

# Research articles

## Hornbill density estimates and fruit availability in a lowland tropical rainforest site of Leuser Landscape, Indonesia: preliminary data towards long-term monitoring

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### Abstract

Reliable density estimates of threatened species provide important information on their conservation status and effectiveness of conservation efforts. Here, we provided the first density estimates that utilize detection probability of four hornbill species in Ketambe Research Station area, Leuser Landscape. Moreover, we made a preliminary investigation of the relationship between hornbill detections and fruit availability. We conducted line transect and phenology surveys during July-September 2019. Our findings highlighted the importance of Ketambe as a stronghold for hornbill populations, indicated by relatively higher density estimates of four species (*Wreathed Hornbill* *Rhyticeros undulatus*, *Rhinoceros Hornbill* *Buceros rhinoceros*, *Bushy-crested Hornbill* *Anorrhinus galeritus*, and *White-crowned Hornbill* *Berenicornis comatus*) than elsewhere. We noted the lack of *Helmeted hornbill* detections which might be due to illegal poaching. Further, we identified the positive influence of ripe fruit supplies on hornbill presence. Our study presented the baseline density data for future hornbill population monitoring studies.

We also recommended conducting studies of hornbill diet and foraging behaviour and their responses to human disturbances.

**Keywords:** hornbill, population monitoring, food availability, detection probability, Leuser landscape

### Introduction

Effective conservation strategies are necessary to halt and reverse the declining trends of biodiversity as a result of detrimental human activities (Butchart et al. 2010; Hoffmann et al. 2010). Population data such as density provides important information on the current conservation status of focus species or taxa (Marthy et al. 2016). Furthermore, this density baseline serves as starting point for wildlife monitoring to assess the rates of population change and to evaluate the effectiveness of conservation measures (Ariefiandy et al. 2013; Marthy et al. 2016).

Hornbills represent an ideal surrogate for population monitoring because of their ecological roles and conservation values. Hornbills are among the

most charismatic bird species in tropical rainforest (Kinnaird and O'Brien 2007). They act as indicators of forest condition and anthropogenic disturbance as hornbills require large and relatively-undisturbed forests with large and tall trees for feeding and nesting (Hadiprakarsa and Kinnaird 2004). Moreover, these species contribute in maintaining forests due to their ability to disperse fruit seeds over vast distances (Kitamura 2011).

Conservation of hornbills is especially relevant to the Island of Sumatra, which harbors nine out of the 13 hornbill species in Indonesia (Kinnaird and O'Brien 2007) where the threats to hornbills are pervasive. The Sumatran lowland tropical rainforest, the primary hornbill habitat is rapidly disappearing with 21.3% of the area being lost between 2000 – 2012 (Margo-no et al. 2014). This problem is exacerbated by the trading of hornbills for use as pets and ornaments (Beastall et al. 2016). These pressures were reflected in the hornbill IUCN Red List, where by the end of 2018, seven of the 13 hornbill species in Indonesia received an upgraded threatened status from either Least-Concern (LC) or Near-Threatened (NT) to Vulnerable (VU), Endangered (EN), or Critically Endangered (CR) (IUCN 2019).

For three species, the changes were highly significant: Wrinkled Hornbill *Rhabdotornhinus corrugatus* and White-crowned Hornbill *Berenicornis comatus* went from NT to EN, and Wreathed Hornbill *Rhyticeros undulatus* went from LC to VU. Furthermore, four other species were up-listed by one threatened status in 2018: Rhinoceros Hornbill *Buceros rhinoceros*, Great Hornbill *Buceros bicornis*, and Black Hornbill *Anthracoseros malayanus* from NT to VU and Bushy-crested Hornbill *Anorhinus galeritus* from LC to NT. It should not be forgotten that the Helmeted Hornbill *Rhinoplax vigil* in 2015 made the most worrying jump from NT to CR, recognizing that it is on the edge of extinction.

Considering their threatened statuses, monitoring the population of these hornbills to understand their breeding success is crucial because it is the essence of their existence in the wild. However, hornbill population data from Sumatra is limited as only few studies have been published (e.g. Anggraini et al. 2000; Kinnaird and O'Brien 2007; Nur et al. 2013; Marthy et al. 2016; Kamal et al. 2018). Among these studies, only studies from Anggraini et al. (2000), Kinnaird and O'Brien (2007), and Marthy et al. (2016) had incorporated estimates of detectability to provide unbiased density estimates with comparable results over time and space (Karanth and Nichols 1998; Buckland et al. 2001).

In addition to hornbill population monitoring, assessment on the hornbill food sources is also important as a potential factor influencing the hornbill density. Anggarini et al. (2000) and Kinnaird and O'Brien (2007) suggested that hornbill densities in Sumatra and Sulawesi were significantly influenced by the type, amount, and ripeness of fruits available in their habitat. Hornbill densities can also highly fluctuate overtime in relation to fruit production, as reported by Kinnaird et al. (1996) who found Red-knobbed hornbill *Rhyticeros cassidix* monthly densities ranged from 9.30 – 82.7 individuals/km<sup>2</sup> during their two-year study at Tangkoko, Sulawesi. This density-fruit relationship thus emphasizes the importance of fruit availability—the principal hornbill diet—in assessing the hornbill population status.

Recalling the importance of providing reliable baseline data for long-term monitoring, this study aimed to: 1) estimate the baseline density of hornbill species accounting for detectability and 2) investigate the relationship between

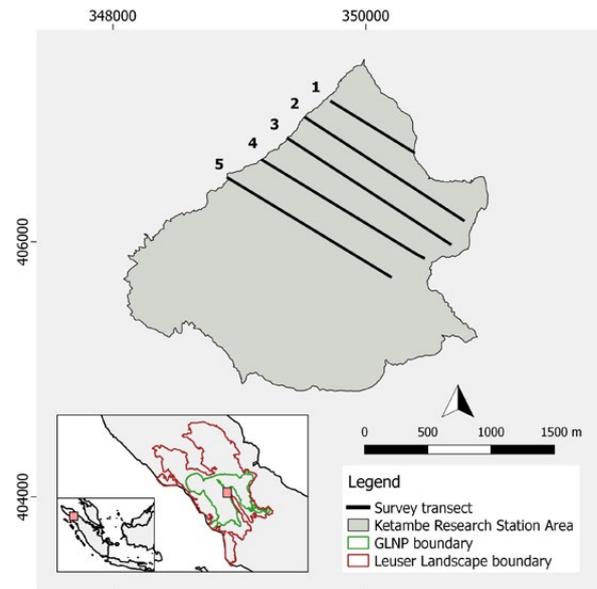
hornbill detections and fruit availability in a monitoring site of the Leuser Landscape. We chose Leuser Landscape as it represents the largest extant Sumatran tropical forest (2.6 million ha) and is a global biodiversity hotspot with significant conservation focus, but is also increasingly threatened by infrastructure development, encroachment and other human activities (Sloan et al. 2018).

## Method

### Study Area

We conducted our study in Ketambe Research Station site (3°41'N, 97°39'E) within Gunung Leuser National Park (GLNP), part of Leuser Landscape, Sumatra (Hardus et al. 2012; Figure 1). Ketambe Research Station was established in 1971 and is widely known for its orangutan long-term studies. The site area (450 ha) is dominated by primary forest across elevation of 350 – 1 000 m a.s.l. (Wich et al. 1999). Approximately one fifth (83.1 ha) of the eastern part of Ketambe Research Station area is secondary forest as the result of selective logging in the early 2000's (Hardus et al. 2012). The site is located next to the boundary of GLNP, separated from nearby villages by the Alas River in the east.

We used five straight-line transects of 0.78 km (transect 1) and 1.50 km (transect 2-5) separated by 200 m. Transect 1 was shorter as it was located in the furthest north of the site area and limited by the site river boundary (Fig. 1). These transects were located to cover around 50% (220 ha) of the site area representing a mix of primary and secondary forests with an elevation between 400 – 600 m a.s.l. This area was chosen as it is the core area for ecological monitoring at the research station and has higher fruit production than the southern area (van Schaik and Mirmanto 1985; Wich et al. 2006). We did not allocate transects in the southern area (600 – 1 000 m a.s.l.) as it has rugged terrain and steep slopes, making straight line transect sampling difficult.



**Fig. 1.** Study area showing transects located in the core ecological monitoring area of Ketambe.

### Hornbill population survey

We conducted variable-width line transect surveys for 2.5 months from July-September 2019 during dry (July-August) and wet (September) seasons (Thomas et al. 2010). The survey was conducted every two weeks, resulting in total of five replications. We started the survey from 07:30 – 10:30 (three hours). We surveyed one transect per day and used alternating transects separated by 400 m on the next day to minimize the chance of double counting (e.g. order of survey = transect 1, 3, 5, 2, 4). We also started from the opposite side of each transect for every other replication.

During the surveys, we collected data of 1) hornbill species, 2) detection type (visual or aural), 3) distance between observer and hornbill, 4) angle between observer position on transect and hornbill, and 5) number of individuals. Visual and aural signs were collected to increase the detection as visual encounter are often difficult in the primary forest with thick canopy cover and the data were combined for density estimates (Kinnaird et al. 1996;

Gale and Thongaree 2006; Marthy et al. 2016). Flying hornbills were recorded but not used for density analysis.

### **Fruit availability survey**

We conducted the phenological surveys on the same day after the hornbill survey using the same transects (11:00 – 17:00), following the methods of Anggraini et al. (2000). We observed trees that produced fruits confirmed or potentially consumed by hornbills (e.g. fleshy fruits, drupaceous, and small-medium sized; Hadiprakarsa and Kinnaird 2004). All potential fruiting trees within 50 m of both sides of transects were mapped and identified to species or genus level. We visually estimated 1) the percentage of canopy covered by fruits (% fruit in canopy) and 2) the percentage of ripe fruits primarily based on changes of color due to ripeness (% ripe fruit). We also calculated the percentage of canopy covered by ripe fruits (% ripe in canopy = proportion of ripe fruit x proportion of fruit in canopy x 100). The identified tree species were then compared with the hornbill fruit diet list in Bukit Barisan Selatan National Park, Southern Sumatra (WCS-IP *unpublished data*) and supplemented with field observation to create a list of hornbill fruit diet in Ketambe.

### **Data analysis**

We calculated the hornbill density estimates using Distance 7.3 (Thomas et al. 2010). A transect was assigned as the sampling unit and all hornbill detections per transect were compiled from five survey replications. The total survey effort was 33.9 km (total transects length of 6.78 km x 5 replications). We evaluated the uniform, half-normal, and hazard rate functions with key adjustments as suggested by Buckland et al. (2001) for each hornbill species. We right-trun-

cated the perpendicular distance data if natural breaks occurred to maintain the detection curve monotonicity. We then grouped the distance data based on the suggestion from automatically grouped detection curves provided by the aforementioned model outputs to improve the estimate precision.

To infer the density estimate of rare species with low detection ( $N < 10$ ), we used a multi-species framework proposed by Alldredge et al. (2007). This framework was applied by “borrowing” the detection data of other species within the same genus or family with similar characteristics such as body size, diet, and calling (Alldredge et al. 2007; Marthy et al. 2016). For example, in this study we combined the detections of rare White-crowned hornbill ( $N = 4$ ) with Bushy-crested Hornbill ( $N=13$ ). Both have similar body size, fruit-animal diet, and live in groups (Kinnaird and O’Brien 2007). In the analysis, we calculated a combined detection probability for both species and post-stratified the density estimate by species (Marthy et al. 2016).

To examine the model goodness of fit, we used the Chi-square test. We selected the best fit model with the lowest Akaike’s Information Criterion (AIC) from the set of models (Buckland et al. 2001). If two or more models have similar AIC values ( $\Delta AIC < 2$ ), we examined the detection curves, model fitness, and the percentage of coefficient of variation (% CV) of the density estimates. We selected the preferred model if the curve has continuous diagram with probability of detection near transect  $g(0) \sim 1$ , higher chi-square p-value, and lower % CV (Buckland 2006). Densities were presented as the number of birds per  $\text{km}^2 \pm \% \text{CV}$ .

To assess the relationship between hornbill detections and fruit availability, we used the

**Table 1.** Density estimates of four hornbill species in the lowland part of Ketambe Research Station site during July-September 2019 surveys.

Species	No. of detections	Average group size	Probability of detection	Density (ind/km <sup>2</sup> )			
				Estimate	Lower 95% CI	Upper 95% CI	CV
Wreathed hornbill	86	2.00	0.31	39.26	25.68	60.01	19.10
Rhinoceros hornbill	37	1.38	0.69	3.05	1.30	7.11	34.10
Bushy-crested hornbill	13	7.26	0.54	6.06	2.37	15.51	38.80
White-crowned hornbill	4	3.50	0.43 <sup>a</sup>	0.51	0.12	2.20	67.46
Helmeted hornbill <sup>b</sup>	1	-	-	-	-	-	-

<sup>a</sup>Probability of detection was combined with Bushy-crested hornbill.

<sup>b</sup>Insufficient number of detections for density analysis.

negative binomial regression model that accounts for over-dispersed count data (Dobson and Barnett 2008). We assigned the number of hornbill detections per survey (detections) as the response variable. Relationships between explanatory variables were evaluated using correlation test with cut-off R value of 0.70 used in ecological studies (Mackenzie et al. 2006). Two final explanatory variables were used: 1) mean percentage of canopy covered by fruits (% fruit in canopy) and 2) mean percentage of canopy covered by ripe fruits (% ripe in canopy). Models were ranked using AIC and the best fit model was selected. We then identified the significant variables for hornbill detection based on the model output.

## Results

### *Hornbill density estimates*

We recorded a total of 141 detections of five hornbill species: Wreathed, Rhinoceros, Bushy-crested, White-crowned, and Helmeted hornbills. Aural detections accounted for 58% of total detections. We were able to estimate four species densities except for Helmeted hornbill that was only detected once (Table 1). The den-

sities ranged from 39.6 individuals (ind)/km<sup>2</sup> ± 19.1% for Wreathed hornbill to 0.51 ind/km<sup>2</sup> ± 67.46% for White-crowned hornbill. Overall, the analysis generated reasonably precise density estimates indicated by the CV < 40%. An exception occurred for White-crowned hornbill estimate, which has a high CV due to limited detections.

### ***Relationship between fruit availability and hornbill detections***

We recorded 166 observations of fruiting trees, consisting of 16 species within nine families during the survey (Appendix 1). A comparison between seven explanatory models showed that the model "Hornbill.detection ~ % ripe in canopy" was the best fit model with the lowest AIC (Table 2). We found that the mean percentage of canopy covered ripe fruits was a significant explanatory variable for hornbill detection (Table 3). Nevertheless, we acknowledged the possibility that not all fruit species consumed by hornbills in Ketambe were observed and verified. We also put caution on the limited sample size for replications (N = 5) when performing the analysis.

## Discussion

Information on species population such as density estimate is not only important for the conservation managers to assess the species status in their area but also for assessment of their global conservation status (i.e. as in the IUCN species assessment). Here we presented the first density estimates of four hornbill species in Ketambe Research Station derived from sampling methods that utilized detection probability. As one of the last strongholds of tropical forests in Sumatra, the density estimates for these four hornbill species (Wreathed, Rhinoceros, Bushy-crested, and White-crowned hornbills) were relatively higher compared to other sites in Sumatra, Kalimantan, and Thailand (Table 4). We also found that the ripe fruit abundance had an influence on hornbill detection in the study area.

In our study site, the Wreathed Hornbill had the highest density estimates, approximately 5.23 times higher than density in South-west of Sumatra (Anggraini et al. 2000) and the highest from all hornbill population studies that utilized detection probabilities (McConkey and Chivers 2004; Gale and Thongaree 2006; Marthy et al. 2016). Kinnaird and O'Brien (2005, 2007) reported the influence of food availability on hornbill density, in which the birds follow the figs, a favorite dietary item of most hornbills. Here, the large number of Wreathed hornbills were related to the fruiting seasons of their favorite diet *Dysoxylum alliaceum* which occurred during the first three surveys of this study. From our observation, *Dysoxylum alliaceum* distribution is highly limited in Leuser Landscape and no documentation has been recorded in other locations outside Ketambe yet.

**Table 2.** Top three negative binomial regression models ranked to assess the performance of food availability variables in explaining hornbill detections. Models incorporated percentage of canopy covered by ripe fruits (% ripe) performed better than other predictors.

Models	AIC	$\Delta$ AIC	Resid.df	2 x log-Likelihood
Hornbill.detection ~ % ripe in canopy	37.44	0	3	-31.44
Hornbill.detection ~ % ripe in canopy + % fruit in canopy	37.87	0.43	2	-29.87
Hornbill.detection ~ % fruit in canopy	41.06	3.62	3	-35.06

**Table 3.** Negative Binomial Regression Model describing explanatory variables of hornbill detection based on the top model. Back transformed coefficients ( $\beta$ ) and confidence interval (CI) represents the strength and direction of influence. Significant variables (p-value < 0.05) are highlighted in bold.

Explanatory variable of hornbill detection	$\beta$	95% CI	z value	p value
<b>Intercept</b>	7.73	3.14-18.20	4.57	0.000
<b>% ripe in canopy</b>	1.07	1.03-1.13	3.02	0.003

**Table 4.** Density estimates (individuals/km<sup>2</sup> with 95% CI) of hornbill species from this study compared to other studies accounted for detectability.

Species	This study	South-west Sumatra (Anggraini et al. 2000)	South-central Sumatra (Marthy et al. 2016)	Central Kalimantan <sup>b</sup> (McConkey and Chivers 2004)	Southern Thailand (Gale et al. 2006)	Ketambe <sup>c</sup> (Van Schaik et al. 1992)
Wreathed hornbill	39.26 (25.68-60.01)	7.50 <sup>a</sup>	+	0.10	0.69 (0.40-1.18)	4.12 (1.19)
Rhinoceros hornbill	3.05 (1.30-7.11)	2.60	1.20 (0.70-2.10)	3.70 (2.50-4.90)	2.69 (1.99-3.64)	2.39 (0.91)
Bushy-crested hornbill	6.06 (2.37-15.51)	3.05	4.60 (1.70-12.20)	5.50	0.64 (0.32-1.28)	4.89 (3.06)
White-crowned hornbill	0.51 (0.12-2.20)	+	+	0.30	0.08 (0.03-0.24)	-
Helmeted hornbill	+	1.90	0.40 (0.20-0.70)	0.70	1.21 (0.81-1.82)	0.42 (0.20)

+ Species present

- Species absent

<sup>a</sup>Density estimate calculated from Distance but without information on CI.

<sup>b</sup>Density for Rhinoceros hornbill was calculated from distance sampling; other species were estimated from strip transects

<sup>c</sup>Used multiple triangulation method in January 1992; density estimates were presented with standard deviation (SD).

The detections of Wreathed hornbill were decreased in last two surveys, potentially because they were foraging on fruits outside the study area, as Wreathed hornbill have large home range up to 2400 ha (Kitamura 2011).

The density of Rhinoceros hornbill was also higher than in other studies. As this species is highly frugivorous and favored figs (Hadiprakarsa and Kinnaird 2004; Kitamura 2011), the numerous fig trees in Ketambe provide abundant food supplies. Furthermore, Hadiprakarsa and Kinnaird (2004) observed similar diet preferences between Rhinoceros and Helmeted hornbills which could lead to competition between these two species. Helmeted hornbills

are rarely present in Ketambe; thus, we expect Rhinoceros hornbills to dominate food sources and result in higher density.

Bushy-crested hornbill density was also considerably higher in Ketambe. This species strongly avoids disturbed areas and prefer closed-canopy forest (Anggraini et al. 2000). As Ketambe is surrounded by human settlements and plantations in the eastern side, our study site represents an ideal habitat for this species. However, we also observed Bushy-crested hornbills in our secondary forest site next to the Alas River. This observation indicated that Bushy-Crested might not be as highly sensitive to disturbed areas as previously reported.

This study also provides the density of rare White-crowned hornbill, the first in Sumatra. This omnivorous species has a substantial meat diet but limited information of its fruit-meat diet has been published (Kinnaird and O'Brien 2007; Kitamura 2011). Thus, their density might be influenced by not only fruit supplies, but also small animals and insects, requiring further assessments. It is worth noting that White-crowned hornbill density was generated through the multi-species framework in Distance (Allredge et al. 2007), which can be useful in estimating density of rare species in future studies.

While there is no estimate for Helmeted hornbill density in Ketambe, the single aural detection indicated a much lower density compared to other studies. The scarcity of Helmeted hornbill detection might have resulted from illegal poaching in Ketambe that started in the early 2010s due to rapid demand of the hornbill casques (Beastall et al. 2016). Since 2011, the research and patrol activities within Ketambe Research Station were interrupted until 2015, thus routine protections were absent during that period. The population decrease was confirmed by van Schaik's study (cited from Kinnaird et al. 1996) that reported higher density of this species in Ketambe three decades ago (0.42 ind/km<sup>2</sup>), although differences in survey method and analysis were acknowledged. This decrease was also reported by local research staff including authors MI, ARW, and IBR who frequently encountered Helmeted hornbills in 1990's and 2000's. We suggested poaching as the main factor for low detection of Helmeted hornbill in the study area as there was no notable habitat loss or disruption in Ketambe and the fruiting trees, especially figs, were still present in large numbers.

Previous study in Ketambe site by van Schaik in 1992 (cited from Kinnaird et al. 1996) report-

ed the presence of Wrinkled hornbill and Black hornbill. Great hornbills are present in forests around Ketambe and usually are found together with a flock of Rhinoceros hornbills in fruiting trees (ARW and MI, personal observation). However, these three species were not detected in this survey. We have limited knowledge of the absence of Great, Wrinkled, and Black hornbills. Without ruling out the possibility of poaching, we suspect the competition with abundant closely-related species like Wreathed and Wrinkled or Rhinoceros and Great hornbills might contribute to this lack of detection which requires further investigation.

Previous studies reported the increase of hornbill abundance in relation to fruit availability (Kinnaird et al. 1996; Anggraini et al. 2000; Naniwadekar et al. 2015). This study found that overall hornbill detections were positively correlated with the percentage of ripe fruits in the canopy. Our findings confirmed the importance of ripe fruits reported by Anggraini et al. (2000), as we observed hornbills were attracted to the trees with abundant ripe fruits. The ripe fruits provide more sugar and nutrients than unripe fruits, thus they are preferred by hornbills (Hadiprakarsa and Kinnaird 2004; Kitamura 2011). A notable observation of this relationship was the Wreathed hornbill in Ketambe. Their detections were significantly lower in the last two surveys because there were no ripe fruits of *Dysoxylum alliaceum* present.

### **Implications for hornbill conservation management**

Our study provides reliable baseline density data of four hornbill species in Ketambe Research Station, Leuser Landscape. The findings confirm the importance of Ketambe for hornbill conservation indicated by overall species higher densities, especially Wreathed hornbills which

to our knowledge is the highest ever documented from studies utilized detection probabilities (Table 4). Future research is needed to better understand the population and ecology of hornbills in the Leuser Landscape. Firstly, as hornbill densities fluctuate over time in relation to fruit supplies, long term monitoring (e.g. monthly and yearly) is needed to identify the population dynamics. Secondly, as the current study only cover a small portion of the Leuser Landscape, we suggested conducting population surveys in additional sites in this landscape. Thirdly, hornbill foraging behavioral studies are needed to identify the species and characteristics of fruits and animals consumed by hornbills. Finally, as Ketambe is located next to human settlements, comparative studies of hornbill densities between pristine, edge, and disturbed habitats could provide important information on hornbill responses to disturbances.

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### Appendix 1. Summary of observed fruiting trees categorized by family.

Family	Species	No. of observation
Annonaceae	<i>Canarium odorata</i>	12
Clusiaceae	<i>Garcinia dioica</i>	6
Combretaceae	<i>Terminalia bellirica</i>	6
Elaeocarpaceae	<i>Elaeocarpus petiolatus</i>	3
Euphorbiaceae	<i>Sapium baccatum</i>	1
Meliaceae	<i>Dysoxylum alliaceum</i>	98
Moraceae	<i>Antiaris toxicaria</i>	1
	<i>Ficus altissima</i>	5
	<i>Ficus annulata</i>	1
	<i>Ficus benjamina</i>	2
	<i>Ficus drupacea</i>	2
	<i>Ficus obscura</i>	8
	<i>Ficus parietalis</i>	1
	<i>Ficus</i> sp.	8
Myristicaceae	<i>Knema laurina</i>	11
Sapindaceae	<i>Pometia pinnata</i>	1