RAFFLES BULLETIN OF ZOOLOGY **72**: 135–149
Date of publication: 12 June 2024
DOI: 10.26107/RBZ-2024-0012
http://zoobank.org/urn:lsid:zoobank.org:pub:2ED61905-11E3-4B9F-8D59-BA6E7EBAE710

Predator-prey co-occurrence: defining a management plan for a newly established protected area in central Myanmar

Margaret Nyein Nyein **Myint**[1,2*], Dusit **Ngoprasert**[1], Niti **Sukumal**[1], Nay Myo **Shwe**[1,2], Paing **Soe**[2], Zin Mar **Hein**[2] & Tommaso **Savini**[1]

Abstract. Protected areas (PAs) have an important role in biodiversity conservation, protecting wildlife habitats and mitigating wildlife population decline. Over the past decade, Myanmar, which is part of the Indo-Burma Biodiversity Hotspot, has expanded its number of PAs, preserving substantial portions of the remaining natural habitat. Unfortunately, comprehensive data on the biodiversity status of most newly established PAs is lacking. In this study, we estimated the occurrence of predators and photographic capture rates of prey in North Zarmari Wildlife Sanctuary, a recently established PA situated in the Bago Yoma Range of central Myanmar, to provide data for its management plan development. A camera trap survey conducted between March and June 2020 at 63 grid cells (4 × 4 km^2) recorded 25 mammal species, including Critically Endangered and Endangered prey and predators, and illegal activities such as illicit hunting and logging. Leopard (*Panthera pardus*) occurrence was positively influenced (AUC of 0.92) by the presence of primates and wild pigs. Bears showed a preference for areas surrounding human settlements, perhaps due to the availability of food related to human activities. Small carnivores were also affected by human disturbances. Despite evidence of low protection and high human disturbance, our results suggest a notably rich diversity of both prey and predator species within the area. The occurrence of species was impacted by a combination of factors, including habitat type and the level of human disturbance. Additionally, many species appeared to be affected by their proximity to the forest edge and human settlements. Therefore, the success of future conservation efforts hinges on the implementation of well-informed management action, underscored by a thorough understanding of the distribution and interplay among key wildlife species. This entails addressing relevant threats, as well as understanding the species' interactions with environmental factors at play within their habitats.

Key words. Bago Yoma range, camera trap, North Zamari Wildlife Sanctuary, species distribution, wildlife threats

INTRODUCTION

Myanmar, situated within the Indo-Burma Biodiversity Hotspot, boasts remarkable biodiversity, ranking among the top 10 irreplaceable global hotspots but, unfortunately, also among the top five for threats to its biodiversity and endemism (Naing Tun et al., 2021). The country shows a poor conservation record (Avillés-Polanco et al., 2019), resulting in extensive habitat loss, deforestation, illegal hunting, and logging (Mon et al., 2010; Saung et al., 2021). Moreover, illegal wildlife trade is particularly rife in Myanmar, aggravated not only by a high rate of local consumption at the national level (Mon et al., 2010; Rustad et al., 2012; Saung et al., 2021), but also by a high international demand, primarily from neighboring countries such as China (Mak & Song, 2018).

Located in the south-central part of Myanmar, the Bago Yoma range, also known as the Pegu Range, was historically heavily forested with natural teak (*Tectona grandis*) (Kyaw, 2018) and inhabited by several threatened species such as the Asian elephant (*Elephas maximus*) (Evans et al., 2020), Western hoolock gibbon (*Hoolock hoolock*), gaur (*Bos gaurus*) (Duckworth et al., 2016), and Popa langur (*Trachypithecus popa*) (Roos et al., 2020). For over 50 years, the area has served as the country's major teak hardwood resource (Evans et al., 2020). Moreover, mirroring the rest of the country, the range has undergone severe forest degradation over the past decades coupled with extensive threats to its wildlife, including illegal human encroachment (Saung et al., 2021), bamboo extraction, hunting, wildlife trade, and logging (Win et al., 2023), facilitated by its many access points.

To reverse biodiversity degradation, Myanmar has established a number of protected areas (Lwin et al., 2022), but these have not consistently received adequate financial and resource support (Win et al., 2023). The lack of enforcement and effective management has resulted in continuous habitat loss, degradation, and illegal hunting (Bonham et al., 2008), reducing those areas to ineffective 'paper parks' (Aung, 2007). Among the newly established protected areas, the North Zamari Wildlife Sanctuary, established in 2014 (Hein

---

Accepted by: Norman Lim T-Lon

[1] Conservation Ecology Program, King Mongkut's University of Technology Thonburi, Bangkok, Thailand

[2] World Wide Fund for Nature - Myanmar, Bahan Township, Yangon, Myanmar, 11201; Email: margaret.nmyint@gmail.com (*corresponding author)

ISSN 2345-7600 (electronic) | ISSN 0217-2445 (print)

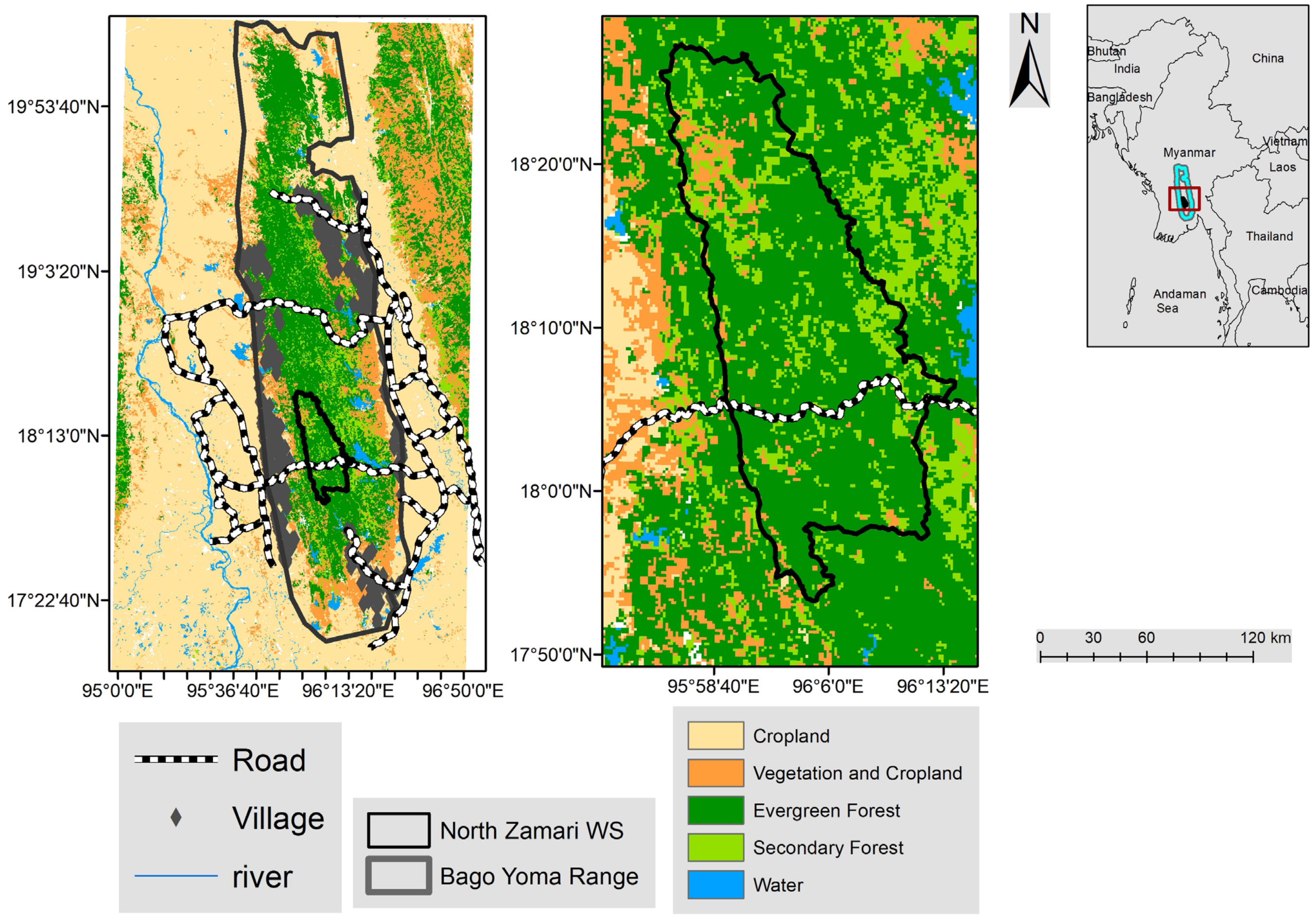

Fig. 1. Maps indicating the Bago Yoma Range and North Zarmari Wildlife Sanctuary in Bago Region, Myanmar.

et al., 2020), has been identified as crucial for conserving the endangered green peafowl (*Pavo muticus*) and as one of the last six remaining strongholds in Southeast Asia (Sukumal et al., 2020; Win et al., 2023). Despite almost a decade of legal protection, the area still lacks a comprehensive management plan and has limited manpower to fully implement effective protection measures, such as intensive patrolling and bans on human encroachment, shifting cultivation, and non-timber extraction (Win et al., 2023). The area also experiences extensive hunting primarily for local consumption, medicinal uses or trade within the villages surrounding the North Zamari Key Biodiversity Area (Evans et al., 2020).

To enhance its management design and planning, it is imperative to gain a thorough understanding of the distribution and interactions of key wildlife species with the environmental factors in the area. This study, therefore, aimed to understand the spatial distribution of key wildlife species, including large and small carnivores such as the leopard (*Panthera pardus*), Asiatic black bear (*Ursus thibetanus*), sun bear (*Helarctos malayanus*), leopard cat (*Prionailurus bengalensis*), Asiatic golden cat (*Catopuma temminckii*), common palm civet (*Paradoxurus hermaphroditus*), large Indian civet (*Viverra zibetha*), and crab-eating mongoose (*Herpestes urva*) using a camera trap survey. Additionally, we determined the species' relationships with landscape variables, including threats collected using the Spatial Monitoring and Reporting Tool (SMART). These species play a crucial role in this landscape and can serve as good indicators for monitoring the overall health of the ecosystem in the area. Our findings provide invaluable data for the formulation of a comprehensive management plan for North Zarmari Wildlife Sanctuary.

MATERIAL AND METHODS

Study site. North Zarmari Wildlife Sanctuary (hereafter referred to as 'Zamari') is located in the southern part of the Bago Yoma range (central Myanmar) (18°04′N, 96°02′E) and consists of 981 km^2 of evergreen forest (79.18%), secondary forests (15.39%), vegetation mixed with cropland (5.38%) and cropland (0.05%) at an elevation ranging from 44 to 815 m (Fig. 1). Zamari was established specifically to protect the endangered Asian elephant, which serves as a flagship species for the protection of other flora and fauna (Ministry of the Natural Resources and Environmental Conservation of Myanmar Forest Department, 2015). The area was legally classified as a reserve forest until 2013, and was subsequently changed to a wildlife sanctuary in 2014. Zamari is severely degraded as a consequence of heavy logging (primarily teak) in the past, combined with recurring seasonal fires, some of which are linked to humans living in the area (Hein et al., 2020). Extensive logging has caused heavy deforestation and increased soil erosion in the area. Prior to its designation as a protected area, a two-lane road was built, bisecting the

Table 1. Descriptions of covariates used in the generalised linear models.

Covariates	Name	Description and measurement details
1. Landscape		
NDVI value 500m	NDVIval500m	https://developers.google.com/earth-engine/datasets/catalog/COPERNICUS_S2_SR NDVI value within 500 m radius around the camera trap location
NDVI value 1km	NDVIval1km	https://developers.google.com/earth-engine/datasets/catalog/COPERNICUS_S2_SR NDVI value within 1 km radius around the camera trap location
NDVI value 2km	NDVIval2km	https://developers.google.com/earth-engine/datasets/catalog/COPERNICUS_S2_SR NDVI value within 2 km radius around the camera trap location
Distance to year-round stream	DistBigStr	nearest distance (km) from camera trap location to year-round stream
Forest cover	ForestCover	https://doi.org/10.5281/zenodo.3939050 forest cover in percentage using land cover resolution 100m, tree cover fraction 2019
2. Human disturbance		
Poacher	Poacher	Number of independent pictures of poacher from camera trap per 100 trap days
Dog	Dog	Number of independent pictures of dog from camera trap per 100 trap days. of independent pictures per 100 trap days)
Domestic cattle	Domscattle	Number of independent pictures of buffalo and cow from camera trap per 100 trap days
Distance to village	DistVillage	nearest distance (km) from camera trap location to village

area from east to west (Hein et al., 2020). The regular use of this road by people traversing through Zamari is having, or may have, negative impacts on the wildlife. This road is widely recognised to constitute a major threat to the area (Evans et al., 2020).

Camera trap survey. We conducted a camera trap survey to gain insights into the presence and absence of prey and predators. Across 63 grid cells (4 × 4 km^2) covering the entirety of Zamari, we deployed a Browning Strike Force HD Pro model camera-trap in each cell, positioned at or within 500 m from the centre point, knee height (~40 cm) (Kolowski & Forrester, 2017) and turned on the picture mode only. The cameras were strategically placed along animal trails (50 sites) and near animal signs (five sites), waterholes (three sites), and animal tracks (five sites). During the four-month survey (March–June 2020), the camera traps were operating 24 hours daily, representing a 'trap-day' in our assessment of survey effort. We identified each animal in the photographs to the species level, recorded the time and date of capture, and classified each photograph as an independent detection when the same species was detected at the same location with a time gap of >30 min (O'Brien et al., 2003; Sanderson & Harris, 2013). We calculated photographic rates as an index of species activity at a site (Sollmann, 2018), by standardising the photographic rates for each species (number of independent photographs per 100 trap days) (O'Brien et al., 2003).

Landscape, human disturbance and prey variables. We collected data on the important landscape attributes and human disturbances, which were used as predictor variables to investigate the distribution of key wildlife species and the threats they face. The landscape variables were selected according to their relevance to wildlife distribution (Simcharoen et al., 2014; Williams et al., 2018). We considered the following landscape variables, namely: (1) the Normalized Difference Vegetation Index (NDVI), (2) forest cover, and (3) distance to year-round stream. NDVI was evaluated at three scales (0.5, 1.0 and 2.0 km; Table 1). NDVI serves as an indicator of vegetation health, helping to identify areas where damage has occurred and predicting changes in mammalian species and abundance (Oindo, 2002). While some studies have compared NDVI with plant biomass in open fields, few studies have observed the correlation between NDVI and changes in ground vegetation biomass in forested ecosystems (Borowik et al., 2013). We determined the forest cover percentage for the whole study area using a land cover resolution of 100 m and tree cover fraction data from 2019 (Masiliūnas et al., 2021). Distance to year-round stream was measured as the distance from camera trap locations to the nearest year-round stream (km). We considered this a critical landscape variable in our analysis because it can affect the behavioral ecology of our key species in the dry forest ecosystem (Khoewsree et al., 2022).

For human disturbance variables, we used an index of disturbance activity based on the presence of poachers, domestic dogs, and domestic cattle using camera-trap pictures that indicated human activities and caused disturbance to the forest. Locally, poachers use trained hunting dogs when they go hunting (Evans et al., 2020). The index value should represent the threat level posed by humans. We also standardised the number of independent pictures per 100 trap days for three of the human disturbance variables. Additionally, we used the distance between each camera trap location and the nearest village (km) to represent the accessibility from the village. Villages represent areas of human activity, which can include habitat alteration and may act as barriers to movement for some species. All landscape variables were prepared by using ArcGIS 10.8.2 and delivered from Google Earth Engine. In addition, threats such as shifting cultivation practices collected using SMART and hut locations were collected by direct observation during the camera trap survey. We reported additional threats by using the number of locations that were observed.

We included a variety of prey species as covariates in our analysis based on their importance in the diet of our key species. The groups comprised large prey (banteng (*Bos javanicus*), gaur (*Bos gaurus*), sambar (*Rusa unicolor*)), small prey (wild pig (*Sus scrofa*), mainland serow (*Capricornis sumatraensis*), muntjac (*Muntiacus muntjac*), porcupine (*Hystrix brachyura*)), primates (Popa langur (*Trachypithecus popa*), rhesus macaque (*Macaca mulatta*), northern pig-tailed macaque (*Macaca leonina*)), and the combination of all prey. We also standardised the number of independent prey pictures per 100 trap days. However, we excluded large prey and serow from the leopard models due to their low detection rates, but we incorporated only all prey, muntjac, wild pig, and primates instead (Hayward et al., 2006). We excluded prey species in our models for the sun bear, Asiatic black bear, leopard cat, common palm civet, large Indian civet, and crab-eating mongoose as their diets do not rely on those prey species. We used the leopard as a predator in models for the leopard cat and large Indian civet. We used porcupine and muntjac as prey species for the Asiatic golden cat (Grassman et al., 2005; Kawanishi & Sunquist, 2008) while excluding small prey such as rodents. Small rodents were not included because 1) small body rodents tend to have biased detection from our camera trap setup, and 2) the energetic efficient of smaller rodents does not meet that compared to larger rodents (Kamler et al., 2020). We also did not consider small carnivores like civets in the diet because their involvement in intraguild predation rather than being main prey. Asiatic golden cats' diet has been represented as mammals ranging in size from a rat to a muntjac (Grassman et al., 2005). They perhaps rely principally on small prey like large rodents and possibly even largest prey like muntjac (Kamler et al., 2020). Although we detected the marbled cat (*Pardofelis marmorata*), binturong (*Arctictis binturong*), dhole (*Cuon alpinus*), and yellow-throated marten (*Martes flavigula*), we excluded them from our data analysis since their low detection rates could not support a comprehensive analysis.

Data Analysis. The detection data (detected/non-detected for leopard, Asiatic golden cat, leopard cat, and crab-eating mongoose, and number of independent photographs for Asiatic black bear, sun bear, common palm civet, and large Indian civet) were used as a response variable in our analysis. We employed various statistical distributions and modelling techniques to effectively account for the specific characteristics of the species' detection patterns. A logistic regression model with a logit link function was used for the leopard (detected in seven locations), Asiatic golden cat (detected in 10 locations), leopard cat (detected in seven locations), and crab-eating mongoose (detected in seven locations) as they had few detections. For the sun bear, Asiatic black bear, and common palm civet, we applied a generalised linear model (GLM) with the Poisson distribution. We opted to use a negative binomial distribution for the large Indian civet to address the observed overdispersion of data (ratio of variance/mean > 9).

Before analysing the data, we checked for outliers by using scatter plots of the response and predictor variables. We also explored the correlation among all predictor variables by using the Kendall method with the "corrplot" function (Wei & Simko, 2021). To prevent multicollinearity, any two predictor variables showing a correlation of > 0.50 were not included in the same regression model. To address the variation in camera trap survey effort across different locations, we incorporated an offset into the model formula. This approach is equivalent to including survey effort (measured in trap-days) as a regression predictor, but with its coefficient fixed at 1 (Gelman & Hill, 2006). Variance Inflation Factor (VIF) was also used to evaluate multicollinearity within each model; a VIF value of 5 was used as a cut-off for considering multicollinearity between the models (Oksanen, 2010; Zuur et al., 2010). We used the 'performance' package for VIF estimation (Lüdecke, 2021). All covariates were standardised by centring and scaling to one standard deviation (Gelman & Hill, 2006).

We assessed the normality assumption of our model by examining its residuals plot using the 'DHARMa' package (Hartig, 2020). We used the area under the receiver operating characteristic curve (AUC), which ranges from 0.5 (models that are no better than random) to 1.0 (high-accuracy models) to evaluate the classification accuracy of the logistic regression model (Hosmer & Lemeshow, 2000; Franklin et al., 2009). We calculated AUC thresholds using multiple cut-off points (0.2, 0.4, 0.5, 0.6, 0.8) where sensitivity equalled specificity, using the package 'PresenceAbsence' (Freeman & Moisen, 2008). We considered between 0.7 to 0.8 as "acceptable", 0.8 to 0.9 as "excellent" and greater than 0.9 as "outstanding" discrimination. We compared all the models using the Akaike information criteria with a correction for small sample size (AICc) to select the best model with the lowest AICc and the highest support in terms of model weights ($AICc_{wi}$) (Akaike, 1973; Burnham & Anderson, 2002). To deal with model selection uncertainty, we used model averaging of regression coefficient using the 'AICcmodavg' package (Mazerolle, 2021) for uncertainty candidate models. The candidate set included models with ΔAICc values <6 (Burnham & Anderson, 2002; Richards, 2008). Then, we examined the effect size (beta coefficient) of the covariates to determine their importance and the confident interval not overlapped zero. We used an 85% confidence interval to identify covariates that exert a significant impact on the distribution of key species (Arnold, 2010). When

Table 2. Photographic rates of key mammal species per 100 trap days and number of locations occupied from a total of 57 camera trap locations, obtained from March to June 2020 (total effort = 4,267 trap days and total independent pictures = 1,193) in North Zarmari Wildlife Sanctuary, Bago Region, Myanmar.

Species	**Scientific name**	**No. of independent pictures**	**Photographic Rate/100 trap days**	**No. of locations occupied**	**Raw occupancy (%)**
Red_muntjac	*Muntiacus muntjak*	453	10.62	47	82.46
Wild_pig	*Sus scrofa*	102	2.39	32	56.14
Large_Indian_civet	*Viverra zibetha*	129	3.02	26	45.61
Rhesus_macaque	*Macaca mulatta*	82	1.92	22	38.60
Malayan_porcupine	*Hystrix brachyura*	124	2.91	17	29.82
Common_palm_civet	*Paradoxuurs hermaphroditus*	30	0.70	13	22.81
Banteng	*Bos javanicus*	25	0.59	11	19.30
Sun_bear	*Helarctos malayanus*	21	0.49	11	19.30
Nothern_pig-tailed_macaque	*Macaca leonina*	62	1.45	10	17.54
Asian_golden_cat	*Caopum emminckii*	14	0.33	10	17.54
Asiatic_black_bear	*Ursus thibetanus*	21	0.49	9	15.79
Leopard_cat	*Prionailurus bengalensis*	21	0.49	7	12.28
Crab_eating_mongoose	*Herpestes urva*	29	0.68	7	12.28
Mainland_serow	*Capricornis sumatraensis*	16	0.37	7	12.28
Leopard	*Panthera pardus*	13	0.30	7	12.28
Yellow-throated_marten	*Martes flaviqulla*	10	0.23	6	10.53
Popa langur	*Trachypithecus popa*	11	0.26	5	8.77
Sambar	*Rusa unicolor*	5	0.12	4	7.02
Binturong	*Arctictis binturong*	3	0.07	3	5.26
Dhole	*Cuon alpinus*	3	0.07	2	3.51
Large_spotted_civet	*Viverra megaspila*	2	0.05	2	3.51
Gaur	*Bos gaurus*	14	0.33	2	3.51
Marbled_cat	*Pardofilis mamorata*	1	0.02	1	1.75
Black_giant_squirrel	*Ratufa bicolor*	1	0.02	1	1.75
Pangolin_spp	*Manis* spp.	1	0.02	1	1.75

using AIC for model selection, reporting the 85% confidence interval is consistent with the model selection strategy. This is equivalent to conducting a likelihood ratio test to compare models using AIC, which corresponds to a significance level of $p = 0.157$ under the Chi-square distribution with one degree of freedom (Arnold, 2010; Sutherland et al., 2023). We used the 'ggplot2' package (Wickham et al., 2016) to predict the probability of species occurrence at a 25 km distance from the village for the leopard, sun bear, and Asiatic black bear. All data analysis was conducted in R version 4.2.1 (R Core Team, 2022).

RESULTS

Camera trap survey. Our camera trap ran for 4,267 trap days, recording 1,500 independent pictures (Table 2). Six camera traps were missing at the end of the survey. We detected a total of 25 mammal species and 14 bird species in the study area. Of particular concern, we detected 11 globally threatened species, including the Critically Endangered pangolin *Manis* spp. (Table 2). We recorded 13 independent pictures of leopards, 129 independent pictures of the large Indian civet, and 21 independent pictures of the sun bear, Asiatic black bear, and leopard cat. During the camera trap survey period, the photographic rates for different species were as follows: 0.3 trap days for the leopard and Asiatic golden cat, 0.5 for

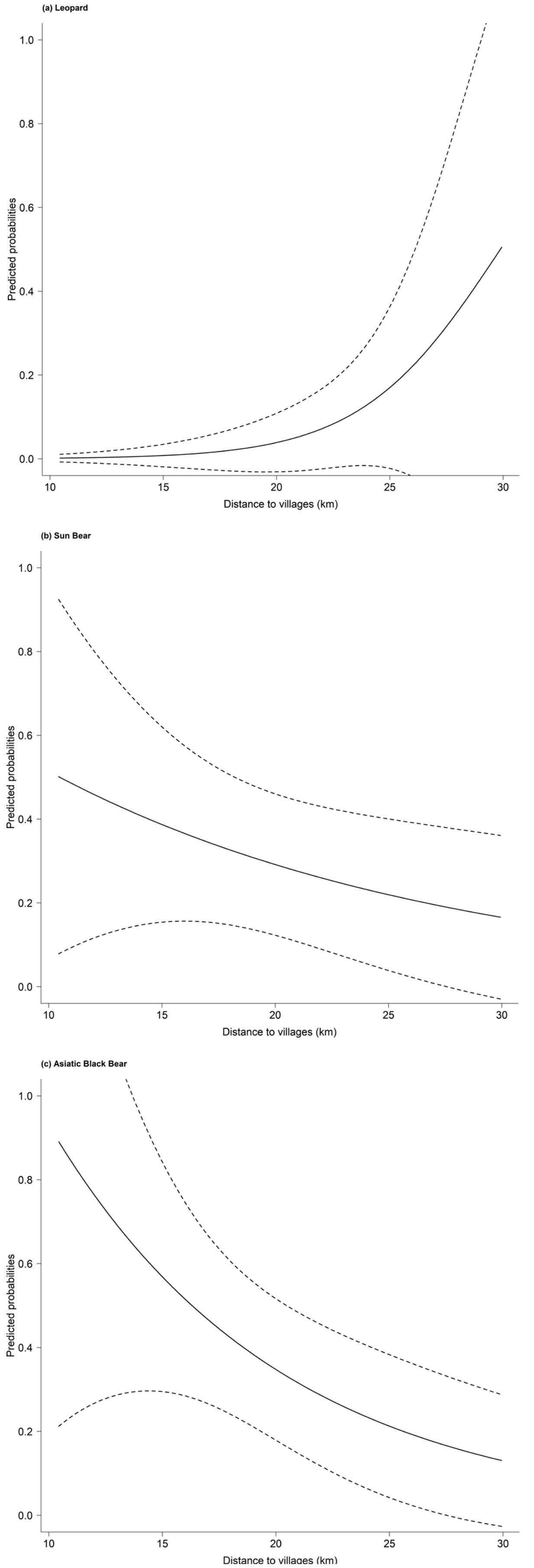

Fig. 2. The predicted probabilities of leopard (a), sun bear (b), and Asiatic black bear (c) occurrences based on the best logistic regression models (85% CIs) utilising distance to the village as the predictor variable.

the sun bear, Asiatic black bear and leopard cat, 0.7 for the common palm civet and crab-eating mongoose, and 3.0 for large Indian civets (Table 2). Moreover, we recorded forest fire pictures at 29 locations and illegal logging activities at 19 locations out of 57 camera trap locations. The presence of huts was observed directly during the camera trap survey at 12 locations and shifting cultivation were recorded at 12 locations as well from the 2020 SMART patrol result.

Data Analysis. The occurrence of leopards was positively influenced by the availability of prey species such as primates (average standardised regression coefficient, $\beta_{avg.}$ = 1.51, 85% CI: 0.02 to 3.01) and wild pigs (β_{avg} = 1.84, 85% CI: 0.42 to 3.27) and, to a lesser extent, by the distance to a village (β_{avg} = 1.77, 85% CI: −0.02 to 3.56, Table 3, Table S1). Our predictions suggest a 17% chance of detecting leopards at a distance of 25 km from a village (Fig. 2a).

The top model proved was effective at discriminating camera-trap stations with leopard presence and absence, with a high AUC value of 0.92, suggesting its reliability in predicting leopards' likelihood of occurrence based on the selected variables.

The occurrence of sun bears and Asiatic black bears was impacted by distinct factors. Sun bears were positively impacted by NDVI at a scale of 500 m, whereas forest cover and distance to year-round streams exerted a negative influence on their presence. Additionally, a marginal effect of the distance to a village was also found (Table S1). Sun bear detection was enhanced in areas with higher NDVI values (average standardised regression coefficient, β_{avg} = 0.51, 85% CI: 0.15 to 0.86), closer proximity to year-round streams (β_{avg} = −0.47, 85% CI: −0.88 to −0.06), and lower forest cover (β_{avg} = −0.36, 85% CI: −0.66 to −0.06). In contrast, the distance to a village emerged as sole and strongest predictor of Asiatic black bears (Table S1), indicating a higher likelihood of occurrence closer to villages (β_{avg} = −0.53, 85% CI: −0.88 to −0.17) (Table 3). The model predicted a 22 % probability of detecting sun bears (Fig. 2b) and a 21 % probability of detecting Asiatic black bears (Fig. 2c) at a 25 km distance from a village.

Furthermore, the Asiatic golden cat's occurrence was strongly positively influenced by the presence of porcupines (average standardised regression coefficient, β_{avg} = 0.93, 85% CI: 0.41 to 1.44). The model suggested that the higher the photographic rate of porcupines, which serves as an indicator of their abundance, the higher the occurrence of Asiatic golden cats. The top model was Porcupine which showed excellent discrimination (Table S1) and showed high accuracy in predicting the presence or absence of the Asiatic golden cat (AUC = 0.80).

For leopard cats, the model suggested that a decrease in the NDVI value (average standardised regression coefficient, β_{avg} = −1.08, 85% CI: −2.06 to −0.10) increases their probability of occurrence. Similarly, the closer the proximity to a village, the higher the likelihood of detecting the species (β_{avg} = −1.17, 85% CI: −2.03 to −0.32). The top model which included

Table 3. Average standardised regression coefficients from model averaging (ΔAICc <6) with standard errors (SE) and 85% confidence interval (CI).

Species	**Estimate**	**SE**	**85% CI**	
			lower	**Upper**
Leopard[a]				
Intercept	−7.32	0.98	−8.73	−5.92
WildPig	1.84	0.99	0.42	3.27
DistVillage	1.77	1.25	−0.02	3.56
Primate	1.51	1.04	0.02	3.01
Poacher	0.14	0.88	−1.12	1.41
DistBigStr	−1.02	1.02	−2.49	0.45
NDVIval500m	−0.9	1.36	−2.86	1.06
Muntjac	−0.54	1.22	−2.30	1.22
Leopard cat[a]				
Intercept	−6.75	0.67	−7.72	−5.78
Leopard	0.55	1.33	−1.37	2.46
DistBigStr	0.50	0.47	−0.18	1.18
Poacher	0.47	0.37	−0.07	1.00
DistVillage	−1.17	0.60	−2.03	−0.32
NDVIval500m	−1.08	0.68	−2.06	−0.10
ForestCover	−0.38	0.44	−1.01	0.25
Asiatic golden cat[a]				
Intercept	−6.06	0.44	−6.69	−5.42
Porcupine	0.93	0.36	0.41	1.44
DistVillage	0.35	0.39	−0.21	0.91
DomsCattle	0.35	0.38	−0.2	0.89
ForestCover	0.21	0.46	−0.45	0.87
Poacher	−0.69	0.83	−1.88	0.51
Muntjac	−0.12	0.42	−0.72	0.48
DistBigStr	−0.01	0.45	−0.66	0.64
Sun bear[b]				
Intercept	−5.47	0.29	−5.89	−5.04
NDVIval500m	0.51	0.25	0.15	0.86
Dog	0.15	0.25	−0.22	0.51
DistBigStr	−0.47	0.29	−0.88	−0.06
ForestCover	−0.36	0.21	−0.66	−0.06
Poacher	−0.32	0.41	−0.91	0.27
DistVillage	−0.31	0.23	−0.64	0.02
Domscattle	−0.15	0.37	−0.67	0.38
Asiatic black bear[b]				
Intercept	−5.50	0.28	−5.90	−5.09
ForestCover	0.21	0.24	−0.13	0.55
NDVIval2km	0.17	0.22	−0.14	0.48
Domscattle	0.13	0.40	−0.45	0.71
DistVillage	−0.53	0.24	−0.88	−0.17
Poacher	−0.49	0.45	−1.15	0.16
Dog	−0.07	0.29	−0.48	0.34
DistBigStr	−0.04	0.28	−0.45	0.36
Large Indian civet[c]				
Intercept	−3.60	0.28	−4.00	−3.19
NDVIval2km	0.69	0.38	0.14	1.24
Poacher	−0.54	0.37	−1.08	0.00
Dog	−0.30	0.31	−0.75	0.15
DistBigStr	−0.08	0.32	−0.53	0.36
Leopard	−0.05	0.79	−1.19	1.09

Species	Estimate	SE	85% CI	
			lower	Upper
ForestCover	−0.05	0.28	−0.46	0.36
DistVillage	−0.02	0.31	−0.47	0.43
Common palm civet[b]				
Intercept	−5.36	0.25	−5.72	−5
NDVIval2km	0.5	0.18	0.24	0.75
DomsCattle	0.49	0.32	0.03	0.95
Dog	0.32	0.19	0.04	0.6
ForestCover	0.25	0.23	−0.08	0.59
DistBigStr	0.06	0.25	−0.3	0.42
DistVillage	−0.48	0.22	−0.8	−0.16
Poacher	−0.38	0.43	−0.99	0.24
Crab-eating mongoose[a]				
Intercept	−6.56	0.58	−7.4	−5.72
NDVIval2km	0.36	0.45	−0.28	1
Dog	0.04	0.47	−0.64	0.71
DistVillage	−1.03	0.55	−1.82	−0.25
Poacher	−0.82	0.98	2.23	0.58
DistBigStr	−0.03	0.52	−0.78	0.72

[a] Logistic regression; [b] Poisson regression and [c] Negative binomial regression

NDVIval500m and distance to village provided excellent discrimination, with high accuracy to predict the leopard cat's presence or absence (AUC = 0.82).

Large Indian civets showed a high probability of occurrence with increased NDVI values at a 2 km distance away from the camera trap locations (average standardised regression coefficient, $\beta_{avg} = 0.69$, 85% CI: 0.14 to 1.24). In addition, the analysis indicated a negative relationship between species occurrence and poaching ($\beta_{avg} = -0.54$, 85% CI: −1.08 to 0.00, Table 3).

Common palm civets were positively impacted by higher NDVI values at a 2 km distance away from the camera trap locations (average standardised regression coefficient, $\beta_{avg} = 0.5$, 85% CI: 0.24 to 0.75) and by the presence of dogs ($\beta_{avg} = 0.32$, 85% CI: 0.04 to 0.6). However, the likelihood of detecting the species diminished in proximity to a village ($\beta_{avg} = -0.48$, 85% CI: −0.8 to −0.16).

For the crab-eating mongoose, the top model was Distance to Village, which suggested a higher probability of the species' occurrence closer to a village (average standardised regression coefficient, $\beta_{avg} = -1.03$, 85% CI: −1.82 to 0.25). The top model demonstrated excellent discriminatory ability in predicting the presence or absence of the species (AUC = 0.81).

DISCUSSION

Species distribution and abundance. Our findings highlight the importance of North Zarmari Wildlife Sanctuary for preserving biodiversity, especially large mammals, despite facing significant challenges such as high human disturbance and low protection levels. In particular, the results clearly show how this newly established protected area can emerge as a hotspot for biodiversity conservation across the Bago Yoma Range.

The area boasts a large diversity of predators (large and small carnivores) and large prey species (Table 3). The study also highlights the potential importance of maintaining suitable habitats for prey species, which ultimately contributes to the conservation of endangered large carnivores (i.e., leopards and dholes) in the Bago Yoma region. Unsurprisingly, our results demonstrate the connection between the occurrence of species in the area and a combination of habitat type and the level of human disturbance. In addition, many of the species appeared to be directly affected by their proximity to the forest edge and nearby settlements. These results can, therefore, allow park management to target both specific areas of interest and related specific threats.

The activity index (abundance) of key large prey species such as the banteng, gaur, and sambar (Table 3) is significant as they play a critical role in maintaining the balance of ecosystems within their respective habitats and are sought after by predators such as big cats (Pakpien et al., 2017). Smaller prey species like wild pigs, mainland serow, muntjac, and porcupines also serve as vital sources of sustenance for a variety of predators, including large carnivores (Rasphone et al., 2022). Lastly, the primates serve as prey for leopards (Hayward et al., 2006) and are important contributors to the ecosystem owing to their feeding habits, as well as their roles in seed dispersal and in maintaining forest health.

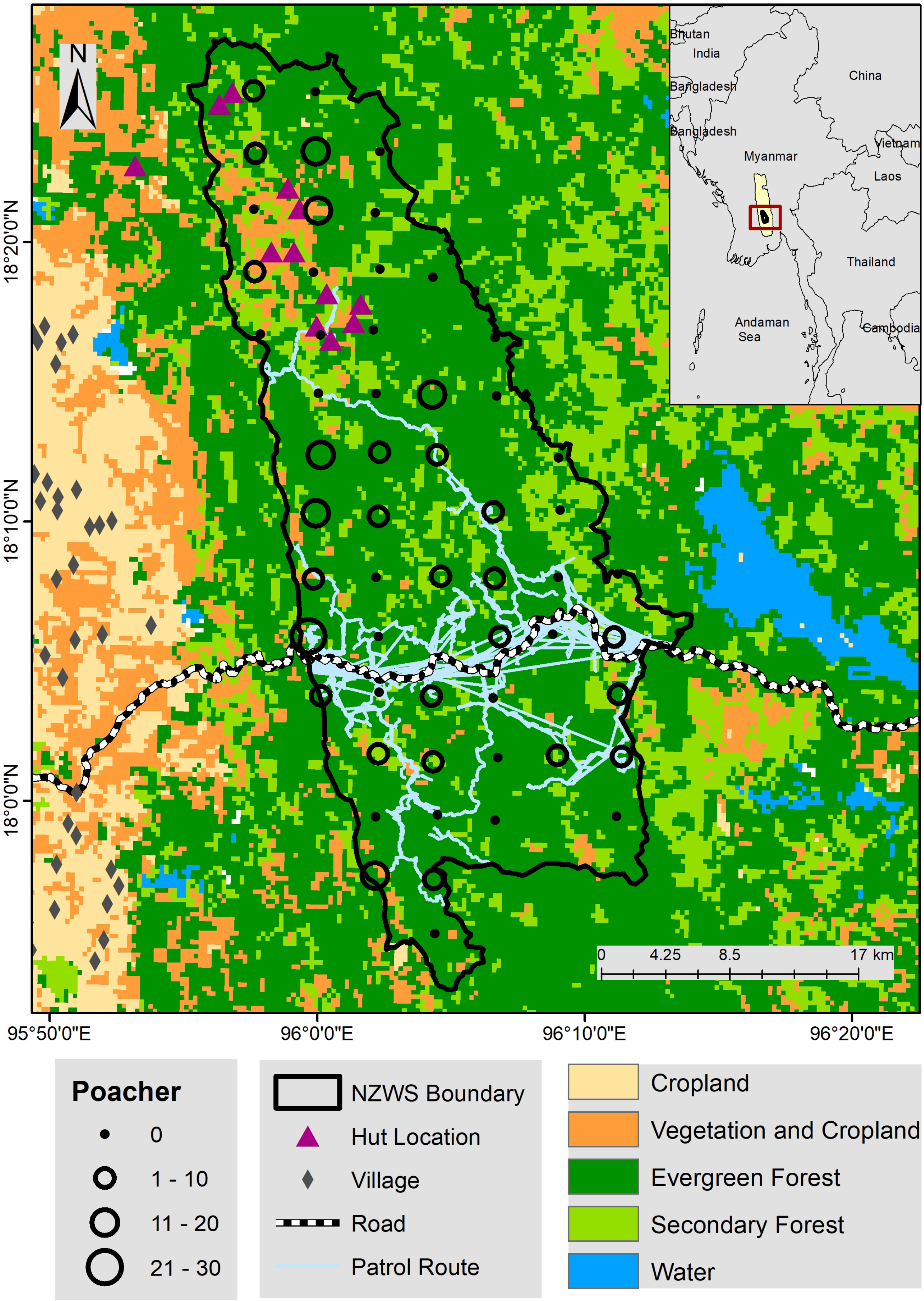

Fig. 3. Map indicating camera trap survey locations from March to June 2020 in North Zarmari Wildlife Sanctuary, Bago Region, Myanmar. Poachers' detection per 100 trap days, patrol routes in the year 2020, and hut locations. The figure suggests a high hunting pressure with low patrol efforts in the area.

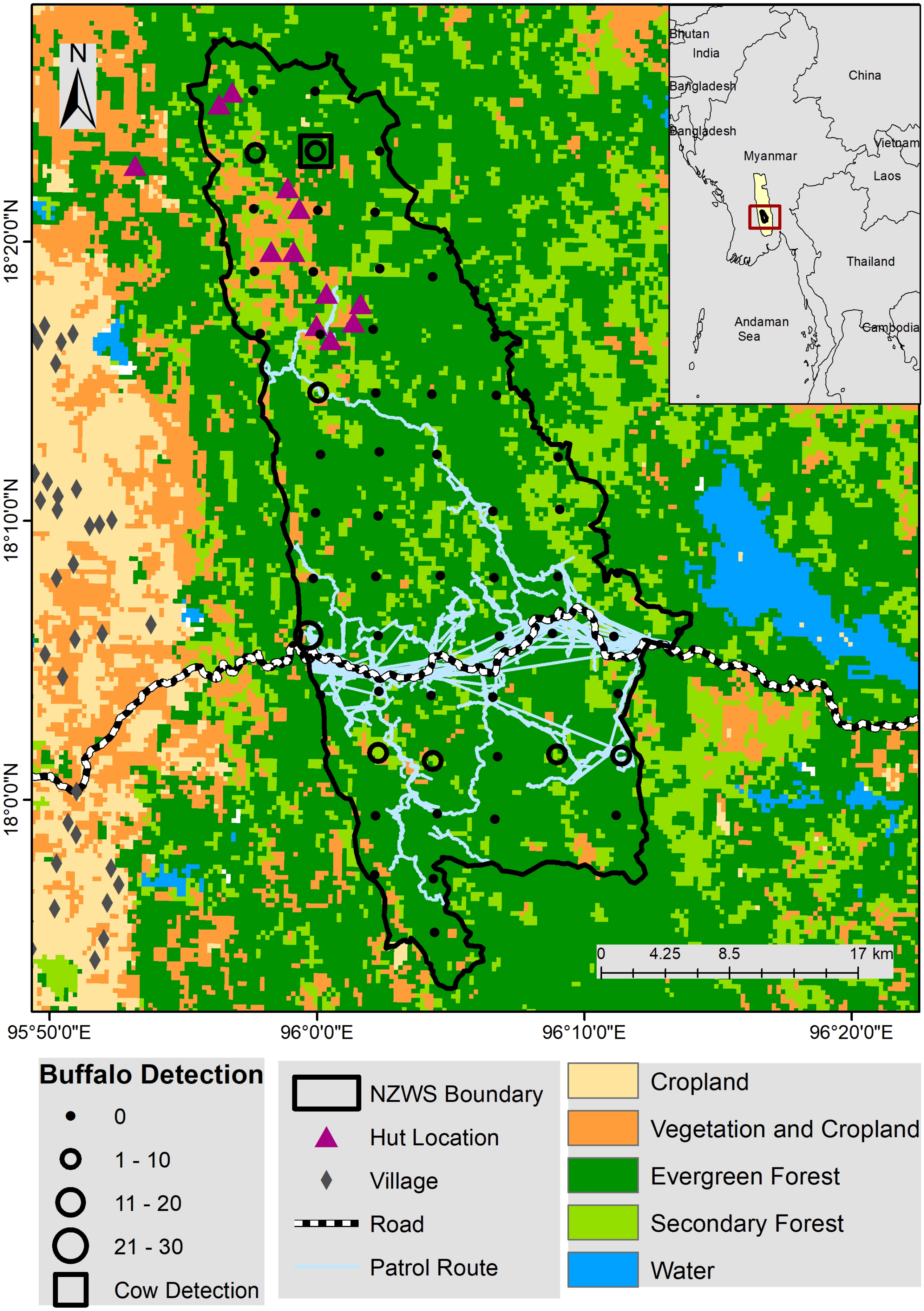

Fig. 4. Map indicating camera trap survey locations from March to June 2020 in North Zarmari Wildlife Sanctuary, Bago Region, Myanmar. Domestic cattle detection per 100 trap days; the figure suggests the presence of human disturbance in the area.

The presence of wild pigs and primates had a strong positive effect on leopard occurrence. The marginal effect noted in relation to the distance to the nearest village indicates that leopards are more inclined to inhabit areas located farther away from human settlements (Fig. 2a). Their avoidance of villages is likely a result of the hunting pressure they face for the purposes of trade and as a source of traditional medicine (Evans et al., 2020). These findings highlight the ecological relationships between leopards and their prey species, particularly the importance of primates and wild pigs in influencing leopard distribution. Understanding these factors can facilitate conservation efforts by helping to identify key areas where leopards are likely to occur. Furthermore, the inclusion of the distance to villages as a marginal factor emphasises the potential impact of human activities on leopard habitat use.

The occurrence of sun bears was influenced by the NDVI value, which serves as an indicator of healthy vegetation and suitable habitat with high availability of food resources (Steinmetz et al., 2013). Sun bears also appeared to favour areas close to year-round streams as forests along these streams offer slightly enhanced habitat conditions compared to other areas. The best predictor of the occurrence of Asiatic black bears was the distance to the nearest village (Fig. 2c). The analysis revealed that Asiatic black bears were more likely to occur in areas closer to villages. This observation indicates a potential association between the presence of human settlements and the occurrence of Asiatic black bears.

Our findings suggest that the omnivorous bears are able to tolerate some level of disturbance perhaps due to food availability from human settlements (Guharajan et al., 2018). Food availability is a significant factor in determining the habitat use and distribution of sun bears (Wong et al., 2013). The co-occurrence of the two bear species in the NZWS suggests that they share similar habitat preferences (Steinmetz et al., 2011). However, their habitat requirements may differ due to their respective food preferences. The degraded forests of the sanctuary may be responsible for this difference. Black bears have a higher demand for fruits and are known to raid crops in human-dominated landscapes. In contrast, sun bears have not been reported to raid crops in mainland southeast Asia (Ngoprasert et al., 2011). Sun bears tend to shift their diet towards insects when there is low availability of fruits (Fredriksson et al., 2006; Steinmetz et al., 2013), which could make them more tolerant of the NZWS rather than risking feeding in human-dominated areas, and also to avoid areas with a high potential for direct encounters with black bears (Ngoprasert et al., 2022). Bears are known to use mixed habitats covering human-altered landscapes, such as agricultural areas, and natural habitats (Scotson, 2010; Guharajan et al., 2018). Similar to bears, other large carnivores can survive in human-dominated areas by relying fully or partially on food resources connected with humans (Athreya et al., 2016). Furthermore, large omnivores such as sun bears and Asiatic black bears are commonly attracted to crops like rice, sesame, corn, and peanuts both within and outside protected areas.

Our key small carnivore species (the Asiatic golden cat, leopard cat, common palm civet, large Indian civet, and crab-eating mongoose) were not distributed close to year-round streams, suggesting that these species may have adapted to drier environments where they can find the necessary resources and suitable habitats for survival, despite the lack of a year-round water source nearby (Balme et al., 2007; Bashir et al., 2020; Sharma et al., 2021). Several species have evolved to meet their water needs through other means, including by extracting water from their prey or from fruits (Zhou et al., 2011; Bu et al., 2016).

Leopard cats are capable of tolerating habitats degraded by human activities, including agricultural landscapes and large plantations, due to the abundance of murids in such landscapes (Chua et al., 2016; Petersen et al., 2019a). Although common palm civets are typically found in forests (Petersen et al., 2019b), our findings suggest they can adapt to certain forms of human-induced disturbance, such as logging, forest edges, and secondary forests (Dehaudt et al., 2022). The occurrence and distribution of the crab-eating mongoose were affected by human disturbances near water sources as well as the presence of leopard cats (Sharma et al., 2021).

Management implications. Our camera trap pictures results show the prevalence of illegal activities within the protected areas, including poaching, illegal logging, forest fires, the presence of huts, and practices such as shifting cultivation. These raise concerns about the long-term management and species distribution in the Bago Yoma range (Fig. S1). The presence of the road inside the protected area has facilitated ease of access contributing to illegal hunting and logging activities. In addition, Bago Yoma holds the longest continuous record of timber production within Myanmar and the likely of illegal logging was higher for larger trees (Saung et al., 2021). Moreover, the current patrol system was enforced for only one year (2020) and was limited to the main road, with very low effort made to cover the rest of the protected area (Fig. 3). To improve protection activities, we suggest focusing patrol efforts on areas facing ongoing threats of poaching (Fig. 3) and the presence of domestic animals (Fig. 4). Spatiotemporal heterogeneity in the patrolling effort explains a great deal of the variation in the detection of poachers and ungulates (Marescot et al., 2020) and can support managers to rejig patrolling strategies, by expanding patrolling efforts to prevent illegal activities (Linkie et al., 2015).

Potential threats should be identified on a finer scale and addressed more effectively by carrying out regular patrolling covering the whole of the protected area. Without effective management and enforcement, these illegal activities can continue unchecked, posing a threat to the biodiversity of the North Zamari Wildlife Sanctuary. The implementation of proactive protection measures is critical for the effective management and conservation of biodiversity within the Bago Yoma range (Coad et al., 2019). In parks where illegal poaching is a significant threat, it is important to consider increasing the number of law enforcement agents (Rao et al., 2002; Aung, 2007). The prioritisation of management planning in protected areas should be based on the specific needs and feasibility, taking into account the significant differences in conditions and requirements. Building the technical capacity to manage protected areas is crucial (Rao et al., 2002; Aung, 2007).

In the end, the prevailing political uncertainty in Myanmar has inflicted severe damage on the country's biodiversity, including that of Zamari. Overall, the study underscores the importance of understanding the various factors that influence the detection and distribution of key species in order to effectively guide conservation efforts. Implementing a comprehensive management action for Zamari requires an adequate understanding of the dynamics in the distribution and interactions between key wildlife species and environmental factors at play in the area.

ACKNOWLEDGMENTS

The project was funded by World Wildlife Fund, WWF-US, Grealish. The first author would like to acknowledge the School of Bioresource and Technology (SBT) for a scholarship during her study period. The first author is also thankful for the financial support in her academic career. We thank the field team members Yan Lin Htun, Zay Yar Soe, Tun Tun Oo, Kaung Htet Paing, Aung Kyaw Myint, Wai Yan Tun, Win Ko Ko Naing Tun, and Soe Htike Aung from Friends of Wildlife, FOW. We thank the Forest Department for granting permission to carry out this research. Particular thanks also go to A. Christy William and Nick Cox for their coordination with government authorities. Special thanks go to Daw Khine Khine Swe and U Myint Aung from Friends of Wildlife, FOW for their cooperation throughout the project.

Author contributions. Project design: Margaret Nyein Nyein Myint, Paing Soe, Zin Mar Hein; data analysis and writing: Margaret Nyein Nyein Myint, Dusit Ngoprasert, Niti Sukumal, Nay Myo Shwe, Tommaso Savini.

LITERATURE CITED

Akaike H (1973) Maximum likelihood identification of Gaussian autoregressive moving average models. Biometrika, 60(2): 255–265.

Arnold TW (2010) Uninformative parameters and model selection using Akaike's Information Criterion. The Journal of Wildlife Management, 74(6): 1175–1178.

Athreya V, Odden M, Linnell JD, Krishnaswamy J & Karanth KU (2016) A cat among the dogs: leopard *Panthera pardus* diet in a human-dominated landscape in western Maharashtra, India. Oryx, 50(1): 156–162.

Aung UM (2007) Policy and practice in Myanmar's protected area system. Journal of Environmental Management, 84(2): 188–203.

Avillés-Polanco G, Jefferson DJ, Almendarez-Hernández MA & Belatrán-Morales LF (2019) Factors that explain the utilization of the Nagoya protocol framework for access and benefit sharing. Sustainability, 11(20): 1050.

Balme G, Hunter L & Slotow ROB (2007) Feeding habitat selection by hunting leopards *Panthera pardus* in a woodland savanna: prey catchability versus abundance. Animal Behaviour, 74(3): 589–598.

Bashir F, Nawab M, Ganai BA & Bashir T (2020) Patterns of resource use by Asiatic black bear *Ursus thibetanus* during pre-hibernation in Kashmir Himalaya, India. Journal of Natural History, 54(37–38): 2455–2469.

Bonham CA, Sacayon E & Tzi E (2008) Protecting imperiled "paper parks": potential lessons from the Sierra Chinajá, Guatemala. Biodiversity and Conservation, 17: 1581–1593.

Borowik T, Pettorelli N, Sönnichsen L & Jędrzejewska B (2013) Normalized difference vegetation index (NDVI) as a predictor of forage availability for ungulates in forest and field habitats. European Journal of Wildlife Research, 59: 675–682.

Bu H, Wang F, McShea WJ, Lu Z, Wang D & Li S (2016) Spatial co-occurrence and activity patterns of mesocarnivores in the temperate forests of Southwest China. PLOS One, 11(10): e0164271.

Burnham KP & Anderson DR (2002) Model Selection and Multimodel Inference: A Practical Information–Theoretic Approach. Springer, New York, USA, 488 pp.

Chua MAH, Sivasothi N & Meier R (2016) Population density, spatiotemporal use and diet of the leopard cat (*Prionailurus bengalensis*) in a human-modified succession forest landscape of Singapore. Mammal Research, 61: 99–108.

Coad L, Watson JE, Geldmann J, Burgess ND, Leverington F, Hockings M, Knights K & Di Marco M (2019) Widespread shortfalls in protected area resourcing undermine efforts to conserve biodiversity. Frontiers in Ecology and the Environment, 17(5): 259–264.

Dehaudt B, Amir Z, Decoeur H, Gibson L, Mendes C, Moore JH, Nursamsi I, Sovie A & Luskin MS (2022) Common palm civets *Paradoxurus hermaphroditus* are positively associated with humans and forest degradation with implications for seed dispersal and zoonotic diseases. Journal of Animal Ecology, 91(4): 794–804.

Duckworth JW, Sankar K, Williams AC, Kumar NS & Timmins RJ (2016) *Bos gaurus*. The IUCN Red List of Threatened Species. http://dx.doi.org/10.2305/IUCN.UK.2016-2.RLTS.T2891A46363646.en (Accessed 25 May 2016).

Evans TS, Myat TW, Aung P, Oo ZM, Maw MT, Toe AT, Aung TH, Hom NS, Shein KT, Thant KZ, Win YT, Thein WZ, Gilardi K, Thu HM & Johnson CK (2020) Bushmeat hunting and trade in Myanmar's central teak forests: Threats to biodiversity and human livelihoods. Global Ecology and Conservation, 22: e00889.

Franklin J, Wejnert KE, Hathaway SA, Rochester CJ & Fisher RN (2009) Effect of species rarity on the accuracy of species distribution models for reptiles and amphibians in southern California. Diversity and Distributions, 15(1): 167–177.

Fredriksson GM, Wich SA & Trisno (2006) Frugivory in sun bears (*Helarctos malayanus*) is linked to El Nino-related fluctuations in fruiting phenology, East Kalimantan, Indonesia. Biological Journal of the Linnean Society 89: 489–508.

Freeman EA & Moisen GG (2008) A comparison of the performance of threshold criteria for binary classification in terms of predicted prevalence and kappa. Ecological Modeling, 217(1–2): 48–58.

Gelman A & Hill J (2006) Data Analysis Using Regression and Multilevel/Hierarchical Models. Cambridge University Press, Cambridge, 648 pp.

Grassman LI, Tewes ME, Silvy NJ & Kreetiyutanont K (2005) Ecology of three sympatric felids in a mixed evergreen forest in north-central Thailand. Journal of Mammalogy, 86(1): 29–38.

Guharajan R, Arnold TW, Bolongon G, Dibden GH, Abram NK, Teoh SW, Magguna MA, Goossens B, Wong ST, Nathan SKSS & Garshelis DL (2018) Survival strategies of a frugivore, the sun bear, in a forest-oil palm landscape. Biodiversity and Conservation, 27: 3657–3677.

Hartig F (2020) DHARMa: residual diagnostics for hierarchical (multi-level/mixed) regression models. R package version 0.3.3. https://cran.r-project.org/package=DHARMa (Accessed 22 August 2020).

Hayward MW, Henschel P, O'Brien J, Hofmeyr M, Balme G & Kerley GI (2006) Prey preferences of the leopard (*Panthera pardus*). Journal of Zoology, 270(2): 298–313.

Hein ZM, Williams A., Soe P, Cox NJ, Htun NZ, Oo TN, Aye YY, Htun YL & Yoganand K (2020) Status of two species of threatened wild cattle (*Bos gaurus* and *Bos javanicus birmanicus*) in North Zamari Wildlife Sanctuary, Bago Region, Myanmar. BULLetin, 5: 18–26.

Hosmer DW & Lemeshow S (2000) Applied logistic regression. 2nd edition. John Wiley, New York, USA, 375 pp.

Kamler JF, Inthapanya X, Rasphone A, Bousa A, Vongkhamheng C, Johnson A & Macdonald DW (2020) Diet, prey selection, and activity of Asian golden cats and leopard cats in northern Laos. Journal of Mammalogy, 101(5): 1267–1278.

Kawanishi K & Sunquist ME (2008) Food habits and activity patterns of the Asiatic golden cat (*Catopuma temminckii*) and dhole (*Cuon alpinus*) in a primary rainforest of Peninsular Malaysia. Mammal Study, 33(4): 173–177.

Khoewsree N, Pla-ard M, Sukmasuang R, Paansri P, Chanachai Y, Kaewdee B & Phengthong P (2022) Spatio-temporal analysis of dholes (*Cuon alpinus*) in Khao Yai National Park, Thailand. Biodiversitas Journal of Biological Diversity, 23(5): 2668–2678.

Kolowski JM & Forrester TD (2017) Camera trap placement and the potential for bias due to trails and other features. PLOS One, 12(10): e0186679.

Kyaw TY (2018) The Use of Landsat Imagery and Spatial Analysis to Detect Forest Cover Change and Degradation, and Determine Forest Management Suitability Indices in the Bago Mountain Region of Myanmar. Unpublished MSc Thesis. State University of New York College of Environmental Science and Forestry. Syracuse, New York, 180 pp.

Linkie M, Martyr DJ, Harihar A, Risdianto D, Nugraha RT, Leader Williams N & Wong WM (2015) Safeguarding Sumatran tigers: evaluating effectiveness of law enforcement patrols and local informant networks. Journal of Applied Ecology, 52(4): 851–860.

Lüdecke D, Ben-Shachar MS, Patil I, Waggoner P & Makowski D (2021) performance: An R package for assessment, comparison and testing of statistical models. Journal of Open Source Software, 6(60): 3139.

Lwin N, Ngoprasert D, Sukumal N, Browne S & Savini T (2022) Status and distribution of hoolock gibbon in the newly established Indawgyi Biosphere Reserve: Implication for protected area management. Global Ecology and Conservation, 38: e02209.

Mak GJK & Song W (2018) Transnational norms and governing illegal wildlife trade in China and Japan: elephant ivory and related products under CITES. Cambridge Review of International Affairs, 31: 373e391.

Marescot L, Lyet A, Singh R, Carter N & Gimenez O (2020) Inferring wildlife poaching in southeast Asia with multispecies dynamic occupancy models. Ecography, 43(2): 239–250.

Masiliūnas D, Tsendbazar NE, Herold M, Lesiv M, Buchhorn M & Verbesselt J (2021) Global land characterisation using land cover fractions at 100 m resolution. Remote Sensing of Environment, 259: 112409.

Mazerolle MJ (2021) AICcmodavg: Model selection and multimodel inference based on (Q) AIC (c). R package version 2.3–1.2020. https://cran.r-project.org/package=AICcmodavg (Accessed 27 December 2021).

Ministry of the Natural Resources and Environmental Conservation of Myanmar Forest Department (2015) National Biodiversity Strategy and Action Plan 2015–2020, Forest Department, Myanmar, 85 pp.

Mon MS, Kajisa T, Mizoue N & Yoshida S (2010) Monitoring deforestation and forest degradation in the Bago Mountain Area, Myanmar using FCD Mapper. Journal of Forest Planning, 15(2): 63–72.

Naing Tun Z, Dargusch P, McMoran DJ, McAlpine C & Hill G (2021) Patterns and drivers of deforestation and forest degradation in Myanmar. Sustainability, 13(14): 7539.

Ngoprasert D, Steinmetz R, Sribuarod K & Gale GA (2022) The overlap of sympatric sun bears and Asiatic black bears in space and time. Mammalian Biology, 102:143–153.

Ngoprasert D, Steinmetz R, Reed DH, Savini T & Gale GA (2011) Influence of fruit on habitat selection of Asian bears in a tropical forest. The Journal of Wildlife Management, 75(3): 588–595.

O'Brien TG, Kinnaird MF & Wibisono HT (2003) Crouching tigers, hidden prey: Sumatran tiger and prey populations in a tropical forest landscape. Animal Conservation, 6(2): 131–139.

Oindo BO (2002) Predicting mammal species richness and abundance using multi-temporal NDVI. Photogrammetric Engineering and Remote Sensing, 68(6): 623–629.

Oksanen J (2010) Vegan: community ecology package. http://CRAN. R-project. org/package= vegan. https://cran.r-project.org/package=vegan (Accessed 15 May 2023).

Pakpien S, Simcharoen A, Duangchantrasiri S, Chimchome V, Pongpattannurak N & Smith JL (2017) Ecological covariates at kill sites influence tiger (*Panthera tigris*) hunting success in Huai Kha Khaeng Wildlife Sanctuary, Thailand. Tropical Conservation Science, 10: 1–7.

Petersen WJ, Savini T, Steinmetz R & Ngoprasert D (2019a) Estimating Leopard Cat *Prionailurus bengalensis* Kerr, 1792 (Carnivora: Felidae) density in a degraded tropical forest fragment in northeastern Thailand. Journal of Threatened Taxa, 11(4): 13448–13458.

Petersen W J, Savini T, Steinmetz R & Ngoprasert D (2019b) Periodic resource scarcity and potential for interspecific competition influences distribution of small carnivores in a seasonally dry tropical forest fragment. Mammalian Biology, 95(1): 112–122.

R Core Team (2022) R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org/.

Rao M, Rabinowitz A & Khaing ST (2002) Status review of the protected area system in Myanmar, with recommendations for conservation planning. Conservation Biology, 16(2): 360–368.

Rasphone A, Bousa A, Vongkhamheng C, Kamle JF, Johnson A & Macdonald DW (2022) Diet and prey selection of clouded leopards and tigers in Laos. Ecology and Evolution, 12(7): e9067.

Richards SA (2008) Dealing with overdispersed count data in applied ecology. Journal of Applied Ecology, 45(1): 218–227.

Roos C, Helgen KM, Miguez RP, Thant NML, Lwin N, Lin AK, Lin A, Yi KM, Soe P, Hein ZM, Myint NNM, Ahmed T, Chetry D, Urh M, Veatch EG, Duncan N, Kamminga P, Chua MAH, Yao L, Matauschek C, Meyer D, Liu ZJ, Li M, Nadler T, Fan PF, Quyet LK, Hofreiter M, Zinner D & Momberg F (2020) Mitogenomic phylogeny of the Asian colobine genus *Trachypithecus* with special focus on *Trachypithecus phayrei* (Blyth, 1847) and description of a new species. Zoological Research, 41(6): 656.

Rustad SA & Binningsbø HM (2012) A price worth fighting for? Natural resources and conflict recurrence. Journal of Peace Research, 49(4): 531–546.

Sanderson J & Harris G (2013) Automatic data organization, storage, and analysis of camera trap pictures. Journal of Indonesian Natural History, 1(1): 11–19.

Saung T, Khai TC, Mizoue N, Ota T & Kajisa T (2021) Condition of illegally logged stands following high frequency legal logging in Bago Yoma, Myanmar. Forests, 12(2): 115.

Scotson L (2010) The distribution and status of Asiatic Black bear *Ursus thibetanus* and Malayan Sun bear *Helarctos malayanus* in Nam et Phou Louey national protected area, Lao PDR. Unpublished report to International Association for Bear Research and Management, pp. 29–30.

Sharma HP, Adhikari B, Bhandari S, Baral K & Kunwar RM (2021) Crab-eating mongoose *Herpestes urva*: occurrence and its activity in Mid-Hills of Nepal. Journal of Institute of Science and Technology, 26(2): 53–60.

Simcharoen A, Savini T, Gale GA, Roche E, Chimchome V & Smith JL (2014) Ecological factors that influence sambar (*Rusa unicolor*) distribution and abundance in western Thailand: implications for tiger conservation. Raffles Bulletin of Zoology, 62: 100–106.

Sollmann R (2018) A gentle introduction to camera-trap data analysis. African Journal of Ecology, 56: 740–749.

Steinmetz R, Garshelis DL, Chutipong W & Seuaturien N (2011) The shared preference niche of sympatric Asiatic black bears and sun bears in a tropical forest mosaic. PLOS One, 6(1): e14509.

Steinmetz R, Garshelis DL, Chutipong W & Seuaturien N (2013) Foraging ecology and coexistence of Asiatic black bears and sun bears in a seasonal tropical forest in Southeast Asia. Journal of Mammalogy, 94(1): 1–18.

Sukumal N, Dowell SD & Savini T (2020) Modelling occurrence probability of the Endangered green peafowl *Pavo muticus* in mainland South-east Asia: applications for landscape conservation and management. Oryx, 54(1): 30–39.

Sutherland C, Hare D, Johnson PJ, Linden DW, Montgomery RA & Droge E (2023). Practical advice on variable selection and reporting using Akaike information criterion. Proceedings of the Royal Society B, 290: 20231261.

Wei T & Simko VR (2021) R package ‘corrplot’: Visualization of a Correlation Matrix (Version 0.92). https://cran.r-project.org/package=corrplot (Accessed 27 October 2022).

Wickham H, Chang, W & Wickham MH (2016) Package ‘ggplot2’. Create elegant data visualisations using the grammar of graphics. Version 2.1, 1-189. http://ggplot2.org, https://github.com/hadley/ggplot2 (Accessed 3 February 2023).

Williams HF, Bartholomew DC, Amakobe B & Githiru M (2018) Environmental factors affecting the distribution of African elephants in the Kasigau wildlife corridor, SE Kenya. African Journal of Ecology, 56(2): 244–253.

Win L, Sukumal N, Shwe NM & Savini T (2023) Teak plantations are complementary habitat for Green Peafowl within the fragmented Bago Yoma range, south-central Myanmar. Ornithological Application, 125: duac051.

Wong WM, Leader Williams N & Linkie M (2013) Quantifying changes in sun bear distribution and their forest habitat in Sumatra. Animal Conservation, 16(2): 216–223.

Zhou YB, Newman C, Buesching CD, Zalewski A, Kaneko Y, Macdonald DW & Xie ZQ (2011) Diet of an opportunistically frugivorous carnivore, *Martes flavigula*, in subtropical forest. Journal of Mammalogy, 92(3): 611–619.

Zuur AF, Ieno EN & Elphick CS (2010) A protocol for data exploration to avoid common statistical problems. Methods in Ecology and Evolution. 1: 3–14.

SUPPLEMENTARY DATA

The following supplementary data are available online:

Supplementary figures Fig. S1 (https://figshare.com/articles/figure/Supplementary_Figures/25712580)
Supplementary Table S1: (https://figshare.com/articles/dataset/Supplimentary_Table_S1_docx/25712517)