of detection, 'pi'; and 2) estimating animal density D using the formula:

$$\hat{D} = \frac{n}{k\pi w^2}$$

where 'k' is the number of point counts, 'w' is the truncation distance and 'n' is the number of individuals seen within 'w'. Estimation of detection probability was performed using free statistical software R (<www.r-project.org>), ver. 3.5 with the packaged DISTANCE software (Miller et al. 2019). Although field teams collected distance data at multiple distances, all observations were truncated to 100 m to the standard radius of a point transect. The top model was then selected based upon the Akaike information criterion (AIC) as described further in Buckland et al. (2004, 2015) (Table 2, Fig. 2, 3).

Results

PLEO survey

Based on PLEO survey results, the two most abundant species were the pig-tailed macaque *Macaca nemestrina* ($\hat{D}$ = 26.75 (14.71–38.79)) and the long-tailed macaque

Table 1. Density estimates, coefficient of variation and 95% confidence intervals for thirty-six species identified in our study area and used in this study to evaluate the PLEO methodology. Density estimates provided are individuals/km^2, except for felid speices (marked *), which are individuals/100 km^2.

Common name	Latin name (status)	Density	CV%	Lower 95%–upper 95%
Southern pig-tailed macaque	*Macaca nemestrina* (endangered)	26.75	0.22	14.71–38.79
Nicobar crab-eating macaque	*Macaca fascicularis* (vulnerable)	23.63	0.24	12.11–35.14
Greater green leafbird	*Chloropsis sonnerati* (endangered)	14.51	0.23	7.77–21.25
Long-tailed porcupine	*Trichys fasciculata* (least concern)	14.26	0.24	7.43–21.09
Malayan porcupine	*Hystrix brachyura* (least concern)	13.42	0.26	6.55–20.28
Bearded pig	*Sus barbatus* (vulnerable)	12.04	0.22	6.79–17.28
Pale giant squirrel	*Ratufa affinis* (near threatened)	9.91	0.17	6.46–13.36
White-rumped shama	*Kittacincla malabarica* (least concern)	7.39	0.25	3.68–11.11
White-crowned hornbill	*Berenicornis comatus* (endangered)	7.18	0.38	1.74–12.61
Tufted ground squirrel	*Rheithrosciurus macrotis* (vulnerable)	6.65	0.20	3.98–9.33
Sunda flying lemur	*Galeopterus variegatus* (least concern)	6.06	0.16	4.09–8.02
Bushy-crested hornbill	*Anorrhinus galeritus* (near threatened)	5.77	0.24	3.04–8.51
Sunda pangolin	*Manis javanica* (critically endangered)	5.54	0.31	2.05–9.02
Horsfield's tarsier	*Tarsius bancanus* (vulnerable)	5.02	0.18	3.19–6.85
Greater oriental chevrotain	*Tragulus napu* (least concern)	4.67	0.24	2.39–6.96
Wreathed hornbill	*Rhyticeros undulatus* (vulnerable)	4.51	0.36	1.27–7.75
Greater slow loris	*Nycticebus coucang* (vulnerable)	4.18	0.19	2.59–5.78
Rhinoceros hornbill	*Buceros rhinoceros* (vulnerable)	4.00	0.33	1.33–6.68
Oriental pied hornbill	*Anthracoceros albirostris* (least concern)	3.85	0.32	1.35–6.35
Great argus	*Argusianus argus* (near threatened)	3.56	0.24	1.86–5.27
Black hornbill	*Anthracoceros malayanus* (vulnerable)	3.41	0.14	2.45–4.36
Southern red muntjac	*Muntiacus muntjak* (least concern)	3.17	0.15	2.18–4.15
Oriental magpie-robin	*Copsychus saularis* (least concern)	3.06	0.19	1.89–4.23
Red langur	*Presbytis rubicunda* (least concern)	2.96	0.20	1.76–4.16
Silvery lutung	*Trachypithecus cristatus* (near threatened)	2.49	0.20	1.46–3.51
Marbled cat*	*Pardofelis marmorata* (near threatened)	2.26	0.23	1.20–3.32
Leopard cat*	*Prionailurus bengalensis* (least concern)	2.13	0.26	1.02–3.24
Helmeted hornbill	*Rhinoplax vigil* (critically endangered)	1.91	0.20	1.13–2.70
Sambar	*Rusa unicolor* (vulnerable)	1.90	0.23	1.04–2.77
Abbott's gray gibbon	*Hylobates abbotti* (endangered)	1.70	0.16	1.13–2.26
Hill myna	*Gracula religiosa* (least concern)	1.69	0.24	0.88–2.49
Sunda clouded leopard*	*Neofelis diardi* (vulnerable)	1.35	0.30	0.52–2.18
Bornean bay cat*	*Catopuma badia* (endangered)	1.15	0.41	0.20–2.10
Sun bear	*Helarctos malayanus* (vulnerable)	0.32	0.38	0.09–0.56
Straw-headed bulbul	*Pycnonotus zeylanicus* (critically endangered)	0.26	0.31	0.10–0.43
Bornean orangutan	*Pongo pygmaeus* (critically endangered)	0.0034	0.59	0.001–0.01

Macaca fascicularis ($\hat{D}$ = 23.63 (12.11–35.14)) (Table 1). On the other hand, the two species with the lowest PLEO-based density estimates were the bornean orangutan *Pongo pygmaeus* ($\hat{D}$ = 0.0034 (0.001–0.001) and straw-headed bulbul *Pycnonotus zeylanicus* ($\hat{D}$ = 0.26 (0.10–0.43) (Table 1). We pooled coefficients of variation (CV) for uncertainty and compared mammal density estimates with bird density estimates. Averaged CVs revealed that mammal estimates were far less variable (CVs = 0.136) compared to bird species (CVs = 0.467). Furthermore, we found that our PLEO density estimates for the 13 species selected for comparison were within the range of estimates calculated in studies using line transects, grids, point counts and camera traps survey methods.

Table 2. Models derived using point transect data for rhinoceros hornbill (RH), wreathed hornbill (WH), bushy-crested hornbill (BC), black hornbill (BH), helmeted hornbill (HH) and white-rumped shama (WRS).

Species	n	Model	K–S p	p	%CV (p)	D (LCI–UCI)	%CV(D)
Rhinoceros hornbill (RH)	106	Hn	0.33	0.845	0.18	4.95 (4.25–7.54)	0.211
	51	Hn	0.42	0.738	0.25	5.41 (2.93–9.99)	0.316
Wreathed hornbill (WH)	38	Unif+cos(1)	0.40	0.413	0.2	3.34 (1.8–6.18)	0.318
	25	Unif+cos(1)	0.16	0.342	0.38	5.24 (1.95–14.03)	0.52
Bushy-crested hornbill (BC)	41	Unif+poly(2)	0.86	0.5	0.13	4.92 (3.1–7.82)	0.238
	16	Unif+cos(1)	0.02	0.471	0.28	4.08 (1.67–9.96)	0.469
Black hornbill (BH)	32	Unif+cos(1)	no df	0.56	0.25	3.17 (1.65–6.09)	0.337
	16	Hn	0.25	0.54	0.38	3.25 (1.23–8.61)	0.51
Helmeted hornbill (HH)	38	Unif+cos(1)	0.79	0.47	0.19	2.3 (1.38–3.85)	0.262
	16	Hn	0.62	0.59	0.43	1.54 (0.58–4.11)	0.51
White-rumped shama (WRS)	60	Unif+herm(4)	0.61	0.47	0.06	6.95 (3.09–11.21)	0.246

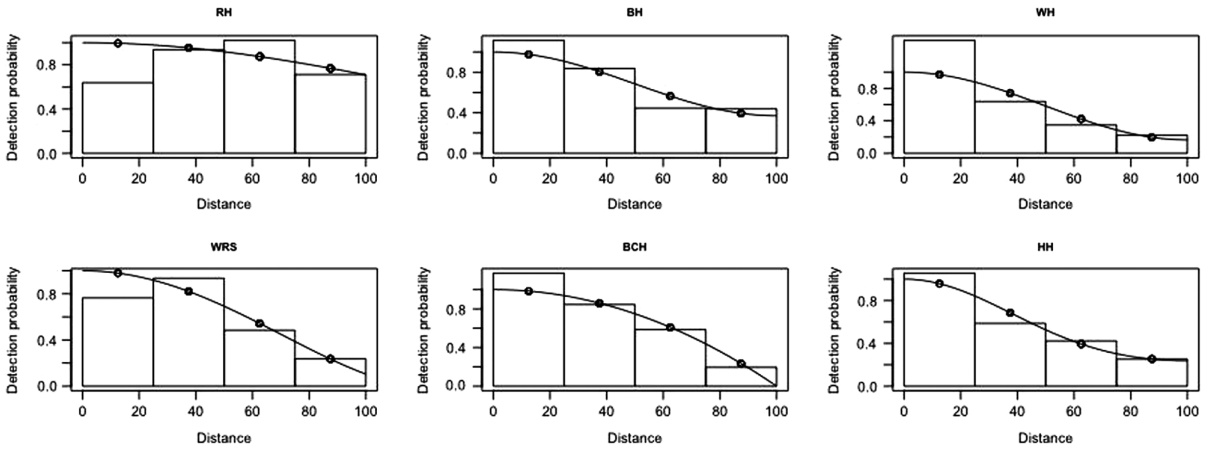

Figure 2. Detection functions ($g(x)$) for the rhinoceros hornbill (RH), black hornbill (BH), wreathed hornbill (WH), white-rumped shama (WRS), bushy-crested hornbill (BCH), helmeted hornbill (HH) included in the CDS analysis.

Distance sampling analysis

Densities and associated measures of uncertainty were estimated using CDS methods described in Buckland et al. (2004, 2015). A central piece of distance sampling is the detection function model. Buckland et al. (2004, 2015) provide a set of criteria for judging the utility of candidate model classes. Detection function models should be:

1. flexible, so that they can take a wide variety of shapes;
2. efficient, in the sense that many plausible shapes can be represented using few parameters;
3. flat at zero distance (i.e. $g'(0; \theta) = 0$), indicating that objects in the immediate vicinity of the observer are equally detectable; and,
4. monotonic non-increasing with increasing distance (i.e. $g'(y; \theta) \leq 0$ for $0 < y \leq w$), as it is typically unrealistic for objects to become more detectable with increasing distance.

The key function plus series adjustment semiparametric (K + A) modelling approach developed by Buckland et al. (2004, 2015) was used. The uniform, half-normal, hazard-rate, negative exponential key functions were used together with simple polynomial, cosine and hermite polynomial expansion terms as required. Models were fitted to the observed distribution of distances based upon key functions and series expansion formulas. The model that minimized Akaike information criterion (AIC) was selected for analysis. Using the key function and series adjustments, detection functions were fitted to observations to estimate densities (Fig. 2, 3). We included both density functions (Fig. 2) and probability density functions (Fig. 3), as models for point transects under CDS are often fit using probability density functions (pdfs) and standard density functions (dfs) (Buckland et al. 2004, 2015).

Based on our pdfs, models appeared to fit best for white-rumped shama (WRS) ($p = 0.06$) and bushy-crested hornbills (BCH) ($p = 0.28$), while other species suggested a lack of model fit (Fig. 3). As the study area was not stratified and the majority of species in Borneo are cryptic and exist in low densities, the number of detections for some species were limited (e.g. often less than 60), which caused limitations in model convergence. Additionally, we argue that using counts from 300 m intervals for hornbill species may lead to double counting as hornbills have large ranges and frequently move

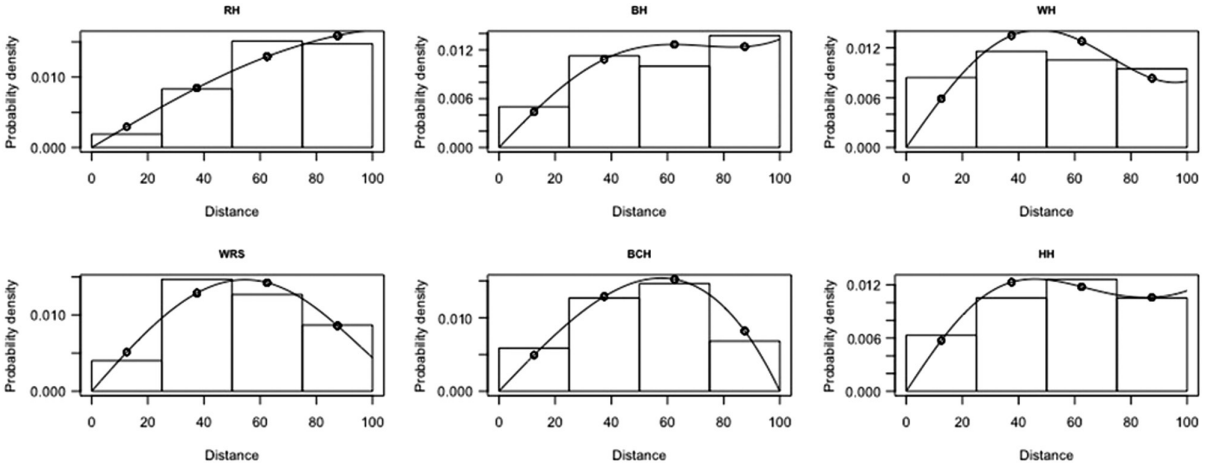

Figure 3. Probability density functions ($f(x)$/pdfs for the rhinoceros hornbill (RH), black hornbill (BH), wreathed hornbill (WH), white-rumped shama (WRS), bushy-crested hornbill (BCH), helmeted hornbill (HH) included in the CDS analysis

throughout their range (Marsden 1999, Gale and Thongaree 2006). Therefore, we conducted a second distance analysis with half the observations by removing every other point along each transect. This removal was not necessary for the only species of songbird included in our analysis, the white-rumped shama (Table 3).

Analysis of results and methods – a comparison

The following sections provide an analysis of density estimates and appraisal of the survey effort associated with the PLEO study, point transect survey and comparison studies. We compared our density estimates with those from other studies carried out in the island of Borneo. In line with process followed by Hoeven et al. (2004) to check for consistency between their PLEO based density estimates and those using traditional methods, the primary objective of comparing our results with those of comparison studies was to look for potential outliers or extreme differences between our PLEO density estimates and density estimates from comparison studies. Due to a complete lack of studies on density estimate for species that occur in GNNR, we restricted choosing only studies carried out on the island of Borneo. Since it was also not possible to find density estimate studies for all the species included in our study, we were only able to compare our PLEO results with six comparison studies covering 12 out of 36 species covered by our PLEO method.

Density estimates

We selected seven hornbill species, one songbird species, two primate species and three felid species that occur in our study area to compare our PLEO based density estimates with estimates from our point transect survey and estimates found in available literature (Table 3, 4). We chose these species because they are often the focus of wildlife studies in Bornean rainforests and are particularly difficult to survey due to their scarcity, cryptic nature and high level of mobility (Marshall 2010, Cheyne et al. 2013, 2016). The comparison studies chosen for our analysis include studies carried out in mixed dipterocarp forests of varying elevations in Borneo, using conventional techniques used to estimate wildlife densities for species selected for our analysis (Table 3, 4). Although various factors likely differ spatially and temporally between our study site and comparison study areas, nevertheless comparisons with selected external studies provided the opportunity to discuss the caveats of using different methods in studying wildlife populations, especially in tropical forests.

Among the seven hornbill species, only five species were comparable between the PLEO based density estimates and point transect density estimates, since the latter method did not return enough data points to determine densities for the oriental pied and white-crowned hornbill species. In the case of these five species, comparisons show that PLEO density estimates were all within the range of estimates derived from the point transect survey (Table 3). Specifically, PLEO based estimates were slightly higher for three species (helmeted hornbill, black hornbill and bushy-crested hornbill) and slightly lower for two species (rhinoceros hornbill and wreathed hornbill). Alternatively, the comparison study McConkey and Chivers (2004) used line transects and DISTANCE software to carry out their analysis of data collected from the lowland dipterocarp forests of the Bario Ulu landscape in Central Kalimantan. Comparing their results with the PLEO based range of estimates revealed that three hornbill species' density estimates were within the PLEO based range of estimates (rhinoceros hornbill, black hornbill and bushy-crested hornbill), while the remaining four species estimates were lower than the PLEO based range of estimates range of estimates (helmeted hornbills, oriental pied hornbills, wreathed hornbills, white-crowned hornbill) (Table 3).

In case of the white-rumped shama, the sole songbird species in our analysis, the PLEO based estimates were within the range of estimates derived from the point-transect survey method. In particular, the point transect survey results for this species presents the lowest p-value (0.06) compared to any other species in our comparison list. However, we were unable to compare our results with external studies due to a lack of such studies for this species in Indonesia at the time of our analysis.

The comparison studies for the two primate species, the red langur and the silvery lutung, used line transects to collect data, with one study following distance analysis method for data analysis (Marshall 2004), and the other study using

Table 3. Comparison of PLEO based estimates, point transect estimates and density estimates from McConkey and Chivers (2004) for seven hornbill species. McConkey and Chivers (2004) research was carried out in the Barito Ulu, Central Kalimantan. McConkey and Chivers (2004) used line transects and DISTANCE to derive their density estimates.

Species	PLEO density estimate (ind km^{-2})	Point transect density estimates (ind km^{-2})	Comparison study density estimate (ind km^{-2})	Source
Helmeted hornbill	1.91 (1.37–3.29)	1.54 (0.58–4.11)	0.7	McConkey and Chivers (2004)
Rhinoceros hornbill	4.00 (1.29–10.12)	5.41 (2.93–9.99)	3.7 ± 1.2	McConkey and Chivers (2004)
Black hornbill	3.41 (2.49–4.83)	3.17 (1.28–6.38)	3.4	McConkey and Chivers (2004)
Oriental pied hornbill	3.85 (1.28–7.57)	–	0.1	McConkey and Chivers (2004)
Wreathed hornbill	4.51 (1.6–9.69)	5.24 (1.95–14.03)	0.1	McConkey and Chivers (2004)
Bushy-crested hornbill	5.77 (3.33–10.10)	4.08 (1.67–9.96)	5.5	McConkey and Chivers (2004)
White-crowned hornbill	7.18 (2.11–15.74)	–	0.3	McConkey and Chivers (2004)
White-rumped shama	7.39 (3.68–11.11)	6.95 (3.09–11.21)	–	–

Table 4. Comparison of PLEO based estimates with density estimates from other studies for three primate species and three felid found in our study area. We found that our estimates were within the 95% confidence interval for all six species.

Species	PLEO density estimate (ind. km^{-2})	Comparison study density estimate (ind. km^{-2})	Comparison study survey method	Comparison study site	Source
Red langur *Presbytis rubicunda*	2.96 (1.76–4.15)	6.89 ± 1.76	Line transect, DISTANCE	Gunung Palung (upland forest)	Marshall 2004
Silvery lutung *Trachypithecus cristatus*	2.49 (1.43–3.51)	1.17 ± 2.37	Line transects, Mann Whitney test	Menanggul River, Sabah, Malaysia	Matsuda et al. 2016
Sunda clouded leopard *Neofelis diardi*	1.35 (0.52–2.18)/100 km^2	1.9 (0.7–5.4)/100 km^2	Camera trapping, SECR	Maliau Basin Conservation Area, Sabah, Malaysian Borneo	Brodje and Giordang 2012
Marbled cat *Pardofelis marmorata*	2.26 (1.20–3.32)/100 km^2	7.1 (3.81–10.73)/100 km^2	Camera trapping, SECR	Tawau, Sabah, Malaysia	Hearn et al. 2016
Leopard cat *Prionailurus bengalensis*	2.13 (1.02–3.24)/100 km^2	9.6 (SE ± 1.7)/100 km^2	Camera trapping, SECR	Deramakot Forest Reserve, Sabah, Malaysia	Mohamed et al. 2013

the Mann–Whitney U-test as it compared data collected using different methods (Matsuda et al. 2016) (Table 4). Our PLEO density estimates for these primate species were either lower (red langur) or higher (silvery lutung than the range of estimates of these comparison studies.

In the case of felid species, all comparison studies used camera-traps to collect field data and carried out data analysis following the SECR method, albeit using various software packages (Table 4). Except for the Sunda clouded leopard, PLEO based density estimates for the marbled cat and leopard cat species were lower than the range of estimates found in the comparison studies. The comparison study for the Sunda clouded leopard was carried out in a mosaic of primary mixed dipterocarp forest, heath forests and logged forests (Brodie and Giordano 2012). In the case of the marbled cat comparison study, the study area consisted of primary dipterocarp forests of mixed elevations (Hearn et al. 2016). For the leopard cat comparison study, camera traps were used to study differences in leopard cat population across three adjacent sites, from which density data from only one site (Deramakot Forest Reserve) was used for comparison with this study (Mohamed et al. 2013).

Survey effort

In analyzing survey effort, we compared the effort required to carry out such surveys to estimate wildlife densities using the different methods highlighted in the previous section ('Density estimates' section). In terms of survey effort, McConkey and Chivers (2004) established three trails that totaled 10 km, which were walked twice a month for six days each from October 1996 to September 1997. Perpendicular distances from trail to animal sightings were estimated by eyesight and corrected using mean error method. Regarding effort expended to carry out field research for the primate comparison studies, Marshall (2010) took two years to collect data, taking support from three local individuals to collect data, who along with him walked a total of 1374 km to cover 409 census routes that were on average 3.5 km long. For the silvery lutung primate species, the comparison study was carried out on foot transect surveys by two observers and a research assistant across 16 trails totaling to 24 census over a 12 month period (Matsuda et al. 2016). The comparison study for the Sunda clouded leopard used 25 camera trap stations to obtain 59 detections identifying only four distinct individuals over a period of four months between January and May 2010 (Brodie and Giordano 2012). For the marbled cat comparison study, researchers used 77 camera stations covering around 149 km^2 and identified 73 individuals from data collected over October 2012 to December 2013 (Hearn et al. 2016). For the leopard cat comparison study, 47 camera trap stations were used that covered an area of 123 km^2 of the selected study sites, allowing researchers to identify 23 distinct individuals over a period of four months between September 2008 and January 2009 (Mohamed et al. 2013). In terms of time taken to carry out research, comparison studies show that camera trap surveys may require four months to 14 months to collect field data, whereas transect surveys likely require one year or more of repetitive data collection to provide sufficient data to enable robust analysis.

In case of our PLEO study, we completed all tasks related to data collection within four months that included training two assistants from YPI monitoring team who conducted 58 individual meetings with villagers located in four remote villages near GNNR. Assistants were provided seven days of orientation on the study objective along with use of the semi-structured questionnaire and the survey protocol. In contrast, our point transect surveys, which took four highly trained observers able to identify auditory calls and visual signs of various species, a total of 70 days over a period of five months to collect the point count data used in this study. An additional three months were used to review audio recordings and cross check species logs by team members to verify species identified during the survey. Though the point transect survey is part of regular monitoring activity of YPI, comparing financial costs would suggest that transect surveys are approximately 10 times more expensive than a PLEO survey.

Discussion

The purpose of this study was to compare the reliability and effort required to carry out the PLEO method compared to conventional methods used in estimating wildlife population

densities in tropical forests. We compared our PLEO based density estimates with results from our point transect surveys as well as density estimates from extant literature on selected species. Comparison between PLEO based estimates and point transects revealed no statistically significant difference in density estimates. However, precaution should be taken in comparing our estimates to other studies due to differences in habitat type and survey effort. By following the method of comparing results used by Hoeven et al. (2004), our comparisons were intended to look for extreme differences and outliers in our analysis that could indicate the PLEO methodology is inaccurate and produces potentially misleading results. We also found that the PLEO methodology directly improves participation and engagement with local communities in wildlife management and conservation. This could be a major strength of PLEO, as enhanced participation has been identified as one of the important themes leading to successful community-based conservation initiatives (Berkes 2007). In the following section, we provide a nuanced discussion to highlight some of the advantages and drawbacks of the PLEO method in contrast to widely accepted methods used to determine density estimates for different wildlife species of tropical forests and its potential role in engaging local communities in natural resource management.

While Hoeven et al. (2004) only focused on using the PLEO method to determine density estimates for mammal species, we expanded this to also include a selected number of both mammal and bird species that inhabit our project area. This allowed us to compare whether there were differences in precision between density estimates between the two taxa when using the PLEO method. Comparing averaged coefficients of variation (CV) between density estimates of the two taxa revealed a higher level of uncertainty for bird species compared to the mammal species. This may be due to multiple reasons that include response bias, differences in ecology between the taxa, and inconspicuousness due to size and vocalizations of bird species compared to mammal species As some participants were hunters of game species and may be involved with poaching important bird species (e.g. white-rumped shama) to supply the illegal wildlife trade, there may have been a tendency to respond inaccurately for protected bird species in fear of repercussion. Another reason for higher uncertainty may be due to birds having more variability in their movements compared to mammals. As an example, hornbills are known to fly great distances every day to forage (Poonswad 1998). As some bird species are inconspicuousness due to their size and vocalizations they are often difficult to identify, which may be another reason for such higher level of uncertainty between the taxa (Whitworth et al. 2016, Fontúrbel et al. 2020). While indigenous forest-dependent communities hold an intimate knowledge of their surrounding flora and fauna to the point of ecologically explaining importance of certain species for the ecosystem, they are nevertheless susceptible to misidentification (Silva de Oliveira and Dario 2018). Therefore, response bias, behavioral and physical characteristics of particular avian species may have led to a higher variation in bird species estimates compared to mammalian species estimates using the PLEO method.

In the case of five of the seven hornbill species there were no statistically significant differences between CDS and PLEO based density estimates. Since the survey area for both studies overlapped, it provides evidence that the PLEO method is capable of providing reliable density estimates for large bodied bird species. Alternatively, comparing the PLEO based density estimates for hornbill species with those derived by McConkey and Chivers (2004), four out seven hornbill species density estimates were significantly lower than our PLEO estimates. McConkey and Chivers (2004) suggests that, though their overall study location provided suitable habitat (secondary growth and riverine forests) for both the oriental pied hornbill and white-crowned hornbill, their specific sampling units lacked such habitats, which may have led to low counts and hence the low density estimates for these species. In the case of the wreathed hornbill, they suggest that observed individuals were nomadic and mainly seen during the months of March, April and May, which coincided with the period of high fruit availability at their study location. For the helmeted hornbill, McConkey and Chivers (2004) suggests that the population was resident to the area and likely faced low levels of competition regarding food sources from nomadic populations. Though their density estimates were slightly lower than the PLEO based density estimates, they did fall within the range of estimates derived from our point transect survey, indicating that helmeted hornbills are likely to persist at the same levels as that in the comparison study area. In contrast, the lack of any external studies for the white-rumped shama inhibited comparison of density estimates and their survey effort in other sites. Nonetheless, our point transect survey results for the white-rumped shama showed high precision in terms of model fit ($p = 0.06$) and also closely resembled density estimates derived using the PLEO method, which highlights the PLEO method's accuracy of estimating wildlife densities.

In the case of primate species, PLEO based density estimates for red langur was significantly lower than the density estimates found in the comparison study. Difference in range of density estimates may be due to the dissimilarity in forest types and elevations between GNNR and the comparison study location, as such landscape features are known to affect primate densities (Marshall 2010). Marshall (2010) found that habitat quality determined by classes of food sources (e.g. preferred versus fallback) are correlated with population densities for red langur species. Red langur densities are highly correlated with availability of preferred foods such as seeds, leaves and unripe fruit pulp, (Marshall 2010). Another important aspect was the negative relationship between population densities and altitude for red langur species (Marshall 2010). Alternatively, for the silvery lutung, the comparison study was carried out in a riverine habitat to assess validity of two different methods (foot surveys versus boat surveys) for studying primate species (Matsuda et al. 2016). Though some primate species are known to prefer forests near rivers, silvery lutungs are likely to spend only a fraction of their day foraging in forests near rivers before returning to the interior of the forest at night to rest (Matsuda et al. 2011). Since the PLEO based density estimates were derived by pooling all participant responses, specifics such as relationship between habitat quality and elevation with population densities across the sampling units as well as preferences of habitat type by animal species were intrinsically averaged.

In the case of the felid species, PLEO based density estimates for the Sunda clouded leopard were within the range

of estimates for the comparison study, whereas the PLEO based density estimates for the other two felid species were lower than the range of estimates of the comparison studies. Though the PLEO based estimates fell within the range of estimates of Sunda clouded leopard comparison study, the authors of the comparison study Brodie and Giordano (2012) draws caution in comparing their results with other study sites. They suggest that use of different field methods and modelling approaches may likely lead to different estimates and hence limit comparison. Moreover, their results only provide evidence for a small effective sampling area as they dropped a significant portion of their camera trap data from a logged forest as it failed to return any records of Sunda clouded leopards. Their rationale for not using the data from logged forests was to avoid underestimating density estimates at their study site. Brodie and Giordano (2012) also suggests that studying Sunda clouded leopards using ground based camera traps is difficult due to these animals being cryptic and having semi-arboreal habits that often result in low detection probabilities. Just like Sunda clouded leopards, marbled cats are also difficult to study as they prefer inhabiting undisturbed forests over partially logged or disturbed forest landscapes (Wearn et al. 2013, Hearn et al. 2016). In particular, Hearn et al. (2016) suggests undisturbed lowland hill forests may support higher densities of marbled cats than undisturbed higher elevation forests in Borneo. Though marbled cats exhibit diurnal activity patterns, they have been found to be intolerant of anthropogenic features of landscapes that increase the rarity of sightings. Alternatively, leopard cats are adaptive and found in a broad range of habitats that include plantations and logged forests (Lim 1999, Mohamed et al. 2013). Mohamed et al. (2013) suggest that leopard cats are likely to adapt well to forest disturbance as gaps in the forest canopy and open habitats lead to higher prey availability. Comparing this to our study location, the GNNR consists of dense forest with low canopy gaps that likely provide low levels of complex understory vegetation structure required by prey populations, such as small rodents, to persist (Rabinowitz 1990, Schmid-Holmes and Drickamer 2001). Mohamed et al. (2013) also found that only 5% of all photographs were taken during the day time (06:00 a.m.–06:00 p.m.), which is consistent with the fact that leopard cats are nocturnal. Since hunters spend most of their time in the forest during the daylight hours, this mismatch in activity patterns may potentially lead to lower encounters and sightings. Overall, using transects to measure density of carnivores in the rainforests of Borneo is not ideal and should be used in conjunction with other methods (Mathai et al. 2013).

A particular critique of camera trap data is related to the widespread practice of setting camera traps at non-random locations, which likely only provides snapshots into the non-random space use by animals (Wearn et al. 2013). Choosing locations based on imperfect assumptions can lead to underestimating key population parameters, especially for elusive species for which we know very little (Wearn et al. 2013). In general, camera trap surveys require thorough understanding of biology and ecology of the target species in light of features of the study site to enable gathering high quality data (Cheyne et al. 2013, Mohamed et al. 2013, Hearn et al. 2016). Moreover, significant levels of resources in terms of equipment, knowledge about different modelling techniques, and expertise in data analysis using specialist software plays a limiting role in using such techniques (Foster and Harmsen 2012). In our case, though comparison studies followed similar data analysis techniques, variations in field methods (e.g. creating openings and trails, selecting trap locations, etc.) and subsequent data management protocols between the studies contribute to the difficulty in replicability of such methods leading to low external validity. Point counts and transect surveys present significant challenges for wildlife managers and conservation groups attempting to carry out monitoring of wildlife populations. Often such methods make it difficult to monitor a large number of species, as only a few species have well known calls, leave clear tracks, or have easily detectable nests or nesting behavior (Hoeven et al. 2004, Marshall and Meijaard 2009). Additionally, such methods often cover relatively small proportions of the total study area that leads to insufficient number of detections barring extrapolation of population densities or other population parameters (Wilkie and Carpenter 1999, White and Edwards 2000). Hoeven et al. (2004) attempted to compare PLEO based density estimates directly to those derived using line transect surveys within their study site, but due to limited detections from line transects were unable to make direct comparisons. Similarly, in our case, we were restricted from comparing the bulk of our PLEO based density estimates with point transect survey results due to lack of detections.

An argument against the PLEO methodology is that since study participants likely visit small areas within the sampling units due to their preferences, it may be a source of bias for their reported estimates. However, when a large enough sample of participants are interviewed, whose collective visits form a representative sample of the study area, it is likely to overcome such bias (Mayaka et al. 2015). In our study, we attempted to control for this attribute by expanding our sample area to cover approximately 18% of the total surface area of the GNNR, something which would not be possible using conventional methods employing same level of resources. White et al. (2005) carried out a comprehensive analyses of research papers in ecology that used questionnaires to collect primary data, and provided recommendations on best practices based on their analysis of the literature. In particular, White et al. (2005) recommends that research papers using questionnaires in ecology need to explicitly document target populations and hypotheses well, pilot questionnaire formats before use, select sample sizes that allow for statistical analysis, document rationale for using the selected method, minimize selection of non-respondents, use a simple questions and answer format, use an unambiguous structure for the questionnaire, quantify bias arising from non-response, ground truth accuracy of responses, and use models to analyze interrelated data. The PLEO method enunciated by Hoeven et al. (2004) conforms to all the recommendations except the last recommendation as it is beyond the scope of the PLEO method.

As a simple and cost-effective method in estimating wildlife densities, the PLEO method is radically different when compared to conventional density estimation methods. In particular the use of semi-structured questionnaire on key actors to gain expert opinions in conservation requires

attention to both theoretical and practical consideration (Newing et al. 2011). While theoretical underpinnings of using social research is steeped in the ontological debate of achieving objectivity, practical considerations include gaining access to research participants and ensuring 'free, prior and informed consent' (Newing et al. 2011). Objectivity of the PLEO method is achieved by the tools used (area map, picture booklet and semi-structured questionnaire). In case of practical considerations of gaining access to 'experts' in wildlife within communities, we followed the methods used by Hoeven et al. (2004) to engage community leaders and reach respondents who were knowledgeable about their surrounding wildlife. While practical reasons may limit researchers' ability to access indigenous communities in a 'respectful' manner, researchers also need to continously account for cultural sensitivities and historical contexts of their partnering communities during the different phases of their research (Castleden et al. 2012, Ramos 2018). In our case, gaining access to study participants through community leaders respected cultural norms while also addressing issues related to data quality. By engaging community leaders as 'gatekeepers' of their communities to select the suitable study participants for the PLEO survey, we were able to establish rapport with community leaders and community members who participated in the survey. Such rapport building with community leader also helped us to obtain consent from study participants during the individual meetings. Hence, using the PLEO method more frequently over larger scales will inevitably require researchers to first establish communication with community leaders following principles of 'veracity', 'privacy' and 'confidentiality' throughout the pre-fieldwork and fieldwork stages (Kent 2000, Newing et al. 2011).

Gaining access to study participants knowledgeable about surrounding wildlife is an important aspect of the PLEO method as it will reduce error rates and prevent findings from being skewed through misidentifications and lower detection accuracy by non-experts. Camino et al. (2020) evaluated two different LEK-based methods of surveying wildlife with standard wildlife survey methods for large bodied terrestrial mammals and found that probabilities of false-presences for non-hunter responses were significantly higher compared to responses by hunters. As a result, Camino et al. (2020) suggests to find respondents with the 'relevant knowledge and expertise', as non-hunters were likely to pay less attention to presence of wildlife compared to hunters due to its effect on each groups livelihood opportunities. We also talked to other community members who were knowledgeable about surrounding wildlife due to the nature of their work as SMART patrol members. Talking to them enabled us to account for a balance in opinions from each community beyond that of local hunters only. As interests in conserving wildlife was different between both hunters and non-hunters (SMART patrol members), using a second group familiar about surrounding wildlife but having varied interests compared to hunters enabled us the opportunity to account for a balance in opinion regarding wildlife densities from each village. Additionally, voluntary participation of participants likely filtered out those participants who would otherwise only participate in the survey for personal financial gain, and highlights non-financial motivations of respondents to participate in the survey (Lewis and Nkuintchua 2012). Luzar et al. (2011) found that communities dependent on hunting generally showed a higher motivation to participate in research related to wildlife populations likely out of opportunities to use project data to inform a village's wildlife management decisions.

As the PLEO method involves local communities to provide wildlife monitoring data, it can become a powerful citizen science tool if used effectively. In comparison to standards of scientific data collection, citizen science methods are often assumed to be afflicted by issues related to data accuracy and precision (Lukyanenko et al. 2016). Galloway et al. (2006) noticed that data collected by students (non-professionals) on the structure of an oak tree stand compared to those collected by professionals were more likely to skew results in a citizen science project due to issues of selective bias in sampling. However, Galloway et al. (2006) suggest that the selective sampling bias in their study may have been a result of explaining to students that larger trees provide habitat for wildlife that likely caused students to document more unique and larger trees. A meta-analysis of citizen science projects that compared measures of precision between data collected by non-professionals and professionals concludes that there is 'no strong evidence to support the belief that volunteer data are consistently less precise than professional data' (Lewandowski and Specht 2015). Contrarily, it is argued that by allowing non-scientists to collect data for a scientific purpose, may even increase data quality if researchers are able to take full advantage of the process (Lukyanenko et al. 2016). In order to take full advantage of the citizen science process, it is suggested to use simple survey methods, provide in-person training, apply data validation using statistics, carry out targeted recruitment and retention of volunteers to increase data quality over time are only some of the recommendations to address data quality issues in citizen science projects (Riesch and Potter 2014, Lewandowski and Specht 2015).

As a novel method of estimating wildlife densities by leveraging local tacit knowledge, the PLEO methodology can play a priming role in enabling participatory conservation planning and practice while simultaneously building local support for conservation action. Citizen science and participatory research methods can play a crucial role in generating data and create 'trust and empowerment' between stakeholders that is crucial for co-management of natural resources (Schewe et al. 2020). Furthermore, employing participatory approaches in environmental management is likely to create local champions who would infuse local perspectives into conservation outcomes and thereby help to 'build a more inclusive knowledge system grounded in the people and place' (Needham et al. 2020). However, since the efficiency and accuracy of data gathered through indigenous knowledge sources can vary depending on how it is extracted, developing standardized data-gathering methods is suggested to reduce error-rates and data falsification (Luzar et al. 2011). Camino et al. (2020) suggest that though LEK methods are capable of generating 'efficient and accurate data for detecting large terrestrial wildlife', they can be further improved if LEK methods are refined through capacity development of local community members and ensuring that participants take up ownership of survey results. Camino et al. (2020)

also suggests that while interviewing local people with LEK is a more cost and time effective method that enables rapid assessments, developing the capacity of local communities to support data collection can be more cost effective in the long-run, bolster support for conservation action, and create pathways for stakeholder engagement enabling co-management of natural resources (Camino et al. 2020). Therefore, by training participants who participate in a PLEO survey has the potential to improve the accuracy of the method in determining wildlife density estimates and create opportunities for working with communities to co-produce knowledge and conservation outcomes.

Conclusion

In our study, we followed the PLEO methodology developed by Hoeven et al. (2004) that uses market-forecasting techniques to determine wildlife densities. Our study provides preliminary evidence that following the PLEO method can be a cost effective method in estimating wildlife population densities in tropical rainforests. Similar methods leveraging local knowledge have also been used to study presence, and distribution of wildlife species across their native habitat (Evangelista et al. 2018, Silva de Oliveira and Dario 2018). We suggest that rather than replacing conventional methods of estimating wildlife densities as suggested by Hoeven et al. (2004), the PLEO method can be used as a supportive tool to help wildlife managers understand the status of wildlife populations in a quick and reliable way. Therefore, mangers are likely to find the PLEO method as a cost-effective and more practical method to help them make short-term decisions rather than wait extended periods for results derived using conventional wildlife survey methods.

Since local people living in tandem with wildlife in remote locations are often the first to notice changes in their surrounding wildlife populations, tapping into such local knowledge can provide opportunities for collaboration between researchers and local communities (Gilchrist et al. 2005, Needham et al. 2020, Schewe et al. 2020). In essence, by integrating scientific methodologies with indigenous knowledge and expertise, the PLEO methodology has the potential to provide the opportunity to develop a 'participatory' ecological monitoring platform, one that complements rather than replaces conventional methods and enable tracking various wildlife species over space and time from intermediate up to regional scales. A future extension will be to conduct repeated individual meetings over multiple years that would provide insights into wildlife population trends. We suggest the PLEO method is used as a complimentary method to conventional more intensive field methods (e.g. transects, camera traps, etc.) and whenever possible researchers should strive to use multiyear studies with sound sampling design in the field.

In particular, the PLEO method may be most useful for governments and practitioners who are implementing management and conservation interventions across large landscapes with limited budgets. It offers a pragmatic approach to gain insights into wildlife populations, which could be used as an indicator for outcomes as a result of a given intervention. This methodology also has important implications for conservation as it includes communities in a participatory process to understand the status of their surrounding natural resources. Our field teams noted that many interesting conversations stemmed from these semi-structured questionnaires around population variation, seasonality, trends and more. Based on this experience, we suggest future iterations of this approach should incorporate such qualitative information and that conservation NGOs working with wildlife can make this part of their monitoring activity and use PLEO repeatedly to assess wildlife population trends.

Acknowledgements – We thank all the participants who participated in the survey, the Balai Konservasi Sumber Daya Alam (BKSDA) of West Kalimantan who oversee management of the landscape where the research was carried out, local communities and field staff of YPI for their assistance during data collection.
Conflicts of interest – The authors would also like to declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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