Survival strategies of the sun bear (*Helarctos malayanus*) in the lower Kinabatangan floodplain, Sabah, Malaysian Borneo

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Roshan Guharajan

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Adviser: David L. Garshelis

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Dedication

This thesis is dedicated to my parents, N. Guharajan Nadarajan and Shanthini Lee, my brothers, Sunil and Deepan Guharajan, and Teoh Shu Woan.

Abstract

The Southeast Asian island of Borneo is an important stronghold for the sun bear (*Helarctos malayanus*). However, largely due to the expansion of oil palm (*Elaeis guineensis*) plantations, prime sun bear habitat in Borneo has either been lost or is severely fragmented. The lower Kinabatangan floodplain in Sabah, Malaysian Borneo is an example of a particularly fragmented landscape, with small forest fragments amid vast areas of oil palm. Our research aimed to understand how sun bears are able to survive in this seemingly poor habitat. First, we used camera traps and sign surveys to examine trends in habitat use and activity patterns of bears within the remaining forested areas. Next, we interviewed oil palm plantation workers and farmers to identify if sun bears and other threatened wildlife fed on oil palm fruits, and if conflicts with people resulted because of this. Our results indicated that, compared to primates, wild pigs, and elephants, sun bears were able to feed on oil palm fruits without causing much damage to the crop, and, more importantly, without being detected. Sun bears did use narrow riparian corridors to move between larger forest fragments, but did not spend much time feeding or resting within these corridors. Larger forest fragments served as a better buffer from human activities, and were more intensively used by bears. On certain habitat features, such as forest trails, sun bears were more nocturnal and reduced using these features in proximity to people. This research highlights the importance of relatively small forest areas (~ 20 km^2) for sun bears in a landscape dominated by agriculture. Sun bears have proved to be remarkably resilient in the face of dramatic habitat and landscape changes.

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INTRODUCTION

The sun bear (*Helarctos malayanus*), arguably the least known species of bear, has a limited geographic range in Southeast Asia that is shrinking with expansion of farming and conversion of forests to plantations of rubber and oil palm. The aim of this research was to understand the effects of large scale forest fragmentation on sun bears in the lower Kinabatangan floodplain, Sabah, Malaysian Borneo. Conversion of natural forests to oil palm is believed to be the primary threat to sun bears in the Sundaic region (Borneo and Sumatra; Fredriksson et al. 2008), but it is also known that sun bears utilize calorically-rich oil palm fruits, and that these could represent a dietary buffer in inter-masting years, when natural fruits are scarce (Wong et al. 2005a, Fredriksson et al. 2007). The aim of this research was to examine this potential tradeoff.

Specifically, our goals were to: 1) understand the movement patterns of sun bears along a riparian forest corridor bounded by oil palm plantations, 2) assess important habitat resources for sun bears, 3) identify the components of the sun bear diet, and how much oil palm fruits were a part of it, and 4) understand how sun bears were perceived by people and if there was human-bear conflict within oil palm plantations.

The first three goals hinged on the use of Global Positioning System (GPS) telemetry, which meant attaching GPS collars to wild sun bears. Almost 2 years of field works went into identifying suitable trapping sites, pre-baiting to attract bears, and checking traps. The traps we used were barrel traps that had been successfully used for trapping sun bears in other studies (Figure A; Wong et al. 2004, Fredriksson 2012). We trapped in September 2012, and then from September to December 2013, January to March 2014, and September to December 2014. Trapping was conducted in these distinct time blocks because the GPS collars frequently malfunctioned, causing us to halt trapping and obtain new ones. Using camera traps to monitor trap sites, we noticed bears were extremely wary of entering the dark barrel traps. Accordingly, we built some new traps that were more open, with doors on both ends (Figure A). We also tried a wide variety of baits. Nevertheless, we were unable to attract many bears to our trap sites. Many traps were also damaged by elephants, which were common during certain seasons. Ultimately, we were able to trap only one bear, M01 (Figure B), and fit it with a GPS collar in March 2014 (Figure C). After three days, camera traps photographed revealed this bear without a collar. Despite intensive effort in trapping, we were not able to trap any other individual bears or recapture M01.

As we were not able to use GPS collared sun bears to answer questions about sun bear habitat use, we turned to alternative methods to achieve this. A large scale camera trapping project focused on Sunda clouded leopards (*Neofelis diardi*), smaller sympatric

Figure 1: A standard barrel trap with only one entry (right) and a modified barrel trap with two entries (left).

carnivores, and ungulates had also recorded photographs of sun bears. This study was part of a MS research project by Gilmoore Bolongon from Universiti Malaysia Sabah. With his permission, we used the sun bear photographs to investigate the activity patterns and habitat use of sun bears. We also obtained data from a Cardiff University student, Grace H. Dibden, who conducted sun bear sign surveys as part of her undergraduate professional training year project. We used these data to further assess bear habitat use and selection. This work represents Chapter 1 of this thesis.

The second chapter of this thesis is focused on the fourth goal of this study: the perceptions of oil palm plantation workers and farmers towards sun bears and other wildlife within the agricultural landscape. Here, we relied on interviews with plantation workers.

The two chapters are formatted differently because they are intended for publication in different journals.

Figure 2: Bear #M01, a large male sun bear, cautiously approaching a culvert trap.

Figure 3: Bear #M01 being fitted with a GPS collar.

CHAPTER 1

Strategies of the sun bear (*Helarctos malayanus*) in response to anthropogenic disturbances in the lower Kinabatangan, Sabah, Malaysian Borneo.

Summary

As large areas of forest are lost throughout the tropics, prime habitat of many unique species decline and becomes fragmented. The island of Borneo is no exception, with accelerated clearing of forests primarily for oil palm expansion. Borneo is recognized as an important region for the conservation of the sun bear (*Helarctos malayanus*), but it is unclear how habitat reduction and fragmentation is affecting this ursid. We used camera traps and sign surveys to understand patterns of sun bear habitat use in a particularly fragmented site, the lower Kinabatangan floodplain in Sabah, Malaysian Borneo. Our results showed bears exhibiting behavioral and habitat selection shifts to cope with the challenges of living in this human dominated landscape. Bears were primarily active at night, and did not spend much time in areas more prone to human disturbances. Compared to narrow riparian forest corridors, larger fragments can serve as buffers from anthropogenic activities, and sun bears spent more time feeding and resting there. We conclude that even relatively small forest fragments (~2,000 ha) within large agricultural lands can be important for sun bears, as they feed on crops at night and retreat to safety during daylight. Our research highlights the remarkable adaptations this species has employed to persist in a very drastically modified landscape.

1. Introduction

The island of Borneo has recorded high rates of forest loss (30.2%) over the last four decades (Gaveau et al. 2014). In particular, the State of Sabah (Malaysian Borneo) alone lost 39.2% of natural forest cover in that time (Gaveau et al. 2014). Agricultural expansion (mostly for oil palm plantations) is one of the primary drivers behind forest loss, fragmentation, and degradation in the region (Koh & Wilcove 2008, Abram et al. 2014). The resulting pockets of forest stand out as islands amid a sea of monoculture (Ancrenaz et al. 2004, Abram et al. 2014). Although these forests may not be pristine in floral and faunal composition, they still have tremendous value for biodiversity (Maddox et al. 2007, Alfred et al. 2012, Estes et al. 2012, Nakashima et al. 2013, Ancrenaz et al. 2015).

The tropical rain forests of Borneo are an important stronghold for the sun bear (*Helarctos malayanus*; (Fredriksson et al. 2008). Being a forest-dependent species, sun bears are negatively affected by forest loss (Wong et al. 2013), but can still survive in secondary forests provided that important habitat resources such as fruiting trees are available (Wong et al. 2004, Fredriksson 2012, Wong & Linkie 2013). In primary old growth forests, the availability of sun bear food resources is tied to dipterocarp mast-fruiting cycles (Wong et al. 2005a, Fredriksson et al. 2006). During inter-mast periods, sun bear food sources become scarce, leading to starvation (Wong et al. 2005a), greater predation risk (Fredriksson 2005a), and increased human–bear conflicts (Fredriksson 2005b). With widespread deforestation and degradation occurring in Borneo, little is known about how sun bears are responding to the extreme changes in their habitat. Research on other bear species in human-modified landscapes have shown them to be highly adaptable, supplementing their diet with crops (Maddrey & Pelton 1995, Charoo et al. 2011, Northrup et al. 2012, Takahata et al. 2014, Ditmer et al. 2015) and selecting habitats that minimized contact with people (Nielsen et al. 2004a, Ordiz et al. 2011). However, there are significant risks to both bears and people as bears use such landscapes more frequently (Nielsen et al. 2004b, Jorgenson & Sandoval-A 2005, Charoo et al. 2011, Northrup et al. 2012, Scotson et al. 2014).

Our goal was to better understand the effects of the oil palm–forest landscape on sun bears. We used two methods, camera trapping and sign surveys, that have been used successfully to study the lesser-known species of bears (Akhtar et al. 2004, Ríos-Uzeda et al. 2007, Steinmetz 2011, Steinmetz et al. 2011, Ramesh et al. 2012).

Camera traps have become a standard tool for monitoring low density and cryptic large mammals in Southeast Asian forests (Kawanishi & Sunquist 2004, Linkie et al. 2007, Ngoprasert et al. 2012, Rayan et al. 2012, Wong et al. 2013, Wong & Linkie 2013). Camera trap studies of sun bears have yielded density estimates (Ngoprasert et al. 2012), measures of habitat use and distribution (Wong et al. 2013, Wong & Linkie 2013), and assessments of activity patterns (Wong et al. 2004, Cheah 2013a). Sign surveys within strip transects have also been used to monitor sun bear populations (Augeri 2005, Steinmetz et al. 2011, 2013, Ngoprasert et al. 2011, Fredriksson 2012). Sign surveys are useful as a measure of bear presence/absence, relative abundance, and habitat use (Steinmetz & Garshelis 2010). We employed both these methods in our study to understand the strategies used by sun bears surviving in radically modified habitats. Specifically, we wished to learn whether sun bears could effectively use remnant forest surrounded by expansive oil palm plantations, and if so, we sought to understand features of this habitat that they used or avoided, and aspects of their behavior that enabled them to survive there.

2. Materials and Methods

2.1 Study site

Our study site was situated in the lower Kinabatangan floodplain, in the eastern part of Sabah, Malaysian Borneo. This fertile area is dominated by oil palms, with only small forest fragments remaining (Abram et al. 2014). A network of protected areas consists of seven variably sized forest reserves as well as the Lower Kinabatangan Wildlife Sanctuary (LKWS), which itself constitutes ten different forested areas, termed lots (Ancrenaz et al. 2004). Forest fragments that are currently protected have remained relatively unchanged since 1998 (Francis et al., unpublished data). We surveyed five lots (1 and 4 – 7), three forest reserves (Keruak, Gomantong, and Pin Supu), and private lands within the floodplain. Forest types present in these areas included freshwater swamp forest, peat swamp forest, dryland forest, and limestone forest (Abram et al. 2014). Besides sun bears, large mammalian species present in the lower Kinabatangan include the Bornean orangutan (*Pongo pygmaeus*), Asian elephant (*Elephas maximus*), proboscis monkey (*Nasalis larvatus*), Sunda clouded leopard (*Neofelis diardi*), bearded pig (*Sus barbatus*), and sambar (*Rusa unicolor*).

2.2 Data collection

2.2.1 Habitat use

We used the detection of sun bears through camera traps as a primary measure of bear habitat use in the lower Kinabatangan. The camera traps primary purpose was for a study on Sunda clouded leopards and sympatric carnivores. We deployed Reconyx

PC800 and HC500 infrared camera traps (Reconyx Inc., Holmen, Wisconsin, USA) at 77 different sites along riparian trails, forest trails, and ridgelines (Figure 1). Most sites were located in freshwater swamp forest and dryland forest. We secured camera traps to trees, 40–50 cm off the ground, with a minimum distance of 1 km between sites (Ross et al. 2013). Cameras recorded the time and date of every photographic capture. We checked camera sites at intervals of 30–80 days to check their condition, replace batteries, and switch memory cards. For each camera site, we divided the sampling period into 44 weekly occasions from June 2013 until April 2014. Each interval represented a sun bear detection (1) or non-detection (0) event. We only considered independent detections at each site, which we defined as photographs at least 24 hours apart.

We used the detection of sun bear sign as a second measure of bear habitat use. Sun bears leave distinctive sign during foraging and resting events: claw marks on trees, tree nests, ripped open logs, and broken termite nests (Fredriksson 2012, Steinmetz et al. 2013). We searched for sign within 50 strip transects in the LKWS Lot 5 riparian corridor and Lot 6 forest fragment (Figure 2). Transects were 0.25 ha in size (5 x 500 m) and spaced at least 250 m apart, with 20 transects in the corridor and 30 in the fragment. On every survey, one leader trained in bear sign identification was present.

When we found bear claw marks on trees, we further distinguished between within-year and older claw marks (Steinmetz & Garshelis 2010, Fredriksson 2012), measured the circumference at breast height (CBH) of the tree, and recorded if there were ripped open cavities or torn bark (indicating insect feeding; (Fredriksson 2012)). We estimated the height of each climbed tree in 5-m categories. We noted if the transect contained signs of human activity (cut trails, campsites, rubbish, etc.) and counted the number of *Ficus sp*. trees and termite nests, both important food items for sun bears (Wong et al. 2002, Fredriksson et al. 2006, Fredriksson 2012), in each transect. Also, for comparative purposes, we positioned an additional four 5 x 100 m (0.05 ha) transects in an area known to have abundant sun bear sign. These transects were 80–320 m from oil palm plantation and were within Pin Supu Forest Reserve (Figure 2). We located this area after farmers reported they had come across a sun bear in their oil palm plantations bordering the reserve.

We considered six landscape covariates as potentially important functions of sun bear habitat use: forest type, elevation, buildings, roads, intact forest edge, and water bodies. All covariate data were obtained from forest extent analysis based on SPOT5 and Landsat TM5 imagery of the region (Abram et al. 2014). Roads included both surfaced highways and certain plantation roads. Buildings represented all structures visible from satellite imagery. Water bodies encompassed all rivers and oxbow lakes. At each camera trap and strip transect, we measured the distance to the nearest building (hereafter building), intact forest edge (hereafter forest edge), road, and water body using ArcMap 10.3.1 (Esri, Redlands, California, USA). At each camera trap, we calculated the elevation and categorized the forest type as either freshwater swamp forest, dryland forest, limestone forest, or degraded scrub forest. For strip transects, we calculated the mean elevation of the start and end points of each transect.

2.2.2 Activity patterns

In addition to the habitat use survey, we collected sun bear camera trap data from a general biodiversity survey. Sampling was limited to LKWS Lot 5 (10 sites; 2011 – 2015) and Lot 6 (7 sites; 2010 – 2011). We used sun bear photographs from both this survey and the habitat use survey for activity pattern analysis.

2.2.3 Climbed tree characteristics

We compared features associated with claw-marked trees to a sample of trees that sun bears did not climb. We randomly selected 48 claw marked trees from our strip transects as targets for further investigation. For transects completely lacking claw-marked trees, we divided each into five 100-m segments, and searched each segment for an unclimbed tree (lacking claw marks) of suitable size for bears to climb (minimum CBH of 29 cm based on data from this study).

We set up 20 x 20 m plots centered on each of the 96 trees. We measured the CBH and estimated the height of the focal tree. Within the plot, we counted the number of vines present (hereafter vines). For other habitat variables, we first divided each plot into four 10 x 10 m subplots. Two observers used a striped density stick to quantify the percent understory density in each subplot. We used the mean of all subplots as a measure of the plot understory density. We photographed the canopy directly overhead at the center of each subplot and used the software “HabitApp” (Macdonald & Macdonald 2016) to calculate the proportion of the color black in the photo. Larger proportions indicated greater cover. We took the mean of these proportional values as the canopy cover for the plot.

2.3 Data analysis

2.3.1 Habitat use

We conducted all analysis using R (R Core Team 2015). We utilized a single season occupancy model to measure sun bear habitat use (ψ) from camera trap data (MacKenzie et al. 2002). We examined the effects of covariates (building, forest edge, water body, elevation, road, and forest type) on both ψ and detection probability (probability of a sun bear being detected during an occasion given that it is present; p) using the package “unmarked” (Fiske et al. 2011). In addition, we examined whether the number of trap nights a camera trap was operational in a sampling occasion (0-7) had an effect on p. We used Pearson’s correlation coefficient (r_p) to check for multicollinearity among covariates ($r_p \geq 0.7$). We did not fit models with more than one covariate for ψ and two covariates for p to avoid overfitting the model. We began by fitting constant and single covariate models for both ψ and p. We fit two parameter models for p by taking covariates from the best ranked single covariate models (ΔAIC ≤ 2) and using these in combination with other covariates. The best supported models were identified based on Akaike’s Information Criterion (AIC) and model weight. To identify competing models, we ignored models that were similar to a better ranked model but with an extra parameter (Arnold 2010).

From our strip transect detection/non-detection data, we checked if the number of transects with within-year sign and evidence of human activity differed between the corridor and forest fragment using chi-squared and Fisher’s exact tests, respectively. We checked for differences in number of *Ficus sp.* trees, climbed tree CBH, and density of within year sign between transects in the corridor and fragment using Wilcoxon Mann Whitney tests.

We used logistic regression to model the presence of within-year sun bear sign in the lower Kinabatangan. As the age of sign could only be reliably ascertained for claw marks on trees, we did not include other bear sign in this analysis. We did not include data from the four transects purposefully positioned in an area with high sign density. Our suite of predictor variables included building, road, forest edge, water body, elevation, number of *Ficus sp.* trees in a transect (hereafter *Ficus*), presence of human activity in a transect (0 or 1), and transect location (corridor or fragment). We checked for multicollinearity among predictors ($r_p \geq 0.7$). We fit models with single covariates first, ranked them using AIC corrected for small sample sizes (AIC_c), and then fit more complex models with covariates from the top ranked models ($\Delta AIC_c \leq 2$). We repeated this until we identified the best ranked models using AIC_c and model weight. We ignored competing models with only one additional parameter to better supported models (Arnold 2010). We inspected the fit of the top ranked models visually using binned residual versus fitted plots. We used the area under the receiver operating characteristic (ROC) curve to assess the predictive power of the best supported models.

2.3.2 Activity patterns

We used the package "overlap" (Meredith & Ridout 2014) to calculate a kernel density function from times at photographic capture of sun bears in the lower Kinabatangan during 2010-2015. We only used independent detections (one detection at a site 24 $hour^{-1}$) of bears. We then plotted the resulting distribution.

2.3.3 Climbed tree characteristics

We used Wilcoxon Mann Whitney tests to check for differences in understory density, canopy cover, tree height, vines, and CBH between climbed and non-climbed tree plots as well as between trees with within-year claw marks and those without. We used logistic regression to model the habitat characteristics most associated with climbed trees. We ran two groups of models: one with the response being the presence or absence of claw marks on a tree and another with the response being the presence or absence of within-year claw marks on a tree. We used six covariates in total: CBH, tree height, canopy cover, understory density, vines, and location. We checked for correlation among predictors ($r_p \geq 0.7$). We fit models sequentially with single predictors first and then adding predictors from the top ranked models ($\Delta AIC_c \leq 2$). The final suite of best supported models were selected using AIC_c and model weight. All competing models that contained an extra parameter to a better ranked model were ignored (Arnold 2010). We judged model fit using binned residual versus fitted plots and assessed predictive power using area under the ROC curve.

3. Results

3.1 Habitat use

For a total of 11,359 camera trap nights, we obtained 583 photographs of sun bears from the lower Kinabatangan. From these, only 59 represented independent detections according to our criteria (192.5 trap nights/independent detection of sun bear). We detected sun bears at 29 out of 77 camera trap sites. This resulted in a naïve habitat use estimate of 0.38. The best ranked single season occupancy model included ψ as constant and p as a function of forest edge and building (Table 1). The parametric bootstrap goodness of fit test using the model sum of squared errors suggested a good fit for this model (P = 0.37). Our best estimate of ψ was 0.74 (SE = 0.12). Both forest edge and building had positive effects on p (Figure 3). Our best estimate of p was 0.03 (SE = 0.01) at the mean distance to building (2.08 km) and forest edge (0.61 km). The probability of detecting a sun bear throughout the entire survey (all 44 occasions; p*) was 77.6% at the mean covariate values.

We detected sun bear sign in 31 of 50 strip transects (96.8% claw-marked trees, 3.2% ripped open logs, n = 94). In both the corridor and fragment, a large proportion of transects contained sign (60% and 70%, respectively). All ripped open logs were within the fragment transects. Of the 91 claw-marked trees that we observed, about half (48.4%) were judged to have been made within a year. The density of within-year bear claw marks appeared to be higher in the fragment (quartiles = 0, 4, and 7 ha^{-1}; $\bar{x}$ = 4.4 ha^{-1}, SD = 6.33 ha^{-1}) than the corridor (quartiles = 0, 0, and 4 ha^{-1}; $\bar{x}$ = 2.0 ha^{-1}, SD = 3.0 ha^{-1}), but this difference was not significant (W = 236.5, 95% CI -4 – 0.00002, P = 0.17). All four transects in the Pin Supu Forest Reserve contained bear sign, and sign density was extremely high (quartiles = 60, 100, and 150 ha^{-1}, $\bar{x}$ = 110.0 ha^{-1}, SD = 60.0 ha^{-1}).

Six tree families made up 67% of the total climbed trees: Sterculiaceae, Verbenaceae, Lauraceae, Euphorbiaceae, Rubiaceae, and Tiliaceae. Climbed trees in the corridor were smaller ($\bar{x}$ = 132.1 cm, SD = 75.1 cm) than trees in the fragment ($\bar{x}$ =

145.5 cm, SD = 107.2 cm) but this difference was not significant (W = 889.5, 95% CI -24 – 36, P = 0.94). More than half (54.9%) the climbed trees were 15–30 m tall. About a quarter (26.4%) had torn bark or holes that were noticeable, suggesting insect feeding by the bear. In terms of other potential bear food sources, we located 71 *Ficus sp.* trees and 4 termite mounds within transects. A larger number of *Ficus sp.* trees per transect were in the corridor ($\bar{x}$ = 2.55, SD = 3.86) than in the fragment ($\bar{x}$ = 0.6, SD = 0.81; W = 401, 95% CI 0.00004 – 2, P = 0.03). All termite mounds were within the fragment. Corridor transects were more disturbed (59.3% contained human signs) than fragment transects (40.7%; P = 0.004).

Building, elevation, and water were significant predictors of within-year sun bear sign (Table 2). Greater distances from buildings and water, and higher elevations had positive effects on the detection of within-year claw marks (Figure 4). All competing models had moderate predictive power (61–72%). Binned residual versus fitted plots of competing models displayed an acceptable fit.

3.2 Activity patterns

We obtained 953 photographs of sun bears from 2010 to 2015, of which 116 were independent detections according to our criteria. The activity patterns of bears on these features was largely crepuscular with sustained nocturnal activity (Figure 5). Sun bear activity peaked at around 0400 and 2000 hours respectively, with a drastic levelling off during daylight, between 0800 and 1600 hours.

3.3 Climbed tree characteristics

Understory density around climbed trees ($\bar{x}$ = 0.34, SD = 0.13) was less than around non-climbed trees ($\bar{x}$ = 0.57, SD = 0.21; W = 410.5, 95% CI -14.5 – -6.8, P < 0.0001). The number of vines was also significantly less around climbed trees ($\bar{x}$ = 82.4, SD = 56.5) than non-climbed ($\bar{x}$ = 141.5, SD = 123.8; W = 1578.5, 95% CI 15 – 69, P = 0.002). We observed that climbed trees were taller ($\bar{x}_{\text{climbed}}$ = 20.9 m, SD = 7 m; $\bar{x}_{\text{non-climbed}}$ = 12.8 m, SD = 4.4 m; W = 382, 95% CI -10.6 – -6, P < 0.0001) and had a larger CBH ($\bar{x}_{\text{climbed}}$ = 144.5 cm, SD = 90.7 cm; $\bar{x}_{\text{non-climbed}}$ = 98.2 cm, SD = 51 cm; W = 755, 95% CI -59 – -10, P = 0.004). Canopy cover was similar around climbed ($\bar{x}$ = 0.9, SD = 0.06) and non-climbed trees ($\bar{x}$ = 0.83, SD = 0.19; W = 1034, 95% CI -0.04 – 0.01, P =

0.39). Wilcoxon-Mann-Whitley tests further revealed that there were significant differences in understory density ($\bar{x}_{\text{climbed}}$ = 0.33, SD = 0.13; $\bar{x}_{\text{non-climbed}}$ = 0.51, SD = 0.21; W = 482.5, 95% CI -11.8 – -3.5, P = 0.0002), tree height ($\bar{x}_{\text{climbed}}$ = 20.6 m, SD = 7.7 m; $\bar{x}_{\text{non-climbed}}$ = 15.3 m, SD = 6.2 m; W = 550.5, 95% CI -9 – -2.6, P = 0.001), and CBH ($\bar{x}_{\text{climbed}}$ = 148.6 cm, SD = 79.7 cm; $\bar{x}_{\text{non-climbed}}$ = 110.1 cm, SD = 73.2 cm; W = 658, 95% CI -74 – -6, P = 0.02) between trees with within-year claw marks and those without.

The best supported model explaining climbed tree preference contained the predictors understory density and height (Table 3). Tree height was positively associated with climbed trees while understory density seemed to have a negative impact on climbing (Figure 6). The binned residual versus fitted plots showed good fit, and the model had high predictive power (89.2%). For models focused on just within year climbing, the probability of climbing increased significantly (76%; best supported model) if the tree was within the fragment (Table 4). Tree height and understory density were also significant predictors of within year climbing. All top ranked models explaining within year climbing fit decently, and had good predictive power (76-78.5%).

4. Discussion

The relative detection rates of sun bears was moderately high in the lower Kinabatangan (1 bear detection every 192.5 camera trap nights) as compared to other locations across mainland Southeast Asia (Steinmetz 2011). Our habitat use estimates suggested sun bears used many portions (74%) of the riparian trails, forest trails, and ridgelines throughout the study site. This high proportion was likely due to the long survey period in our study, enabling us to detect more bears at a greater number of sites. In addition, the limited areas of forest probably increased our chances of detecting bears. There was a negative impact of buildings and intact forest edge on detection probability of sun bears in camera traps, indicating that the intensity of use declined near these features. This general avoidance of human features and sensitivity to human presence has been observed elsewhere in the sun bear’s range (Augeri 2005, Nazeri et al. 2012, 2014, Wong & Linkie 2013). Forest trails in the lower Kinabatangan are used by people for many reasons, including research, recreation, eco-tourism, and illegal activities

(Bolongon, Universiti Malaysia Sabah, unpublished data). The intensity of all human activities close to buildings and the forest edge is high, making these features a deterrent to sun bears. We found that sun bear sign also diminished near buildings and water courses that people frequently travelled on. In contrast, forest streams not used intensively by people have an opposite effect on sun bear habitat preference (Nazeri et al. 2014). Our study clearly shows that sun bear habitat use in the lower Kinabatangan is primarily driven by a response to anthropogenic disturbance.

Elevation appeared to be a good predictor of within-year sign detection, but the difference in minimum and maximum elevations of our strip transects was only 17 m. Within such a small range in elevation, there is unlikely to be any major impact on sun bear habitat use. The impact of elevation is likely a reflection of some other influence on sign detection. One possibility is that lower lying patches of habitat tend to get inundated and hold water more frequently, having an impact on tree species composition. These water tolerant trees may not be as important for sun bears as those that grow is less wet areas.

Sun bears have been recorded venturing past the forest edge to feed in oil palm plantations (Normua et al. 2004a, Cheah 2013a), but may avoid trails to be less conspicuous. Poachers use tree platforms and snares to hunt at the forest–oil palm interface (R. Guharajan and S. Payar, pers. obs.). It is thus more risky for bears to use trails crossing into the agricultural lands. Bears that used trails or were near other human travelways rarely did so during daylight hours. Crepuscular and nocturnal behavior of sun bears in disturbed habitats was also observed in Peninsular Malaysia (Azlan J 2006, Cheah 2013a). This is in contrast to a site in Indonesian Borneo, with less human disturbance, where sun bears were entirely diurnal (Fredriksson 2012). Interestingly, radiocollared sun bears in Ulu Segama Forest Reserve, also in Sabah, were diurnal whereas those photographed on trails were crepuscular-nocturnal ((Wong et al. 2004).

Bears used the narrow (130 m – 2 km wide) riparian corridor (Lot 5), but they appeared to make more use of the larger forest fragment directly across the Kinabatangan river (Lot 6; Figure 2), based on the density of their sign (Table S1). The riparian corridor serves to connect LKWS Lot 7 and Pin Supu Forest Reserve (together 3,723 ha) with the larger Lot 5 forest block and Gomantong Forest Reserve (together approximately 11,900 ha; (Ancrenaz et al. 2004)). Camera trap photographs from 2010 to 2015 also indicated that sun bears used this corridor, squeezed between oil palm plantations on one side and boat traffic on the other. We found a higher density of *Ficus sp.* trees, an important food source and possible attractant, within corridor transects; however, even with a greater potential density of food in the corridor, sun bears showed a wariness to the proximity of human disturbance. Although the Lot 6 fragment is relatively small (2,673 ha; (Ancrenaz et al. 2004)), it may provide more insulation from these anthropogenic disturbances. Indeed, within-year sun bear sign was more likely to be detected in Lot 6, indicating that bears felt safe enough to feed and rest there. Sign densities were extremely high in one intensively used patch of forest within Pin Supu Forest Reserve (110 within-year sign ha^{-1}; 135 all-aged sign ha^{-1}). Elsewhere, others have reported sun bear sign densities of 4.9 – 8.8 all aged sign ha^{-1} in primary and commercial forests of central Sabah (Teo 2013), 40 – 45 all-aged sign ha^{-1} in prime habitat in Indonesian Borneo (Fredriksson 2012) and only 9 within-year sign ha^{-1} in prime habitat in western Thailand (Steinmetz et al. 2011). It is possible that bears felt safer in this forest fragment and used it as a refuge after feeding forays into the adjacent oil palm, as reported by farmers. This behavior has been observed among radiocollared bears at another site in Sabah (Normua et al. 2004a) and peninsular Malaysia (Cheah 2013a). Pin Supu Forest Reserve, being only 2,000 ha in size (Ancrenaz et al. 2004), and LKWS Lot 6 are examples of how even small intact forests are important for sun bears in the lower Kinabatangan.

Claw-marked trees were the most common sign left by sun bears in this study. We did not find much evidence of foraging on the ground (e.g., broken logs, excavations of termite nests), in contrast to a site further south in East Kalimantan, Indonesia (Fredriksson 2012). Sun bears climb trees to feed on fruits and insects, for refuge while resting, and possibly to cool off from the hot and humid weather. Whereas sun bears from a much larger and intact forest in Sumatra did not show a preference for climbed tree sizes (Powell 2011), sun bears in the lower Kinabatangan did, selecting wider, taller trees. This preference is likely because larger trees have: (1) increased fruit abundance and availability, (2) more cavities for insects like stingless bees (*Trigona spp.*), (3) larger branches for resting sites, and (4) better access for bears to get above the surrounding canopy where there might be breeze. We also showed that sun bears tended to climb trees surrounded by a sparser understory. The sparser understory below these climbed trees is likely a reflection of a less-disturbed and more shaded forest, rather than selection by sun bears for open understory.

5. Conclusions

Sun bears exhibit a clear avoidance of humans and human associated features in the lower Kinabatangan. This strategy is clearly beneficial when living in a landscape dominated by people and agricultural activities, where encounters with humans could result in the bear being killed. This strategy also aids in the utilization of an important food resource: oil palm fruits. The largely crepuscular and nocturnal peaks in sun bear activity coincide with the lowest level of human activity in plantations. Sun bears do feed on this easily available and abundant food source (Guharajan, in prep; (Cheah 2013a) but oil palm workers and farmers in our study area hardly encountered bears (Guharajan, in prep). This highlights how this extremely adaptable species is able to make use of beneficial resources in a degraded and potentially dangerous landscape. Encouragingly, camera trapped sun bears in the lower Kinabatangan did not exhibit any gunshot wounds or snare-related injuries, unlike those from a similar landscape in Peninsular Malaysia (Cheah 2013a). However, discoveries of disemboweled sun bear carcasses with missing paws in the Kinabatangan and adjacent Segama floodplain (L. Liman, WWF-Malaysia, pers. comm., S. T. Wong, Bornean Sun Bear Conservation Centre, pers. obs.) suggest that poaching of this species does occur, though the extent to which it does is still not clear. Our research indicates that sun bears can persist in a landscape like the lower Kinabatangan provided there are pockets of connected forests, even small fragments, which can serve as refuges and core areas. Whereas we do not diminish the severe negative impacts of widespread land conversion and fragmentation on this forest-dependent species, our results offer hope that conservation of forest fragments within the agricultural landscape enables this species to persist.

6. Acknowledgements

We thank the Sabah Biodiversity Council, Sabah Wildlife Department, and Sabah Forestry Department for permission to conduct this study. RG thanks Dr. Senthilvel Nathan for his assistance as a collaborator.

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Table 1: Top ranked and null single season occupancy models explaining sun bear (*Helarctos malayanus*) habitat use (ψ) and detection probability (*p*) in the lower Kinabatangan, Sabah, Malaysia with number of parameters (k), log likelihood, Akaike's information criterion (AIC), change in AIC (ΔAIC), and Akaike weight.

Model[1]	k	Log likelihood	AIC	ΔAIC	Akaike weight
ψ (.) *p* (Building + Forest edge)	4	-227.16	462.32	0.00	0.29
ψ (Water body) *p* (Building + Forest edge)[2]	5	-226.54	463.08	0.76	0.20
ψ (Elevation) *p* (Building + Forest edge)[2]	5	-226.78	463.55	1.23	0.16
ψ (Road) *p* (Building + Forest edge)[2]	5	-227.05	464.09	1.77	0.12
ψ (Forest edge) *p* (Building + Forest edge)[2]	5	-227.06	464.12	1.80	0.12
ψ (Building) *p* (Building + Forest edge)[2]	5	-227.13	464.25	1.93	0.11
ψ (.) *p* (.)	2	-238.35	480.71	18.39	0.00

[1]Covariates: distance to nearest building, Building; elevation of camera trap, Elevation; distance to intact forest edge, Forest Edge; distance to nearest road, Road; distance to nearest water body, Water Body.

[2]Models with an additional parameter within $\Delta AIC_c \leq 2$ of an otherwise similar better ranked model were not considered competitive despite having strong support. The extra parameter represents noise and thus does not necessarily infer biological significance.

Table 2: Logistic regression models predicting the presence of within year sun bear claw marks in the Lower Kinabatangan Wildlife Sanctuary, Sabah, Malaysia with number of parameters (k), log likelihood, Akaike's information criterion adjusted for small sample sizes (AIC_c), change in AIC_c (ΔAIC_c), and Akaike weight.

Model[1]	k	Log likelihood	AIC_c	ΔAIC_c	Akaike weight
Building + Elevation	3	-31.06	68.64	0.00	0.21
Building	2	-32.50	69.25	0.61	0.15
Water Body	2	-32.81	69.88	1.24	0.11
Elevation	2	-32.87	70.00	1.37	0.10
Building + Water Body[2]	3	-31.91	70.33	1.69	0.09
Elevation + Water Body[2]	3	-32.03	70.58	1.94	0.08
Building + Elevation + Water Body[2]	4	-30.92	70.73	2.09	0.07
Intercept only	1	-34.50	71.08	2.44	0.06
Location	2	-33.68	71.61	2.97	0.05
Forest Edge	2	-33.99	72.24	3.60	0.03
Ficus	2	-34.13	72.51	3.88	0.03
Human Sign	2	-34.47	73.19	4.55	0.02

[1]Covariates: distance to nearest building, Building; mean elevation of strip transect, Elevation; number of *Ficus sp.* trees within strip transect, *Ficus*; distance to intact forest edge, Forest Edge; presence of human activity sign within strip transect, Human Sign; location of strip transect LKWS Lot 5 Lot 6, Location; distance to nearest water body, Water Body.

[2]Models with an additional parameter within $\Delta AIC_c \leq 2$ of an otherwise similar better ranked model were not considered competitive despite having strong support. The extra parameter represents noise and thus does not necessarily infer biological significance.

Table 3: Logistic regression models predicting sun bear climbing tree selection in the Lower Kinabatangan Wildlife Sanctuary, Sabah, Malaysia with number of parameters (k), log likelihood, Akaike's information criterion adjusted for small sample sizes (AIC_c), change in AIC_c (ΔAIC_c), and Akaike weight.

Models[1]	k	Log likelihood	AIC_c	ΔAIC_c	Akaike weight
Tree height + Understory density	3	-39.84	85.94	0.00	0.998
Tree height	2	-47.58	99.28	13.35	0.001
Understory density	2	-48.00	100.13	14.20	0.001
Vines	2	-61.06	126.24	40.31	0.000
CBH	2	-61.61	127.34	41.41	0.000
Canopy cover	2	-63.58	131.30	45.36	0.000
Intercept only	1	-66.54	135.13	49.19	0.000
Location	2	-66.54	137.21	51.28	0.000

[1]Covariates: mean canopy cover of sampling plot, Canopy Cover; circumference at breast height of climbed tree, CBH; location of sampling plot in LKWS Lot 5 or Lot 6, Location; height of climbed tree; Tree height; mean understory density of sampling plot, Understory density; number of vines in sampling plot, Vines.

Table 4: Logistic regression models predicting within-year sun bear climbing tree selection in the Lower Kinabatangan Wildlife Sanctuary, Sabah, Malaysia with number of parameters (k), log likelihood, Akaike's information criterion adjusted for small sample sizes (AIC_c), change in AIC_c (ΔAIC_c), and Akaike weight.

Model[1]	k	Log likelihood	AIC_c	ΔAIC_c	Akaike weight
Tree height + Understory density + Location	4	-45.82	100.07	0.00	0.36
Tree height + Understory density	3	-47.63	101.52	1.44	0.17
Understory density + Location	3	-47.76	101.77	1.70	0.15
Understory density	2	-49.06	102.25	2.17	0.12
Understory density + CBH	3	-48.16	102.58	2.51	0.10
Understory density + Canopy cover	3	-49.04	104.35	4.27	0.04
Understory density + Vines	3	-49.06	104.38	4.30	0.04
Tree height	2	-52.38	108.88	8.81	0.00
CBH	2	-55.55	115.24	15.17	0.00
Vines	2	-56.13	116.38	16.31	0.00
Canopy cover	2	-56.65	117.42	17.35	0.00
Intercept only	1	-57.95	117.94	17.87	0.00
Location	2	-57.02	118.17	18.10	0.00

[1]Covariates: mean canopy cover of sampling plot, Canopy Cover; circumference at breast height of climbed tree, CBH; location of sampling plot in LKWS Lot 5 or Lot 6, Location; height of climbed tree; Tree height; mean understory density of sampling plot, Understory density; number of vines in sampling plot, Vines.

Figure 1: Camera trap locations during 2013 – 2014 in the lower Kinabatangan, Sabah, Malaysia and location of the study site on the island of Borneo (inset).

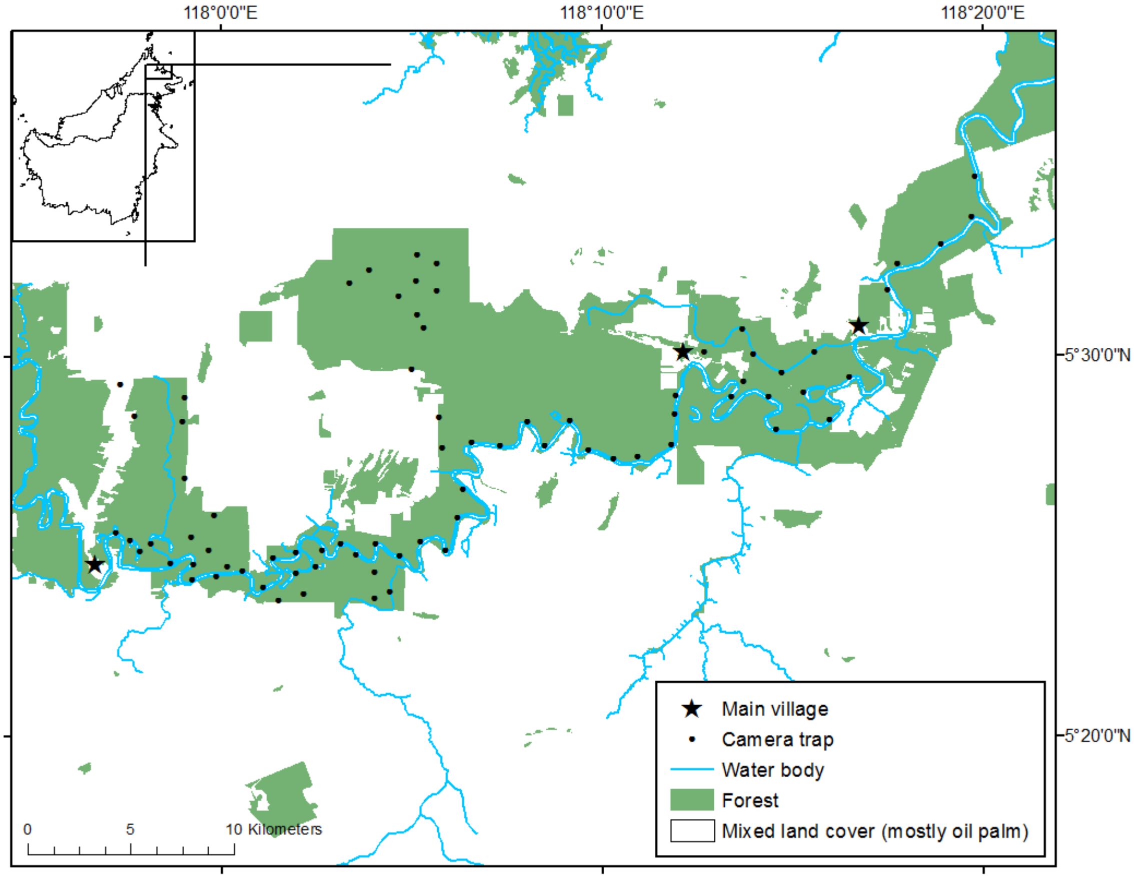


Figure 2: Locations of strip transects in the Lower Kinabatangan Wildlife Sanctuary (LKWS) Lot 5 corridor and LKWS Lot 6 (on opposite sides of the Kinabatangan river) as well as Pin Supu Forest Reserve (inset) during 2013 – 2014 in Sabah, Malaysia.

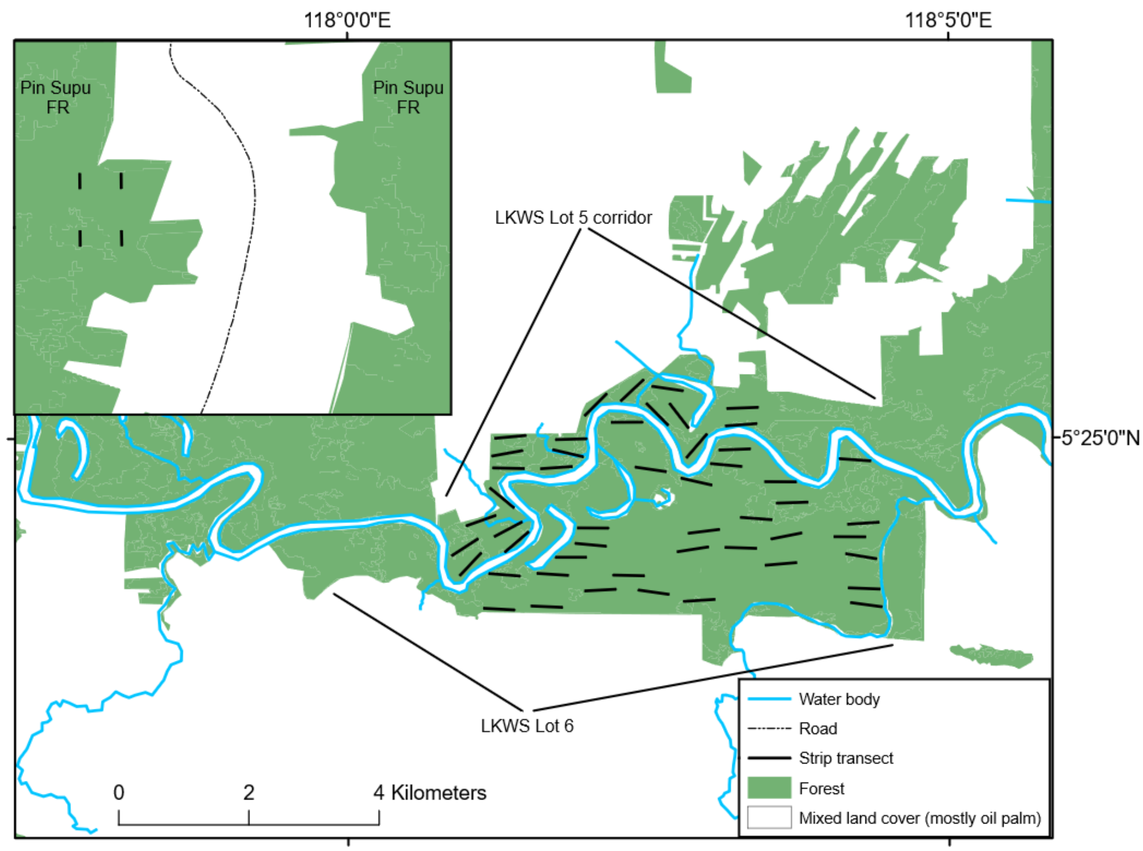


Figure 3: Detection probability as a function of distance to building (top) and intact forest edge (bottom) with 95% confidence intervals (dashed lines) according to the best supported single season occupancy model predicting sun bear (*Helarctos malayanus*) habitat use in the lower Kinabatangan, Sabah, Malaysia. Vertical lines at x-axis represent data points.

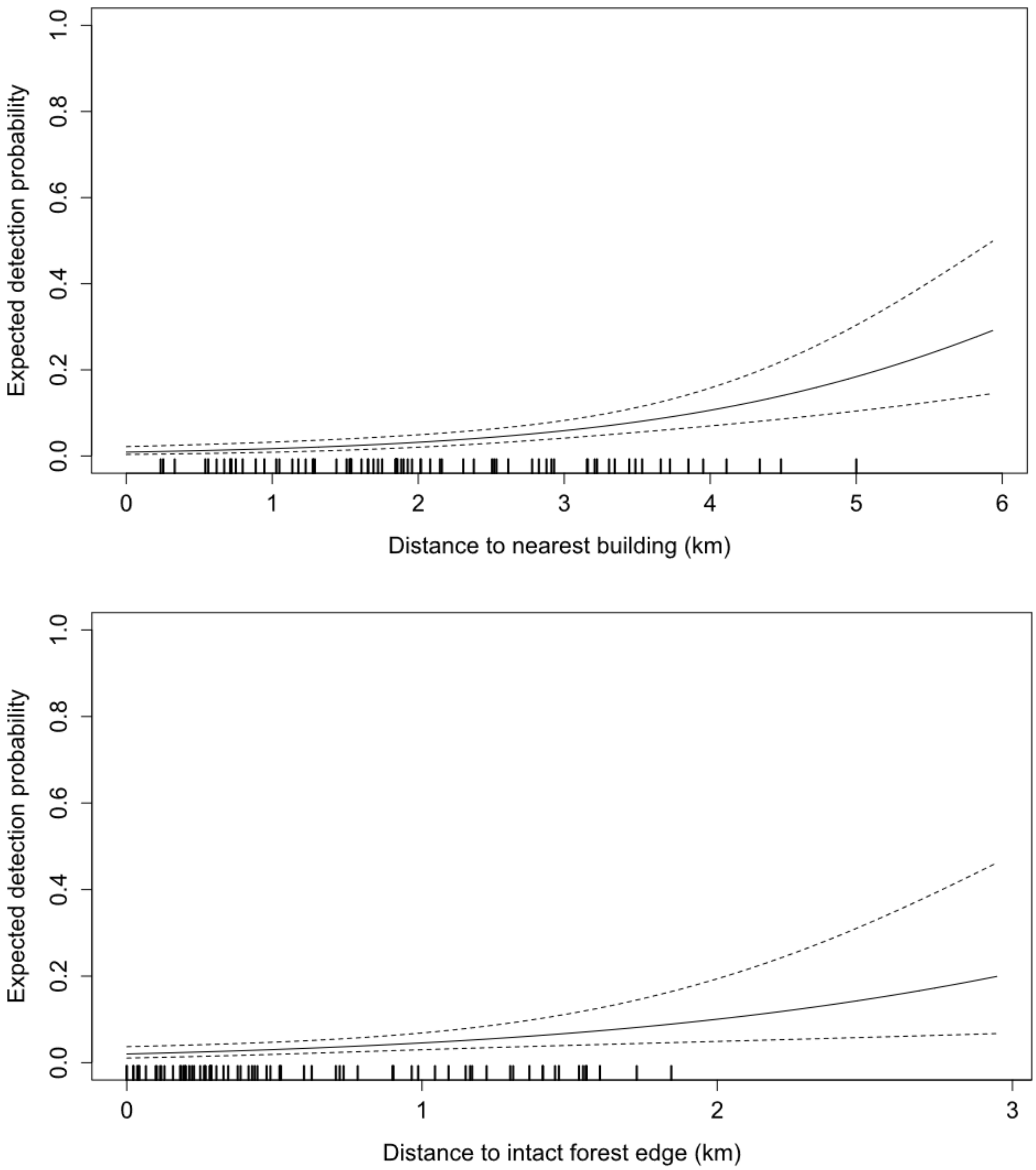
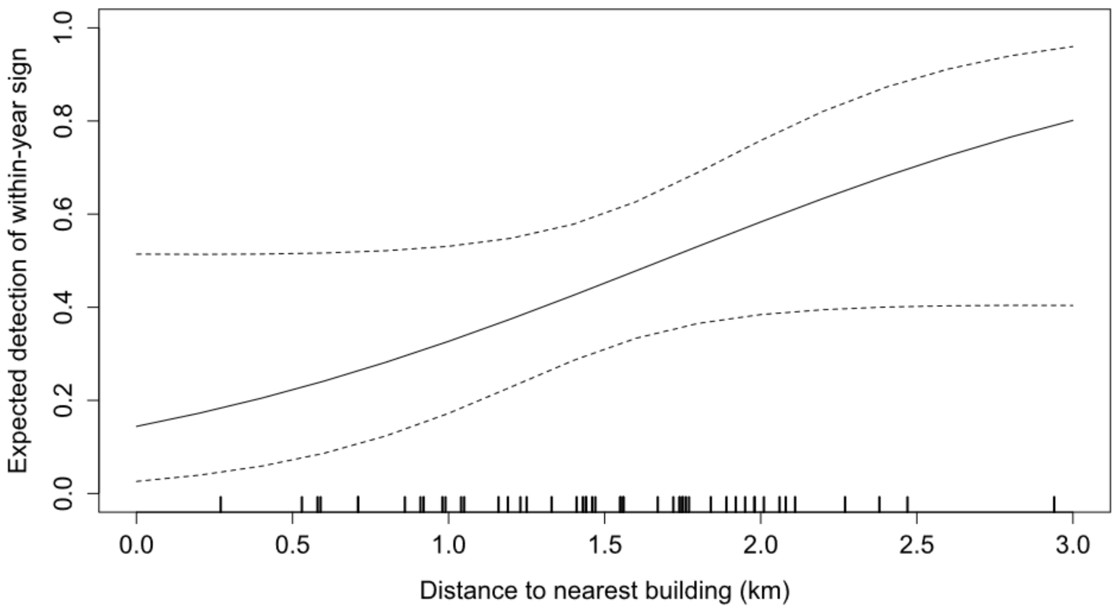
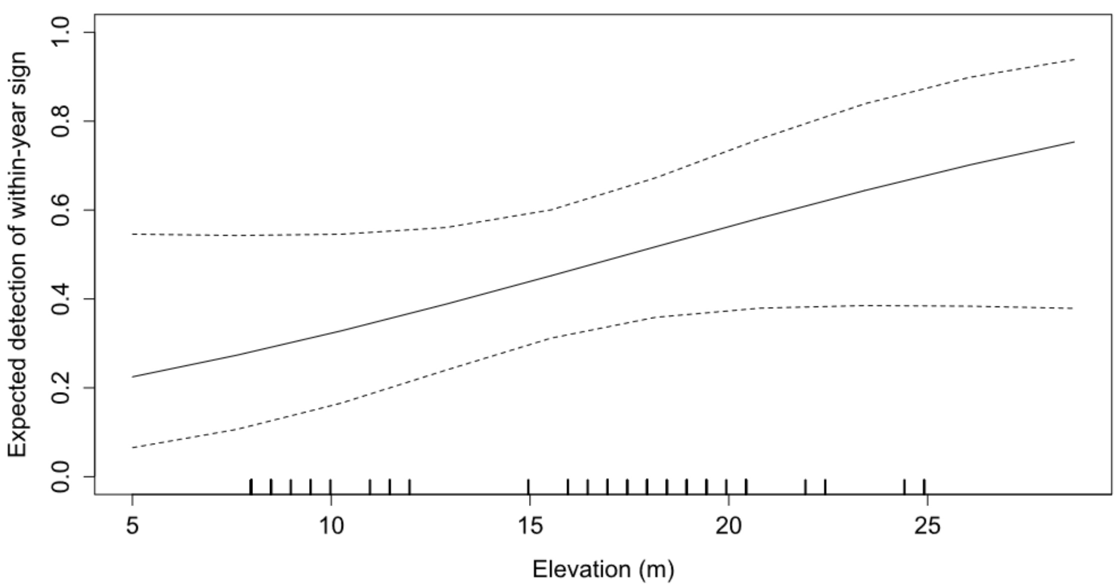


Figure 4: Expected detection of within year sign (claw marks) as a function of distance to building (top), elevation (middle), and water body (bottom) with 95% confidence intervals (dashed lines) in the lower Kinabatangan, Sabah, Malaysia according to the best supported models. Vertical lines at x-axis represent data points.





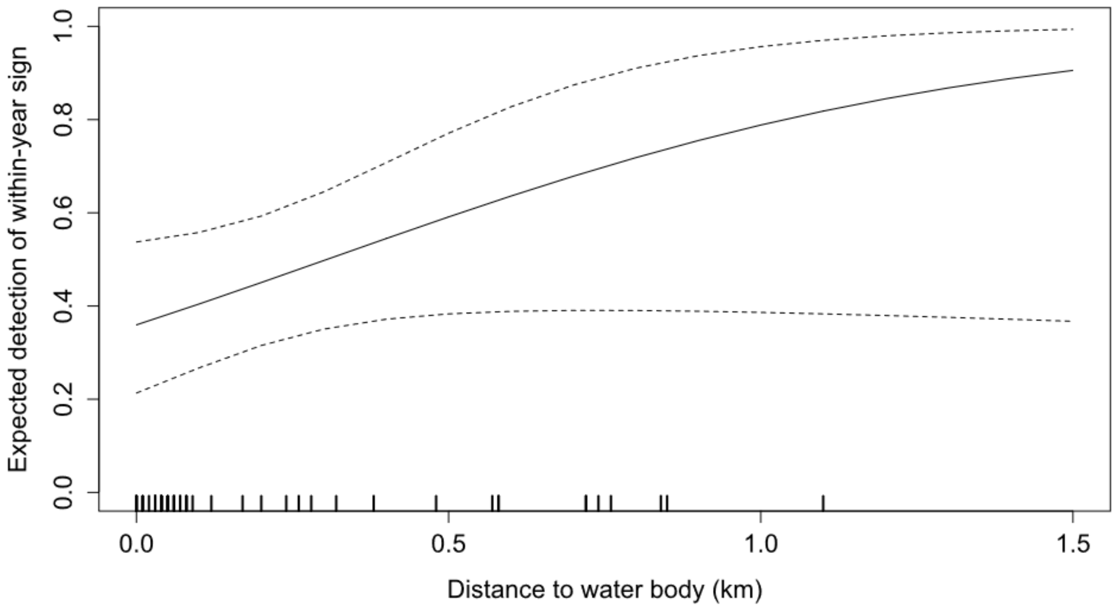

Expected detection of within-year sign
0.0
0.2
0.4
0.6
0.8
1.0
0.0
0.5
1.0
1.5
Distance to water body (km)

Figure 5: 24-hour activity cycle of sun bears (*Helarctos malayanus*) during 2010 – 2015 from camera traps in the lower Kinabatangan, Sabah, Malaysia. Time is centered at midnight (0:00). Vertical lines at x-axis represent data points.

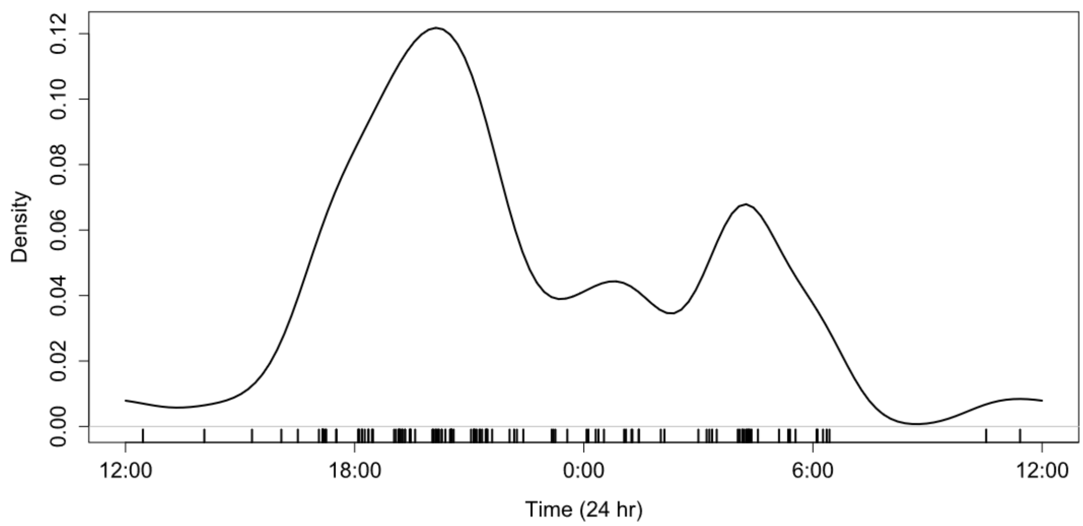
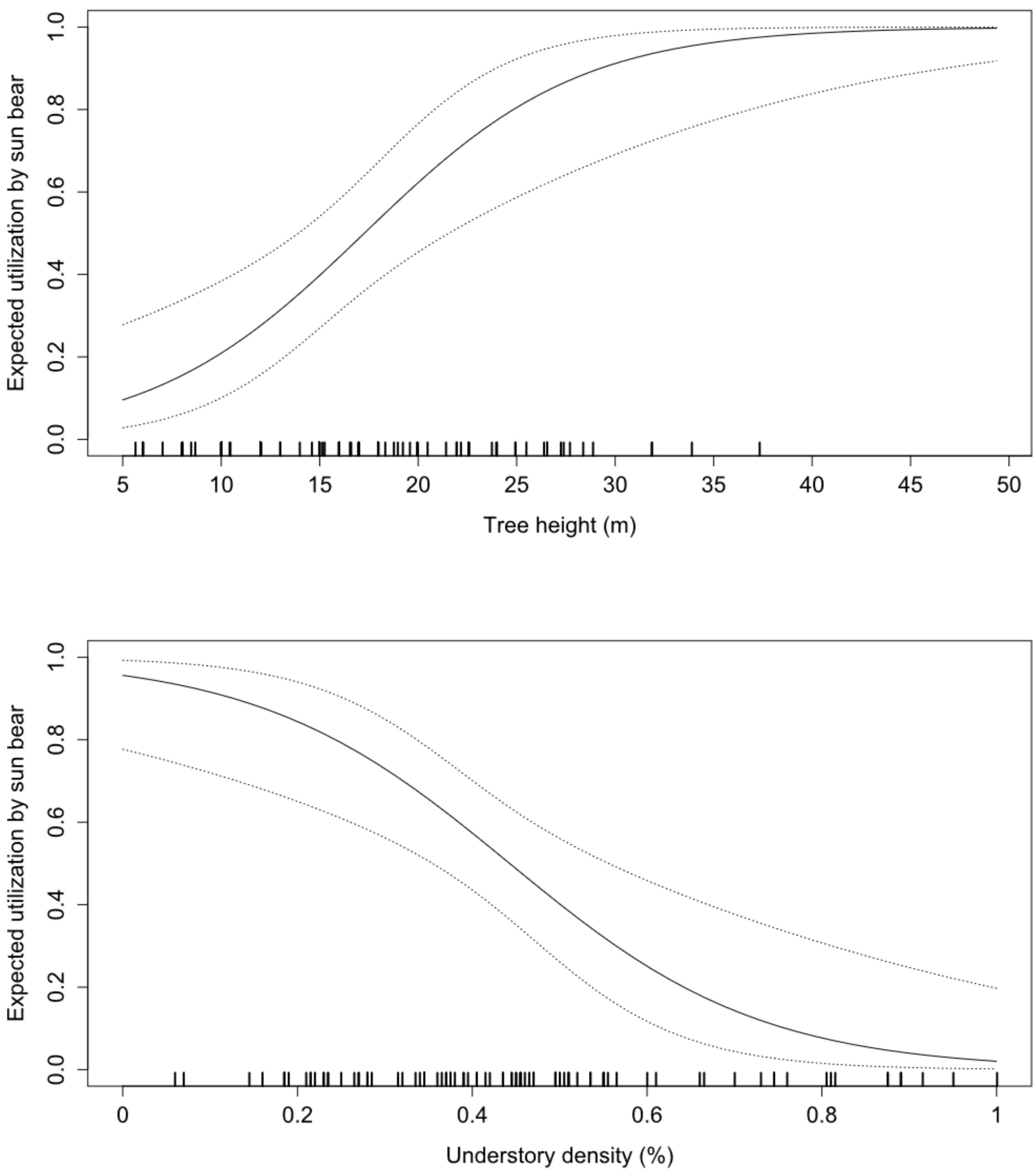


Figure 6: Expected selection of climbing trees by sun bears (*Helarctos malayanus*) as a function of tree height (m; top) and understory density (%; bottom) with 95% confidence intervals in the lower Kinabatangan, Sabah, Malaysia according to the best supported logistic regression model. Vertical lines at x-axis represent data points.



8. Supplementary information

Table S1: Mean sign (claw marked trees) densities of the three locations surveyed for sun bear sign in the lower Kinabatangan, Sabah, Malaysia.

Site*	Mean all aged sign density (ha^{-1})	Mean within-year sign density (ha^{-1})
LKWS Lot 5	5.8	2.0
LKWS Lot 6	8.3	4.4
Pin Supu Forest Reserve	135	110

*Lower Kinabatangan Wildlife Sanctuary, LKWS.

CHAPTER 2

How do sun bears (*Helarctos malayanus*) rank among the most destructive and dangerous wildlife species in the lower Kinabatangan oil palm landscape, Sabah, Malaysian Borneo?

Summary Due largely to the expansion of oil palm, forest fragmentation has occurred on a large scale in Borneo. There is much concern on the ability of forest dependent species, such as the sun bear (*Helarctos malayanus*), to persist in this radically modified landscape. The absence of sufficient natural food in forest fragments might drive sun bears into oil palm plantations, where they risk coming into conflict with people. We interviewed oil palm plantation workers and farmers in the lower Kinabatangan region of Sabah, Malaysian Borneo to identify if sun bears were utilizing plantations, if bear depredation caused damage to the crop, and how bears were viewed by people. To get a comparative baseline, we extended these questions towards other species that were known to use plantations. Our results showed that bears were hardly encountered in plantations and were not considered destructive to the oil palm crop, though they were feared as a dangerous species. Other species, such as macaques, wild pigs, and elephants, had much more destructive feeding habits compared to sun bears. Interestingly, sun bears have the opportunity to use this readily available resource without retribution from people. The lack of negative perceptions towards sun bears coupled with bears avoiding people suggest that this species is well adapted to surviving in the forest-oil palm interface now dominating much of Borneo.

Keywords Borneo, *Helarctos malayanus*, human-wildlife conflict, oil palm, Sabah

Introduction

Malaysia is one of the world's two largest producers of palm oil, the product derived from the oil palm crop (Koh & Wilcove 2008). Development of this crop has led to an increase in revenue for Malaysia's developing economy (Basiron 2007). However, large scale conversion has come at a price to biodiversity: between 1995 and 2005, an estimated 55–59% of Malaysian oil palm plantation land was acquired through forest conversion (Koh & Wilcove 2008). Currently, the State of Sabah, Malaysian Borneo, holds around 28% of the country's total oil palm planted area (Malaysian Palm Oil Board

2014). The eastern part of Sabah, already copious with oil palms, seems likely to see future oil palm expansion because of high yield potential (Abram et al. 2014).

While beneficial to the economies of developing countries, oil palm agriculture has overall negative impacts on biodiversity (Maddox et al. 2007, Koh & Wilcove 2008, Meijaard et al. 2011a, Luskin et al. 2014, Jennings et al. 2015). Wildlife that enter plantations are at risk of being hunted (Luskin et al. 2014, Azhar et al. 2014), while others consume crops and are subject to retribution by people (Lee 2002, Meijaard et al. 2011a, Luskin et al. 2014, Azhar et al. 2014). As the area of oil palm increases at the expense of natural forest, there is a growing concern that some species will not be able to persist (Ancrenaz et al. 2015).

Oil palm plantations can be nutritionally poor for some species like orangutans (Campbell-Smith et al. 2011), whereas other species like wild pigs and smaller mammals can reach high densities in plantation landscapes, taking advantage of the super-abundance of fruits, succulent greens, and crop pests (Rajaratnam et al. 2007, Luskin et al. 2014, Jennings et al. 2015). Some mammal species have been documented using oil palm landscapes as movement corridors (Campbell-Smith et al. 2011, Alfred et al. 2012, Estes et al. 2012) as well as feeding and resting sites (Nakashima et al. 2013, Ancrenaz et al. 2015).

Sun bears (*Helarctos malayanus*) are a forest dependent species. Previous research on Borneo has shown that sun bears are sensitive to extreme variation in annual mast fruiting events, with some bears starving during long intermast intervals (Wong et al., 2005; Fredriksson et al., 2007) and others using agricultural land adjacent to forests to supplement their diet (Normua et al. 2004b, Fredriksson 2005b). With oil palm plantations now bordering many forested areas in Borneo, this crop has potential to become a food source for bears, especially during periods of low natural food availability. However, increased reliance on agriculture for food often comes with the added risk of conflicts and persecution (Fredriksson 2005b, Liu et al. 2011, Scotson et al. 2014). Indeed, the earliest accounts of sun bears described them as destructive of coconut palms (Horsefield 1825).

Interview and questionnaire surveys have been successfully used to gauge perceptions and attitudes towards wildlife in oil palm plantations (Meijaard et al. 2011a, Luskin et al. 2014, Azhar et al. 2014, Ancrenaz et al. 2015). In this study, we utilized interview surveys to understand the drivers influencing how wildlife are perceived by a certain segment of stakeholders in the oil palm industry: oil palm plantation workers and small scale farmers. Previous surveys in this region have primarily focused on primates (Campbell-Smith et al. 2010, Meijaard et al. 2011a, Ancrenaz et al. 2015), with only a few covering multiple species of conservation concern (Luskin et al. 2014, Azhar et al. 2014).

Our primary intent was to obtain information on the use of plantations by sun bears, the degree to which their feeding damaged crops, and if their presence was viewed as a threat to people. Conflicts with bears in agricultural landscapes have been escalating worldwide, resulting in a growing conservation concern (Can et al. 2014). We hypothesized that the expansion of oil palm would have the compounded effect of habitat loss and active retaliation against sun bears. We collected information on all wildlife species reported to use plantations in order to gauge sun bears with respect to other crop depredators.

Study area

Our study area was the lower Kinabatangan floodplain situated in the Malaysian State of Sabah on the island of Borneo. The floodplain consists of 10 variably sized forest fragments (known as lots) collectively designated as the Lower Kinabatangan Wildlife Sanctuary (LKWS) and 7 Virgin Jungle Reserves (Ancrenaz et al. 2004) covering an estimated 45,000 ha. Some forested areas within this protected area network are linked to each other through narrow riparian corridors. The forest types represented in the lower Kinabatangan include mangrove forest, *Nipa fruticans* swamp (locally known as *nipah*), freshwater swamp forest, peat swamp forest, dry lowland forest, limestone forest, and severely degraded areas (Abram et al. 2014). Much of the original forest landscape has been altered by logging and agriculture, beginning in the 1950's (Azmi 1998). Prominent wildlife species found in the lower Kinabatangan include the Bornean orangutan (*Pongo pygmaeus*), Bornean elephant (*Elephas maximus borneensis*), proboscis monkey (*Nasalis* *larvatus*), Sunda clouded leopard (*Neofelis diardi*), sun bear, bearded pig (*Sus barbatus*), muntjac (*Muntiacus spp.*), sambar (*Rusa unicolor*), and small carnivores (*Viverridae*, *Mustelidae*, and *Herpestidae* families).

Methods

Data collection

We interviewed oil palm plantation workers and village landowners (hereafter respondents) in June 2013 and during May–October 2014 in a section of the lower Kinabatangan (Figure 1). Respondents were mostly males involved in field operations. We targeted two different plantation types: large industrial estates run by companies (hereafter estates) and small farms (known locally as *kebun*) owned by villagers. At each plantation we obtained information on the size of the total planted area (hereafter plantation size), the presence of palms less and more than three years old (hereafter immature palms and mature palms, respectively), and if the plantation bordered intact forest (hereafter border).

We collected basic demographic information from each respondent, namely their age, ethnicity, religion, and length of time working in the plantation (hereafter time, in four categories: <1 year, 1–5 years, 5–10 years, >10 years). We asked interviewees to identify wildlife (mammals and snakes) using images of certain protected and totally protected species in Sabah (WWF-Malaysia, 2013). We did not include questions on birds, small mammals (e.g. squirrels, rats, and bats), and monitor lizards, but recorded these when respondents provided information on them. We did not tell respondents that we were specifically interested in sun bears.

To minimize the social desirability bias induced by respondents wanting to portray themselves favorably (Fisher 1993), we asked very few sensitive questions about hunting and killing of wildlife. The few sensitive questions that we did ask were kept towards the end of the interview (Brace 2008). We asked respondents to rate how often they saw specific species (rarely or commonly), where this happened (within plantation, plantation forest border, and/or secondary forest within plantations) and at what time the encounter(s) took place (morning, afternoon, and/or night).

We asked whether the observed species fed on loose palm fruits scattered on the ground (hereafter loose fruits), uncollected fruit bunches that were harvested and left on the ground (hereafter fruit bunches), fruits on the palm trees (hereafter palm fruits), and/or oil palm shoots (hereafter palm shoots). We asked respondents to identify species that were destructive (yes or no) towards oil palms as a result of their feeding habits.

We also asked respondents whether a species was dangerous (yes or no) and if they answered yes, we asked them to rate this danger qualitatively (least dangerous, dangerous, or extremely dangerous). We asked respondents to detail their reaction to encounters with dangerous species: did they retreat, chase the animal away, trap the animal, and/or kill it? We ended each interview by asking respondents how they felt about hunting and protecting wildlife. We removed responses that clearly (or most likely) did not occur in plantations. However, as we had a small sample size, we did not conduct detailed discrimination between high and low quality respondents or responses (Meijaard et al. 2011b)

Data analysis

We checked for differences among respondent ages and plantation sizes between estates and *kebun* using Wilcoxon-Mann-Whitley tests. We compared the number of plantations with immature and mature palms separately between plantation types using Fisher's exact tests. We did not include squirrels, birds, and monitor lizards in summaries, non-parametric tests, and analyses regarding wildlife encounters. We grouped seven species into three different groups for simplicity: macaques (*Macaca fasicularis, M. nemestrina*), snakes (*Naja sumatrana*, *Python reticulatus, P. curtus*), and civets (*Viverra tangalungga, Paradoxurus hermaphroditus*). We calculated the mean commonness (1=rare, 2=common), destructiveness (0=yes, 1=no), and danger level (0=not dangerous, 1=least dangerous, 2=dangerous, 3=extremely dangerous) of each species, in order to rank them (Marchal and Hill 2009). We calculated the proportions of different locations and times of day that respondents encountered wildlife. We used Fisher's exact test to assess the difference in the total encounters of each species between estates and *kebun*. We conducted all analyses in R (R Core Team 2015).

We separated all species into two groups based on body size as we predicted larger animals might be perceived as more destructive and dangerous, as well as more visible. All large bodied mammals (> 15kg; hereafter large mammals) were placed in one group while smaller bodied mammals together with snakes (hereafter small wildlife) were placed in another. We summed the number of species in each group per individual respondent. We compared counts of small wildlife and large mammals between respondents from the two plantation types and borders (plantations with intact forest borders and those without) using Wilcoxon-Mann-Whitley tests.

We fit Poisson generalized linear models (glms) for both groups of wildlife. The response variable was the total count of either small wildlife or large mammals encountered per respondent. We included the binary variables border, immature, mature, the categorical variable plantation type, and continuous variable plantation size as plantation level predictors. We also included the categorical variable time and continuous variable age as respondent level predictors. We began by fitting single covariate models and subsequently adding predictors that were present in models with the lowest Akaike's Information Criterion corrected for small sample sizes ($\Delta AIC_c < 2$) scores in each successive step. We selected top ranked models based on AIC_c and model weights. We ignored competing models if they only contained one extra variable to a better supported model (Arnold 2010). We checked all top ranked models for overdispersion by dividing model residual deviance by degrees of freedom. We assessed multicollinearity between predictor variables in models using generalized variance inflation factor (GVIF) values. We judged model fit visually by plotting the residuals against the fitted values for each model. There was a possibility that respondents from the same plantation would have correlated observations. To account for this potential non independence between respondents, we fit Poisson generalized linear mixed models (glmms) with plantation as the random intercept. We then compared the model coefficients and 95% confidence intervals from the glms and glmms.

For all summaries, non-parametric tests, and models regarding wildlife feeding habits, we included observations of birds and squirrels as our primary objective was to understand food resource use and perceived destructiveness in plantations. We used chi-squared tests to compare the number of wildlife depredations on loose fruits, fruit bunches, palm fruits, and palm shoots between plantation types. We used the same test to compare the number of destructive and non-destructive observations for each of these food resources as well as between estates and *kebun*.

For model fitting, we removed observations that were solely of respondents getting second hand information about depredations and destructive feeding habits. To model the effects of feeding behaviors and plantation characteristics on destructiveness, we fit binomial glms with destructive behavior as the binary response variable and the binary predictor variables loose fruits, fruit bunches, palm fruits, palm shoots, immature, and mature. We also included plantation type as a categorical predictor variable. We fit and selected models similar to the Poisson glms to find the best explanatory models. For all top ranked models ($\Delta AIC_c < 2$), we checked for multicollinearity in our predictor variables using GVIF values. We assessed model fit by visually inspecting the binned residual versus fitted plot of the model. We used the area under the receiver operating characteristics (ROC) curve to discern the model's predictive power. We also fit a binomial glmm with plantation as the random intercept to compare model coefficients and 95% confidence intervals between glms and glmms.

We used Fisher's exact test to examine whether the number of records for each dangerous species differed between plantation types. We further assessed if there were differences in the number of wildlife that were perceived to be least dangerous, dangerous, and extremely dangerous by respondents between estates and *kebun* using Fisher's exact test.

Results

We conducted interviews in 27 plantations of which 17 were *kebun* and 10 were estates. We interviewed the one or two people that managed and worked on each *kebun*. In estates we typically spoke to most of the harvesting and field operations staff. In total, we interviewed 118 respondents, 19 from *kebun* and 99 from estates. The ethnic makeup of our respondents included Buginese (54.2%), Orang Sungai (22%), Kadazandusun (4.2%), various Philippine ethnic groups (8.5%; Tausug, Bajau Laut, and Visayan), and various other Indonesian ethnic groups (11%; Javanese, Kaili, and Timorese). Respondents were mostly Muslim (85.6%); the rest were Christian (14.4%). The largest portion (45.8%) of respondents had worked in their plantation for over 10 years; 16.1% had worked 6–10 years, 26.3% for 1–5 years, and 9.3% for less than a year. Most respondents felt that protecting wildlife was necessary (93.2%). Some (29.7%) respondents felt that they should be allowed to hunt, while a larger number (64.4%) felt this was not necessary (the rest did not answer). More than half the respondents (53.7%) felt they should be given the chance to keep wildlife in captivity.

The mean estate size was 1,389 ha (SD = 766.6 ha) and the mean *kebun* size was 3 ha (SD = 1.7 ha). Respondents from estates had a mean age of 34.5 (SD = 9.8) years while those from *kebun* had a mean age of 42.8 (SD = 13.9). These differences in the plantation sizes (W = 170, 95% CI 740.3 – 1888.7, $P < 0.0001$) and respondent ages (W = 581.5, 95% CI: -15 – -2, P = 0.01) between estates and *kebun* were significant. Most plantations (77.8%) had only mature palms, 7.4% had only immature palms, and 14.8% had a combination of both. Estates and *kebun* did not differ in this regard.

Respondents encountered 24 different species, excluding small mammals, birds, and monitor lizards. Most (57%) encounters occurred within the oil palm plantation, while another 40.8% occurred at the border of forest and plantation and 2.2% in secondary forest patches and riparian areas. Most encounters took place in the morning (42.5%), with some also in the afternoon and evening (34 %), and at night (23.5%). There were more encounters of each species in estates compared to *kebun* (P = 0.04).

Respondents ranked the sun bear as rarely encountered within plantations (Table 1). They were somewhat more commonly encountered than Sunda clouded leopards and Sunda pangolins (*Manis javanica*), but less commonly encountered than Bornean elephants, orangutans, and proboscis monkey. The most commonly reported species were macaques (*Macaca nemestrina*, *M. fasicularis*), bearded pigs, civets (*Viverra tangalungga*, *Paradoxurus hermaphroditus*), and leopard cats (*Prionailurus bengalensis*).

The mean number of small wildlife species encountered was higher in plantations bordering intact forest (5.53, SD = 2.69) than in plantations without adjoining forest ($\bar{x}$ = 4.03, SD = 2.22; W = 912.5, 95% CI -3 – -0.00004, P = 0.01). For large bodied mammals, the number of encountered species was similar for plantations bordering ($\bar{x}$ =

2.66, SD = 1.58) and not bordering forest ($\bar{x}$ = 2.16, SD = 1.8). Encounter rates per person were not significantly different for respondents on estate plantations and *kebun* for either small wildlife ($\bar{x}$ = 5.27, SD = 2.7; $\bar{x}$ = 4.37, SD = 2.22, respectively) or large mammals ($\bar{x}$ = 2.56, SD = 1.71; $\bar{x}$ = 2.32, SD = 1.29, respectively).

The top ranked Poisson glms ($\Delta AIC_c < 2$) for encounters with small wildlife contained the predictors age, immature, border, and time (Table 2). Overdispersion parameters for the best supported models were all under 1.5, suggesting the Poisson distribution was accurate for our data. GVIF values for the models were less than 3, indicating that multicollinearity of covariates was not significant. Residuals versus fitted plots displayed a good fit. The glmm for small wildlife failed to converge, suggesting that the glm was appropriate for our data.

For large mammals, top ranked models ($\Delta AIC_c < 2$) contained the predictors immature, mature, border, and age (Table 3). All overdispersion parameters for the best supported models were less than 1.5, supporting use of the Poisson distribution. GVIF values indicated that multicollinearity between predictors was not significant and residuals versus fitted plots displayed decent fit for all models. The large mammal glmm was not appropriate for our data as there was no variation between the random components (SD = 0).

Respondents identified eight species (excluding squirrels and birds) that fed on loose palm fruits, four that fed on harvested fruit bunches, and eight that fed on fruits still on the palm (Figure 2). Respondents identified another seven species that fed on palm shoots (Figure 2). Many observations from respondents regarding wildlife feeding behavior included visual observations (74.5%); some included respondents finding feeding sign (21.5%), and a few respondents (4%) heard about crop damage from other people.

We found a greater number of palm shoot depredations in estates compared to *kebun* (X^2 = 5.58, P = 0.02). Depredations of loose fruits, fruit bunches, and palm fruits were similar for *kebun* and estates. Respondents considered nine species to be destructive to the oil palm crop, but there was great variation among these (Figure 3). Compared to macaques, pigs, and elephants, sun bears caused little damage (Table S1). Species that destroyed palm shoots were more likely to be labelled as destructive (X^2 = 10.9, P = 0.001).

Model selection for the binomial glms identified top ranked models that all contained palm shoots as a predictor of destructive behavior ($\Delta AIC_c < 2$; Table 4). Other variables included in the top ranked models were plantation type and bunch. GVIF values did not exceed 3, suggesting that multicollinearity was not significant. Binned residual versus fitted plots for all models suggested a good fit. Area under the ROC curve for all models was 0.6 – 0.7, indicating adequate predictive power of all models. We were not able to compare these results to those of the binomial glmm, as it failed to reach convergence.

Eight species groups were considered to be dangerous (n = 84 respondents; Table 5). Among these, clouded leopards and crocodiles were considered to be most dangerous. Sun bears were perceived to be as dangerous as orangutans. Respondents were able to recount only one clouded leopard mauling and three sun bear maulings that had occurred in plantations. For all dangerous species, we found no significant difference between the number of records from estates and *kebun*. The number of least dangerous, dangerous, and extremely dangerous records was similar for plantation types. Most respondents stated that retreating from a dangerous species (58.3%) was their normal response, while 34.6% said they would sometimes chase the animal away. Very few indicated that they would catch or trap a dangerous animal (1.9%), or kill it (5.1%).

Discussion

Most species encountered in oil palm plantations tend to be dietary generalists and favor more open habitats (Chung et al. 2000, Aratrakorn et al. 2006). This was echoed in our findings, with macaques and bearded pigs being the two most common mammals encountered by respondents. Both species made use of multiple food resources in plantations. Snakes, leopard cats, and civets were also commonly encountered. These species can persist in plantations because of the high abundance of rodents, their principal prey (Wood & Fee 2003, Rajaratnam et al. 2007, Nakashima et al. 2013, Jennings et al. 2015). Sun bears however were rarely encountered, which suggests either avoidance of monocultures with sparse understory or behaviors that avoid coming into contact with people.

Respondents from plantations containing immature oil palms encountered more small wildlife and large mammal species than those from plantations without this feature. Wildlife may be more visible in young plantations, and the young palms may be more readily consumed. In addition, large mammal encounters were more associated with plantations that had both immature and mature palms. This likely relates to better cover and habitat heterogeneity, which would be attractive to wildlife (Tews et al. 2004, Peh et al. 2006).

Respondents in plantations bordering intact forests encountered a greater number of both small wildlife and large mammal species. The presence of nearby intact forest increases species richness in croplands (Fitzherbert et al. 2008). Oil palm plantations may represent population sinks for native fauna (Azhar et al. 2014) and intact forests represent source populations for these wildlife to continue persisting in the plantation landscape (Peh et al. 2006). Furthermore, certain forest dwelling species have been documented having home ranges that encompass both oil palm plantations and adjacent intact forest (Normua et al. 2004b, Cheah 2013b, Nakashima et al. 2013).

Older respondents had encountered more species, probably for two reasons. First, they typically worked at the plantation for a longer time. Indeed, for small wildlife, there was a positive association between length of time working in a plantation and the number of species encountered. Second, some species have likely declined in abundance over time with diminishing forest (Maddox et al. 2007) so were less apt to be seen by younger respondents. Indeed, a few older respondents lamented that they encountered more wildlife species and with more frequency in the past.

Bornean elephants, porcupines (*Hystrix sp.*), macaques, and bearded pigs were all perceived to be particularly destructive of oil palm crops. Other studies have also identified these species as destructive (Sabah Wildlife Department 2010, Luskin et al. 2014, Azhar et al. 2014). All fed on palm shoots, which emerged as a strong variable in our modelling of destructive behavior. Feeding on the shoots of young palms often results in severe damage to the palm, sometimes killing it. Newly planted palms are especially susceptible to being fed on, resulting in monetary loss for the planters. Plantation workers from one estate showed us the damage caused by elephants that uprooted mature palms to obtain the succulent palm shoot. Feeding on fruit bunches also had a strong effect on destructiveness. The two species that utilized this resource heavily were bearded pigs and macaques, both very common in plantations. Fruit bunches are already ripe and destined for oil palm mills, so any loss is highly undesirable in plantation operations.

Kebun owners and workers were more likely to view wildlife as destructive because monetary losses caused by crop depredations can be extremely burdensome on these small scale farmers. Additionally, whereas estates often use a combination of ditches and electric fences at the plantation forest border to keep out elephants (Alfred et al. 2012), these are too costly to erect and maintain for most *kebun* owners. Large estates even have specialized units on call to deal with elephant encroachments, using a variety of non-lethal methods. Respondents to our survey indicated more overall damage to oil palms than Ancrenaz et al. (2015) found in an interview survey with estate workers in the same general area. However, their study focused on orangutans in plantations with mature oil palms, where orangutans often built nests but cause minimal damage to the palms.

Three species ranked high in terms of both destructiveness and danger: elephants, macaques, and bearded pigs (Figure 4). Among these, elephants are of high conservation concern. This species is also at the forefront of human wildlife conflict in Sabah (Lee 2002, Sabah Wildlife Department 2010). Sun bears were negligibly destructive but were often feared even though just a handful of human maulings occurred in our study area. In other parts of their range, sun bear attacks on people are much more common (Sethy & Chauhan 2013).

Very few respondents had directly observed bears feeding in plantations. In Krau Wildlife Reserve, Peninsular Malaysia, GPS-collared sun bears made frequent night incursions into oil palm plantations. These bears were some of the heaviest recorded from the wild (Cheah 2013b). It may be that sun bears in our study area simply avoided detection by people when entering plantations. Camera trap records in the adjacent forest in the lower Kinabatangan indicated that sun bears were active mainly during crepuscular and nocturnal hours (when human presence in plantations is minimal; Guharajan, in prep) and were in very good physical condition (Guharajan, unpublished data). This nocturnal strategy for feeding on crops has been observed in other populations of sun bears (Sethy & Chauhan 2013, Wong et al. 2015). Although respondents to our survey sometimes saw bears feeding in palm trees, we suspect that they also fed on fruits on the ground (as they do in the forest; Fredriksson, 2012), but were less visible when they did so. Indeed, a solitary species like the sun bear may simply cross into plantations undetected at night and feed on the abundant fruits close to forest (Figure 5). In contrast to other studies (Fredriksson 2005b, Scotson et al. 2014, Wong et al. 2015), none of the respondents from our study witnessed sun bear crop depredation through the identification of feeding sign.

We were surprised that respondents tended not to view bears as destructive to oil palms, as other studies have found them to be destructive to a variety of other crops (Fredriksson 2005b, Sethy & Chauhan 2013, Scotson et al. 2014, Wong et al. 2015). Indeed, most other species of bears do damage crops (Mattson 1990, Maddrey & Pelton 1995, Jorgenson & Sandoval-A 2005, Charoo et al. 2011, Northrup et al. 2012, Ditmer et al. 2016), making human–bear conflicts related to agriculture a growing conservation concern (Northrup et al. 2012, Can et al. 2014). We had presumed that oil palm plantations not only reduced the area of natural forest, but also increased the mortality risk to bears. Sun bears in a forest-oil palm interface in Peninsular Malaysia had a high degree of snare injures (Cheah 2013b). Through camera trap photographs, we did not find evidence of this in our study area (Guharajan, in prep), nor did respondents report it. However, recent discoveries of disemboweled sun bear carcasses with missing paws (L. Liman, WWF-Malaysia, pers. comm., S.T. Wong, pers. obs.) suggests that poaching of bears for traditional medicine and delicacies is ongoing within the greater floodplain area, though the scale is yet unclear.

Sun bears likely gain nutritionally by eating oil palm fruits, especially in years when wild fruits are scare in the forest. But unlike the destructive feeding of bears in other crops, sun bears can feed on oil palm fruits without damaging the tree. That behavior, combined with mainly nocturnal feeding, increases the ability of sun bears to persist in this radically-modified landscape.

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Table 1: Mean commonness ranks (± SD) and total number of records of mammals and reptiles encountered by respondents from oil palm plantations in the lower Kinabatangan, Sabah, Malaysia.

Species	**Mean Commonness Rank[1] (± SD)**	**Total records[2]**
Sunda clouded leopard *Neofelis diardi*	1.00 (± 0.00)	2
Sunda pangolin *Manis javanica*	1.05 (± 0.22)	27
Muntjac *Muntiacus spp.*	1.09 (± 0.30)	13
Sambar *Rusa unicolor*	1.17 (± 0.38)	32
Sun bear *Helarctos malayanus*	1.25 (± 0.46)	8
Porcupine *Hystrix spp.*	1.29 (± 0.46)	40
Malay badger *Mydaus javanicus*	1.31 (± 0.47)	29
Mousedeer *Tragulus spp.*	1.31 (± 0.48)	16
Western tarsier *Tarsius bancanus*	1.33 (± 0.52)	6
Bornean elephant *Elephas maximus borneensis*	1.33 (± 0.47)	57
Bornean orangutan *Pongo pygmaeus*	1.34 (± 0.48)	52
Proboscis monkey *Nasalis larvatus*	1.38 (± 0.49)	32
Bornean slow loris *Nycticebus menangensis*	1.40 (± 0.55)	6
Bornean Mueller's gibbon *Hylobates muelleri*	1.47 (± 0.52)	16
Smooth otter *Lutrogale perspicillata*	1.48 (± 0.50)	49
Snake *Python spp., Naja sumatrana*	1.48 (± 0.50)	133
Colugo *Galeopterus variegatus*	1.50 (± 0.71)	2
Leopard cat *Prionailurus bengalensis*	1.58 (± 0.50)	38
Civet *Viverra tangalungga, Paradoxurus hermaphroditus*	1.59 (± 0.50)	84
Estuarine crocodile *Crocodylus porosus*	1.60 (± 0.52)	11
Bearded pig *Sus barbatus*	1.77 (± 0.42)	98
Macaque *Macaca spp.*	1.94 (± 0.23)	151

[1]Ranks: rarely encountered, 1; commonly encountered, 2.
[2]Not all records were used in the calculation of the mean rank as not all respondents that encountered a species ranked it.

Table 2: Top ranked models ($\Delta AIC_c < 2$) for small wildlife (small bodied mammals and snakes) encountered by respondents from oil palm plantations in the lower Kinabatangan, Sabah, Malaysia with number of parameters (k), log likelihood, Akaike's information criterion adjusted for small sample sizes (AIC_c), change in AIC_c (ΔAIC_c), and Akaike weight.

Model[1]	**k**	**Log likelihood**	**AIC_c**	**ΔAIC_c**	**Akaike weight**
Immature + IF + Age	4	-259.07	526.51	0.00	0.24
Immature + Mature + IF + Age[2]	5	-258.24	527.03	0.52	0.18
Immature + Type + IF + Age[2]	5	-258.56	527.67	1.16	0.13
Immature + IF + Age + Size[2]	5	-258.62	527.79	1.29	0.12
Immature + IF + Time	6	-257.55	527.88	1.37	0.12
Immature + IF	3	-260.85	527.93	1.42	0.12
Immature + Mature + IF[2]	4	-260.06	528.49	1.99	0.09
Intercept only	1	-270.34	542.71	16.21	0.00

[1]Covariates: oil palms less than three years of age, Immature; oil palms more than three years of age, Mature; plantation type, Type; bordering intact forest, IF; respondent age, Age; total planted area, Size; length of time respondent worked in plantation, Time.

[2]Models with an additional parameter within $\Delta AIC_c \leq 2$ of an otherwise similar better ranked model were not considered competitive despite having strong support. The extra parameter represents noise and thus does not necessarily infer biological significance.

Table 3: Top ranked models ($\Delta AIC_c < 2$) for large mammals encountered by respondents from oil palm plantations in the lower Kinabatangan, Sabah, Malaysia with number of parameters (k), log likelihood, Akaike's information criterion adjusted for small sample sizes (AIC_c), change in AIC_c (ΔAIC_c), and Akaike weight.

Model[1]	**k**	**Log likelihood**	**AIC_c**	**ΔAIC_c**	**Akaike weight**
Immature + Mature + IF + Age	5	-202.98	416.52	0.00	0.33
Immature + IF + Age	4	-204.27	416.91	0.39	0.27
Immature + Mature + IF + Age + Size[2]	6	-202.72	418.22	1.70	0.14
Immature + Mature + Age	4	-204.97	418.31	1.79	0.13
Immature + Type + IF + Age[2]	5	-203.98	418.51	1.99	0.12
Intercept only	1	-211.03	424.10	7.57	0.01

[1]Covariates: oil palms less than three years of age, Immature; oil palms more than three years of age, Mature; plantation type, Type; bordering intact forest, IF; respondent age, Age; total planted area, Size.

[2]Models with an additional parameter within $\Delta AIC_c \leq 2$ of an otherwise similar better ranked model were not considered competitive despite having strong support. The extra parameter represents noise and thus does not necessarily infer biological significance.

Table 4: Top ranked models for wildlife destructiveness in oil palm plantations in the lower Kinabatangan, Sabah, Malaysia Malaysia with number of parameters (k), log likelihood, Akaike's information criterion adjusted for small sample sizes (AIC_c), change in AIC_c (ΔAIC_c), and Akaike weight.

Model[1]	**k**	**Log likelihood**	**AIC_c**	**ΔAIC_c**	**Akaike weight**
Bunch + Type + Shoot	4	-137.11	282.40	0.00	0.24
Type + Shoot	3	-138.38	282.87	0.47	0.19
Bunch + Type + Mature + Shoot[2]	5	-136.88	284.04	1.63	0.10
Type + Mature + Shoot[2]	4	-138.00	284.20	1.79	0.10
Bunch + Type + Palm + Shoot[2]	5	-136.97	284.22	1.81	0.10
Bunch + Immature + Type + Shoot[2]	5	-136.97	284.22	1.82	0.10
Bunch + Shoot	3	-139.06	284.24	1.84	0.09
Bunch + Type + Loose + Shoot[2]	5	-137.03	284.34	1.93	0.09
Intercept only	1	-145.81	293.63	11.23	0.00

[1]Covariates: loose oil palm fruits, Loose; harvested fruit bunches, Bunch; fruits on the oil palm, Palm; oil palm shoots, Shoot; oil palms over three years of age, Mature; oil palms less than three years of age, Immature; plantation type, Type.

[2]Models with an additional parameter within $\Delta AIC_c \leq 2$ of an otherwise similar better ranked model were not considered competitive despite having strong support. The extra parameter represents noise and thus does not necessarily infer biological significance.

Table 5: Mean danger level ranks of wildlife species according to respondents from oil palm plantations in the lower Kinabatangan, Sabah, Malaysia.

Species	**Mean danger level rank (± SD)[1]**	**Total records[2]**
Macaque ***Macaca spp.***	1.90 (0.74)	14
Bearded pig ***Sus barbatus***	1.94 (0.77)	18
Bornean elephant ***Elephas maximus borneensis***	2.18 (0.8)	25
Sun bear ***Helarctos malayanus***	2.33 (1.15)	9
Bornean orangutan ***Pongo pygmaeus***	2.33 (0.71)	11
Snake ***Python spp., Naja sumatrana***	2.36 (0.68)	63
Estuarine crocodile ***Crocodylus porosus***	3.00 (0.00)	10
Sunda clouded leopard ***Neofelis diardi***	3.00 (0.00)	5

[1]Ranks: least dangerous, 1; dangerous, 2; extremely dangerous, 3.

[2]Not all records were used in the calculation of the mean rank as not all respondents that encountered a species ranked it.

Figure 1: Locations of oil palm plantations where interviews took place in the lower Kinabatangan, Sabah, Malaysia and location of the study area on Borneo (inset). Respondents were from Tong Lim Enterprise (1), FGV-Pontian Hillco Estate (2), Global Enterprise Gomantong Estate (3), FGV-Pontian Kuril, Pendirosa, Subok, Fico, and Orico estates (4), Sawit Kinabalu Sungai Pin Estate (5), Linddale Estate (6), and various farms owned by villagers of Kg. Menggaris 1 and 2, Kg. Batu Putih, and Kg. Perpaduan Datuk Moh (7).

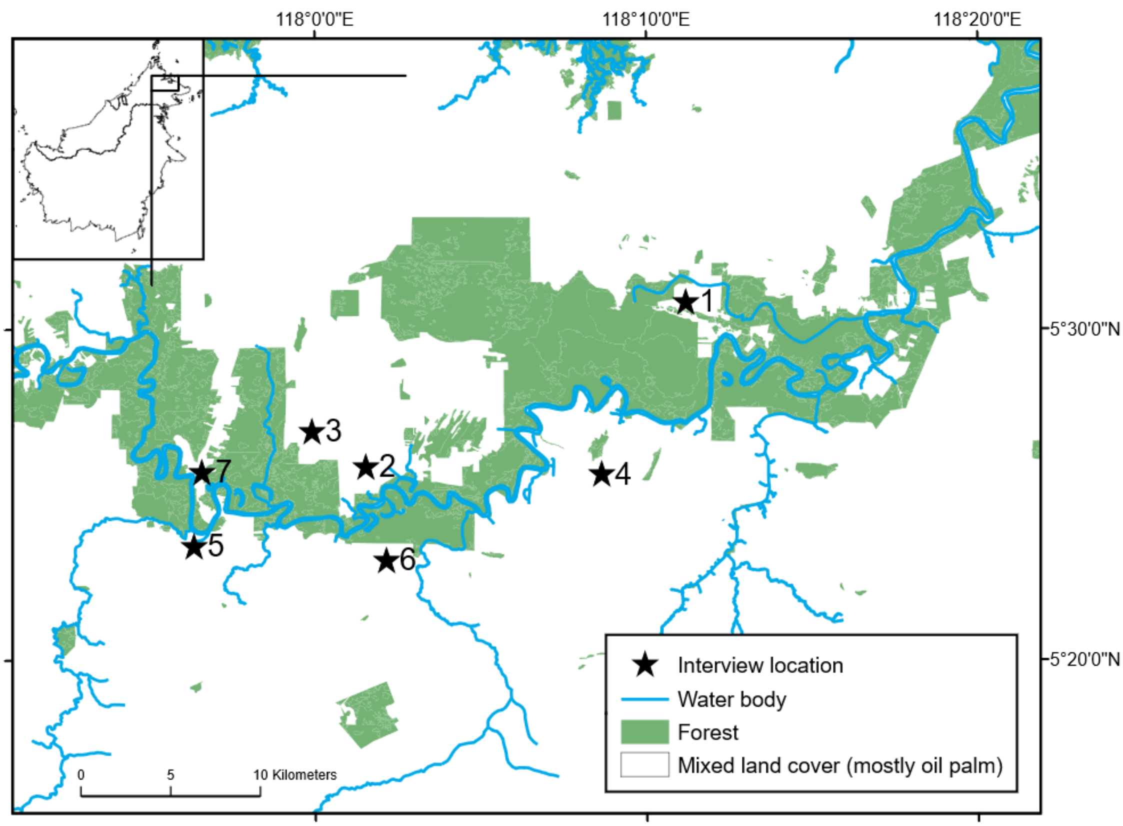
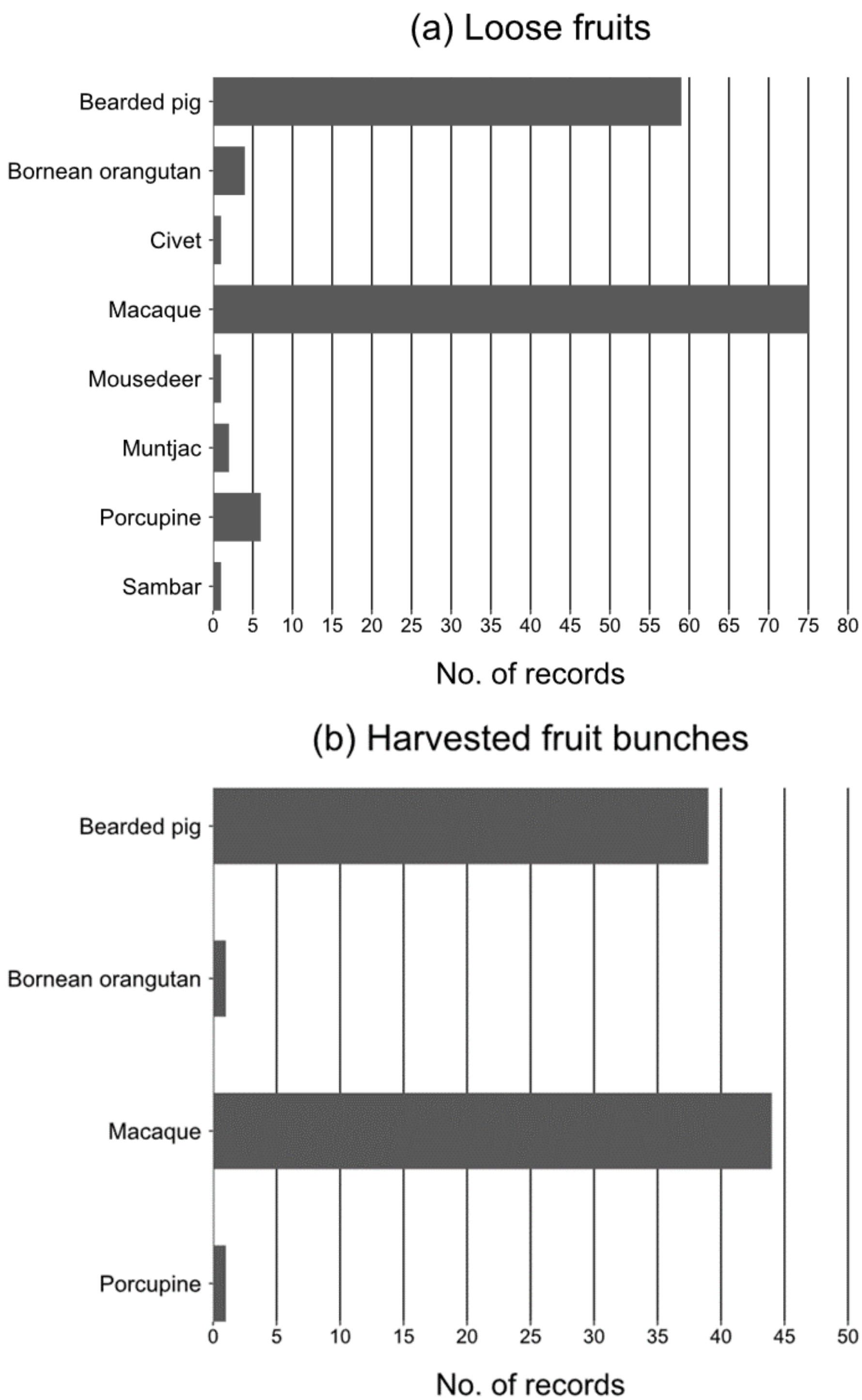
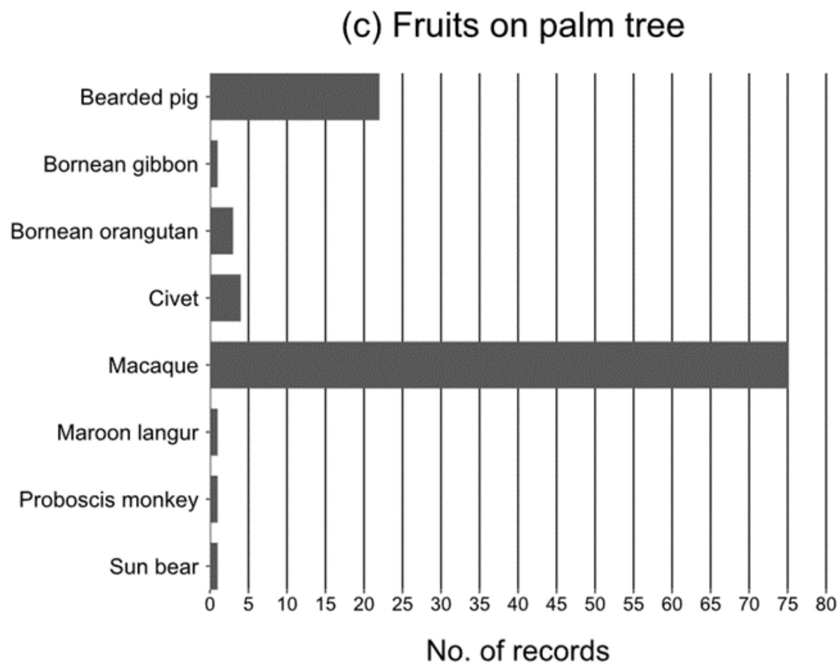


Figure 2: Wildlife species feeding on (a) loose fruits scattered on the ground, (b) harvested fruit bunches, (c) fruits on the palm tree, and (d) oil palm shoots according to respondents from oil palm plantations in the lower Kinabatangan, Sabah, Malaysia.




(c) Fruits on palm tree
Bearded pig
Bornean gibbon
Bornean orangutan
Civet
Macaque
Maroon langur
Proboscis monkey
Sun bear
0 5 10 15 20 25 30 35 40 45 50 55 60 65 70 75 80
No. of records

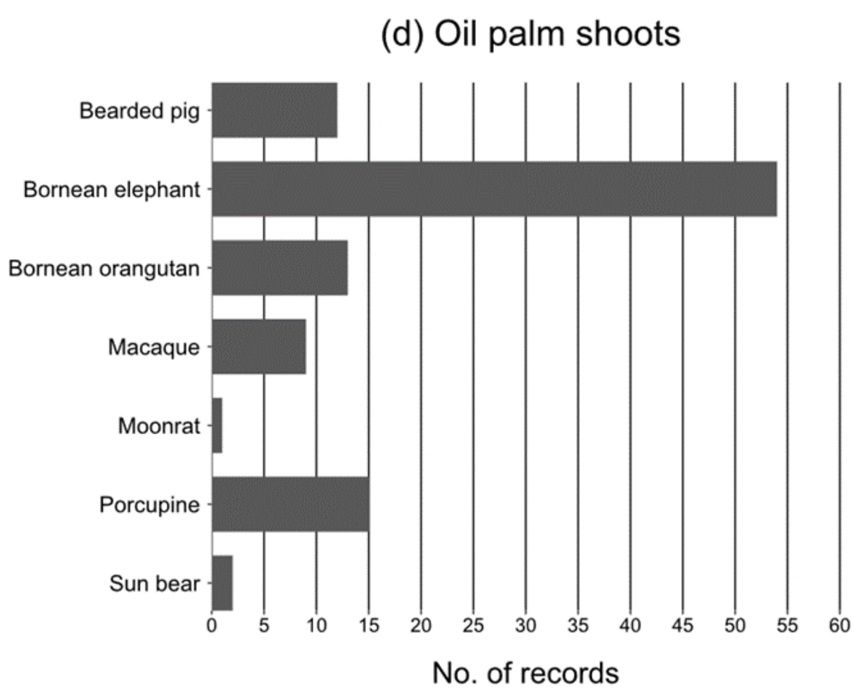

(d) Oil palm shoots
Bearded pig
Bornean elephant
Bornean orangutan
Macaque
Moonrat
Porcupine
Sun bear
0 5 10 15 20 25 30 35 40 45 50 55 60
No. of records

Figure 3: Wildlife species labeled as destructive to the oil palm crop according to respondents from oil palm plantations in the lower Kinabatangan, Sabah, Malaysia.

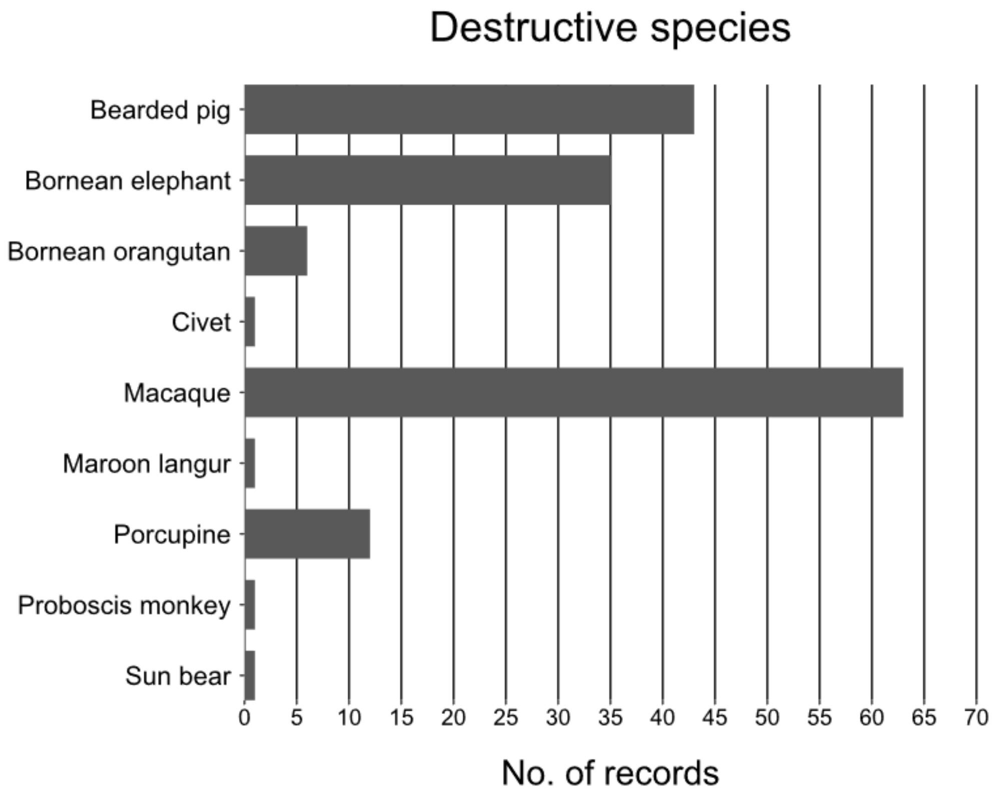


Figure 4: Mean destructiveness and danger level ranks for certain species according to respondents from oil palm plantations in the lower Kinabatangan, Sabah, Malaysia.

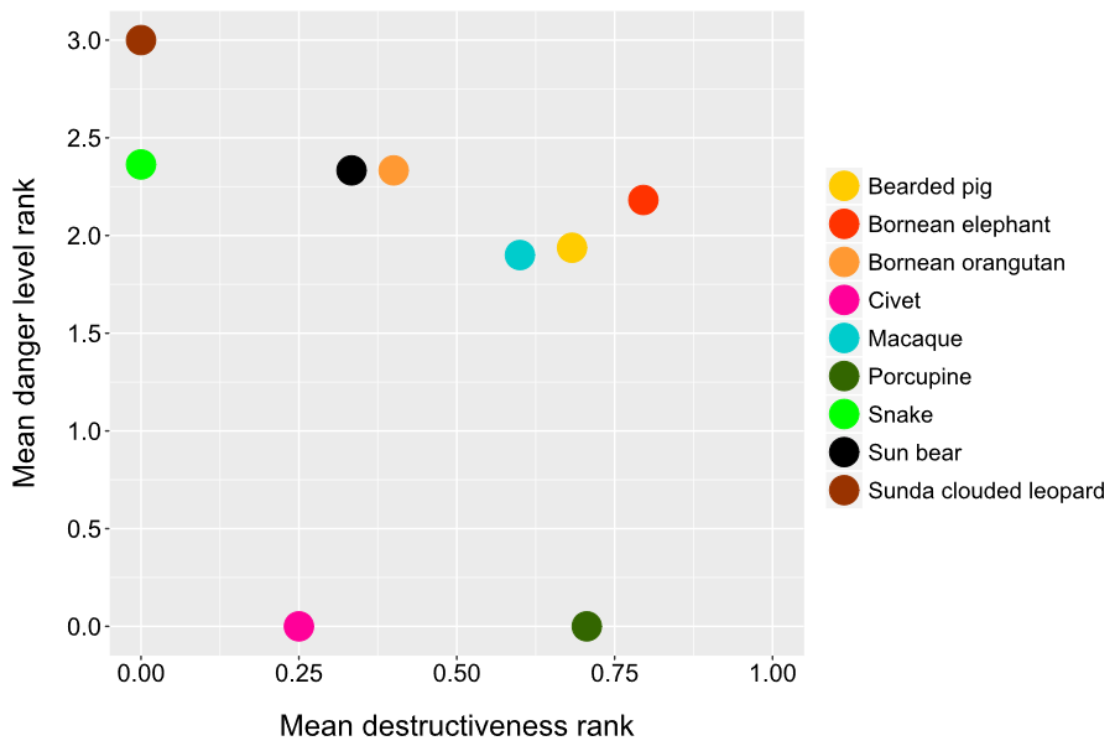


Figure 5: A typical forest-oil palm border in the lower Kinabatangan, Sabah, Malaysia. Sun bears can easily pass under electric fences designed to keep elephants away and feed on the large amounts of loose fruits and fruit bunches left on the ground.

Supplementary information

Table S1: Mean destructiveness of mammal species ranked by multiple respondents from oil palm plantations in the lower Kinabatangan, Sabah, Malaysia.

Species	**Mean destructiveness rank (± SD)[1]**	**Total records[2]**
Muntjac *Muntiacus spp.*	0.00 (0.00)	2
Civet *Viverra tangalungga, Paradoxurus hermaphroditus*	0.25 (0.50)	4
Sun bear *Helarctos malayanus*	0.33 (0.58)	5
Bornean orangutan *Pongo pygmaeus*	0.40 (0.51)	16
Macaque *Macaca spp.*	0.60 (0.49)	117
Bearded pig *Sus barbatus*	0.68 (0.47)	73
Porcupine *Hystrix spp.*	0.71 (0.47)	20
Bornean elephant *Elephas maximus borneensis*	0.80 (0.41)	55

[1]Ranks: not destructive, 0; destructive.

[2]Not all records were used in the calculation of the mean rank as not all respondents that encountered a species ranked it.

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APPENDIX I

Questionnaire used to interview oil palm plantation workers and farmers in the lower Kinabatangan floodplain region, Sabah, Malaysian Borneo (on following page).

Oil Palm Plantation Farmer/Worker Interview Sheet					
Plantation information					
Owner					
Size					
Location					
Palm ages					
Condition(Sparse/Dense understory)					
Respondent information					
Age					
Ethnicity					
Religion					
Occupation					
Time working in plantation	< 1 year 1-5 years 6-10 years > 10 years				
Time in Sabah	< 1 year 1-5 years 6-10 years > 10 years				
Species	0=rare, 1=common, 2=very common	0=Morning, 1= Noon, 2=Evening, 3=Night	0=within planted area, 1=plantation-forest border	Encountered during specific weather season? 0=Unsure, 1=No, 2=Rain, 3=Dry	Encountered during forest fruiting season? 0=Unsure, 1=No, 2=Yes (which month?)

Species (Feeding)	0=Palm fruits, 1=Palm shoots, 2=Other crops (record in notes)	If fruits, 0=loose fruits, 1= fruit bunches, 2=fruits on palm	How do you know? 0=Seen, 1=Sign, 2=Heard from others	0=Not destructive, 1=Destructive	Notes

Species (Feared)	Attacked/Threatened people? 0=No, 1=Yes	No of times? 0=3 or less, 1=4 or more	How dangerous? 0=Least, 1=Dangerous, 2=Extremely	Reaction? 0=None/Retreat, 1=Chase away, 2=Catch/trap, 3=Kill	Notes
Bear focused questions					
Encountered in this plantation? 0=No, 1=Yes	If yes, 0=Seen yourself, 1=Heard from someone	How often? 0=rare, 1=common, 2=very common	What is bear doing?	Encountered during specific weather season? 0=Unsure, 1=No, 2=Rain, 3=Dry	Encountered during forest fruiting season? 0=Unsure, 1=No, 2=Yes (which month?)

Distance from forest? 0=In forest/forest edge, 1=<2km, 2=3km and more	Feeds on OP fruit? 0=No, 1=Yes	0=Loose fruits, 1=Fruit bunches, 2=Fruits on palm	How do you know? 0=Seen, 1=Found sign, 2=Heard from others	Destructive? 0=No, 1=Yes	Reaction? 0=None, 1=Retreat, 2=Chase away
Attacked/Threatened people? 0=No, 1=Yes	Are bears hunted/killed in OP plantations? 0=No, 1=Yes	How do you know? 0=Seen, 1=Found evidence, 2=Heard from others	Is bear hunting targeted (1) or just opportunistic (2)?	Hunted for specific body parts (1, record parts in Notes), to protect crops (2), or in self defense (3)?	What happens to the body parts? 0=Subsistence, 1=Sold (Record information in Notes)
Notes					
What is your opinion on bears?					
Should people be given the chance to hunt/keep wildlife? 0=No, 1=Yes		Which species?			
Should wildlife be protected? 0=No, 1=Yes		Which species?			