Factors Affecting the Distribution of Malayan Sun Bear in Htamanthi Wildlife Sanctuary, Northern Myanmar

Min Hein Htike
University of Massachusetts Amherst

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FACTORS AFFECTING THE DISTRIBUTION OF MALAYAN SUN BEARS IN
HTAMANTHI WILDLIFE SANCTUARY, NORTHERN MYANMAR

A Thesis Presented

By

MIN HEIN HTIKE

Submitted to the Graduate School of the University of Massachusetts Amherst in partial fulfillment of the requirements for the degree of

MASTER OF SCIENCE

MAY 2023

Environmental Conservation
FACTORS AFFECTING THE DISTRIBUTION OF MALAYAN SUN BEARS IN
HTAMANTHI WILDLIFE SANCTUARY, NORTHERN MYANMAR

A Thesis Presented
By
Min Hein Htike

Approved as to style and content by:

______________________________
Curtice R. Griffin, Co-Chair

______________________________
Todd K. Fuller, Co-Chair

______________________________
Tammy L. Wilson Member

______________________________
Paige Warren, Department Head,
Environmental Conservation
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ABSTRACT

FACTORS AFFECTING THE DISTRIBUTION OF MALAYAN SUN BEARS IN HTAMANTHI WILDLIFE SANCTUARY, NORTHERN MYANMAR

MAY 2023

MIN HEIN HTIKE, B.S, UNIVERSITY OF FORESTRY YEZIN,
BURMA/MYANMAR

M.S., UNIVERSITY OF MASSACHUSETTS AMHERST

Directed by: Professors Curtice R. Griffin and Todd K. Fuller

To understand the modeling challenges and to examine the important factors considered in Malayan sun bear (*Helarctos malayanus*) distribution studies, we reviewed 33 peer-reviewed articles published from 2003-2023. These studies used 54 environmental or anthropogenic variable types to investigate the distribution, habitat preference, and home range composition of sun bears. Most variable types are human disturbance (n=4), climate (n=3), topography (n=1), vegetation (n=11), or other ecological factors (n=3). Nevertheless, a number of rarely used variables might also be useful to include in future evaluations (i.e., food abundance), and observational evidence suggests that predator occurrence could also be informative. Importantly, no studies tested the performance of model prediction by using other presence points of the species in a similar or adjacent biogeographical area.

In Myanmar, where the bear’s distribution is not well-known, we set up three annual surveys using 120 camera-trap stations in a portion of the Htamanthi Wildlife Sanctuary (HWS) in northern Myanmar during 2016-17 to 2018-19 to identify factors
influencing bear distribution. From a total effort of 15,315 trap nights, we obtained 47 independent photo events of sun bears at 16%, 13%, and 9% of the stations each year.

We analyzed eight factors potentially influencing bear distribution and found that the top three ranked models were a combination of elevation, NDVI (Normalized Difference Vegetation Index), distance to water and slope. The presence of tigers (*Panthera tigris*) in the area was found to have a positive relation with mean sun bear occupancy.

In this study, we tested the prediction performance of the single-season occupancy model with another dataset. We tested the prediction performance of the top six models in the PresenceAbsence Package and calculated the AUC (Area under receiver curved), TSS (True skill statistics), and Kappa scores. The AUC score ranged from 0.5 to 0.6, while the TSS score ranged from -0.001 to 0.28. None of the top six models’ predictions perfectly agreed with the sanctuary-wide survey data. The discrepancies may be due to the limited sample size, the temporal scale of the prediction, and the presence of other ecological factors (e.g., predators, competitors, or food availability) not accounted for in the habitat use prediction. To improve the prediction performance of occupancy models, we recommend that future sun bear surveys increase the number and size of sampling effort and include ecological covariates such as potential predators when possible.
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CHAPTER - 1

HABITAT DISTRIBUTION MODELING OF MALAYAN SUN BEARS: A REVIEW

Abstract - To understand the modeling challenges and to examine the important factors considered in Malayan sun bear (*Helarctos malayanus*) distribution studies, we reviewed 33 peer-reviewed articles published from 2003-2023. These studies used 54 environmental or anthropogenic variable types to investigate the distribution, habitat preference, and home range composition of sun bears. Most variable types (74%; n = 40) were included in only 1 or 2 studies, and few (41%; n = 22) were found or considered to be important or significant enough to include in models; these included human disturbance (n=4), climate (n=3), topography (n=1), vegetation (n=11), or other ecological factors (n=3). Some of these variable types (Annual temperature, Precipitation [annual and driest month] and moisture, Enhanced Vegetation Index [EVI], Above Ground Carbon Density [ACD] and Forest cover) were most relevant to regional studies, but others (e.g., Distances to reserve boundary, roads, settlements, freshwater, and forest; Human activities; Elevation; Vegetation [indices, fruit, cover, type]; bee and termite nests) were relevant to modeling in specific study areas. Still, only 1 of 10 variables included in ≥3 studies were identified as important or significant 50% of the time (Forest/Habitat type). Nevertheless, a number of rarely used variables might also be useful to include in future evaluations (i.e., food abundance), and observational evidence suggests that predator occurrence could also be informative. Importantly, no studies tested the performance of model prediction by using other presence points of the species in a similar or adjacent biogeographical area.
Keywords. covariates, *Helarctos malayanus*, literature review, occurrence, scale.

Introduction

Malayan sun bears (*Helarctos malayanus*) are the least-known of all bear species (Crudge et al., 2019). In many parts of their range (Fig. 1), sun bears occur in areas undergoing rapid deforestation (Hansen et al., 2013) and where poaching and wildlife trafficking are common (McEvoy et al., 2019). In some places these bears occur in areas of contested authority (e.g., ongoing civil war), making law enforcement nearly impossible. Thus, species numbers are believed to have declined ~35% over the past 30 years (Scotson et al., 2017a), primarily due to hunting and habitat loss (Tee et al., 2021). Globally, the sun bear is listed as Vulnerable on the IUCN (International Association for Conservation of Nature) Red List (Scotson et al., 2017a).

In a review of habitat associations for all species of bears, Garshelis (2022) reported several problematic issues in understanding habitat associations including: “biases in presence data; predictor variables being poor surrogates of actual behavioral drivers; predictor variables applied at a biologically inappropriate scale; and over-use of data repositories that tend to detach investigators from the species. In several cases, multiple models in the same area yielded different predictions; new presence data occurred outside the range of predicted suitable habitat; and future range projections, based on where bears presently exist, underestimated their adaptability” (p.1). So, to help assess the extent to which such problems are valid for our current understanding of sun bear habitat associations and to help better inform conservation actions, we conducted a review of what has been published to date.
Methods and materials

We conducted a comprehensive literature review of factors influencing sun bear distribution and abundance using two Internet search engines: Web of Science (https://www.webofscience.com/WOS, accessed 15 October 2022 and 10 February 2023) and Google Scholar (https://scholar.google.com, accessed 15 October 2022 and 10 February 2023). A systematic search was delimited from 1980 to 2023 and used the following words and phrases alone or in combination to identify relevant papers: "sun bear," "Helarctos malayanus," "distribution," "environmental variables," "food abundance," "habitat," "occupancy," and "modeling".

For each relevant publication, we identified data collection methods, methods of analysis used to inform sun bear distribution, geographic scale of the assessment, and a list of variables included in the assessment. We sorted the data collection methods into eight categories: camera trap, transect, radio collar, barrel traps, interview, GPS (Global Positioning System; secondary presence points), literature review, genetic sampling; and some studies were based on data from multiple sources. The analysis methods were separated into five categories: basic statistics (includes a number of methods such as comparison tests, generalized lineal models, and analysis of covariance), home range analysis (composition of home ranges or selection of ranges within a study area), niche modeling (species-distribution models that used presence data to infer ecological requirements to elucidate potential distributions), occupancy modeling (specific modeling approach base on the proportion of areas or sample units occupied), and other (no model; characteristics based on previous knowledge and species-environment associations from expert opinion to envisage distribution). We also used the geographical extent of each
study to pool studies into two scale categories: regional (range- or country-wide assessments, or studies assessing data from multiple study areas that provide regional scale interpretation), and local (single/specific study areas, usually of <10,000 km²). Some studies were conducted at multiple study areas (<10,000 km²), but conducted analyses for each study area and presented results individually. These studies were put under local scale rather than regional scale.

Because published variable names use different or unique databases to evaluate sun bear occurrence, each variable is described with the methods used to obtain and apply it. Variables identified in models as statistically significant were noted. We also noted variables that studies identified as important that are not traditionally considered in models (i.e., summaries of field data informing distribution assessments); yet, we considered these non-traditional variables as “models”, considering they represent conceptual representations of an ecological system or process. Similar variables with different names were then classified into one-name variables and sorted into sub-categories within the two broader categories of anthropogenic and environmental factors. A comprehensive list of the variables identified in a paper as statistically significant, included in the “best model, or otherwise important was created and considered for use in modeling data for other study areas. We also compare the results of this review to other assessments of sun bear habitat use.

**Results**

*General summary*

We identified 33 peer-reviewed studies in our search. They occurred mostly in Indonesia (n = 10), and Malaysia (n = 10) plus one region-wide study encompassing the
southern regions of sun bear range (Thailand, Malaysia, and Indonesia). Additionally, four region-wide studies were conducted in Malaysia (n=3) and India (n=1). There were five studies in northeast India, representing the northern range limit of sun bears (Fig. 1.2A). All studies were published since 2002, with most during 2011-2022 (Fig. 1.2B).

The use of camera traps (58%; n = 19), and sign transects (42%; n = 14) were the most common methods used (Table 1.1); many studies used both camera traps and transects in the same study, and other techniques were used less often. Five studies were conducted during year-round surveys, while the others were conducted in different seasons of the years with surveys from June to October (wet season) being most common (n = 10-12; Fig. 1.3). Five region-wide studies (based on secondary presence points of sun bears not from actual surveys) were conducted within the range of sun bears. A majority of the studies (n = 27) were conducted in designated wildlife sanctuaries, forest reserves, national parks, and other conservation areas (Table 1.1). Studies covered many ecosystems, including different types of tropical forests and montane forests, as well as some temperate forest types in northeastern India. There were four main modeling approaches used; including niche modeling (36%; n = 12), occupancy modeling (30%; n = 10), basic statistics (18%; n = 6) and home range analysis (9%; n = 3).

**Variables**

Fifty-four different variables were used across the 33 reviewed studies to investigate the distribution, habitat preference, and home range of the Malayan sun bear (Tables 1.2 and 1.3). These variables were summarized into 5 general types, including human disturbance (n = 9), climatic (n = 10), topographic (n = 4), other ecological (n = 9), and vegetation (n = 22). Of the 54 variables, 32 (59%) had no support for being
included in habitat description models (Table 1.2), but the other 22 (41%) were reported at least once as important or significant (Table 1.3).

Anthropogenic variables were related to human disturbance, including distance to reserve boundary, distance to roads, distance to settlements or villages, and human activities. Only some (17%) of the 23 publications described them as significant negative variables affecting on the distribution of sun bears (Table 1.3). Distance to reserve boundary was reported as a significant variable in one publication. Distance to roads was considered an important variable in eight studies but reported as a significant variable in only one study. Distance to settlements/villages and human activities were used as variables in 10 and five studies, respectively; however, they were reported as important variables in only two publications (Table 1.3). All of the human disturbance variables were identified as significant only in local-scale assessments, except distance to settlement. Distance to settlements/villages was identified also identified as significant in one regional-scale assessment.

Among the environmental variables, the most commonly used variables were climatic (n = 6), topographic (n = 14), vegetation (n = 41), and other ecological variables (n = 14). Climatic variables such as annual temperature, annual precipitation, and precipitation of driest month were important only in one study at the regional scale (Table 1.3). For the topographic variables, elevation variable was described as significant predictor of sun bear presence, regardless of the assessment of other topographic features such as aspect, slope, and terrain roughness. Elevation was suggested as an important covariate by 50% (n = 7) of the publications that included it, and all of these studies occurred at the local scale.
Many papers assessed vegetation-related variables such as: Distance to forest (n = 4), Enhanced vegetation index (n = 3), Fig tree (number of fruit trees; n = 3), Fruit abundance (n = 1), Forest/vegetation/canopy cover (n = 13), Forest/Habitat type (n = 8), Ground vegetation cover (n = 5), Normalized difference moisture index (n = 1), Refugia (escape cover; n = 2), and Types of flora (n = 1). However, only 34% of the publications that used one of these variables reported them as important or significant, and these occurred at both the local and regional scales (Table 1.3).

Other ecological factors such as presence of termite nests (n = 1), bee nests (n = 1), and distance to water (n = 12) were also used to assess bear foraging or distribution patterns in natural habitat. Presence of termite nests and bee nests were reported as important by one local-scale study that included them, while the distance to water was identified as important for only 2 of the 12 publications that included this variable (both at the local scale). In total, other ecological variables were considered important for 29% of 14 studies that used them.

Discussion

Data Collection methods

Sun bear presence data are usually retrieved from a camera trap, sign transects, and interview surveys (Table 1.1). However, camera traps are more likely to detect the species than transect surveys (Bisi et al., 2019), likely because frequent rains in tropical climates usually wash away tracks and signs and makes it challenging to observe them. Moreover, occupancy results generated from camera trap observations are more consistent and uniform than the results obtained from transect observations, especially when sampling effort varies (Bisi et al., 2019), and variation in transect distance may
result in different occupancy results. However, only 21% (n=7) of the papers described
the spacing of camera traps (Appendix S1). The spacing ranged from 1-4 km between
cameras, usually based on the predicted home range sizes of sun bears (e.g., 1.2-5.1 km²
in Sabah, Malaysia [Normua et al., 2004]) and 14.8 km² in Sumatra, Indonesia [Wong et
al., 2004]). The survey seasons also varied from year-round to single-month studies.

Considering that food availability likely influences bear activity patterns and occurrence
in an area, variation in the timing of surveys can very likely affect sun bear abundance
and distribution results. Nevertheless, the spacing, number of camera traps used for the
specific study area, and seasonality are typically determined by available budget and
human resources. With the exception of Bisi et al. (2019), no studies addressed the
potential effects of camera trap spacing and sampling effort on their sun bear distribution
results.

**Modeling approach**

The majority of sun bear distribution studies used presence-absence surveys via
camera traps, and many studies used presence-only models, such as Ecological Niche
Factor Analysis (ENFA) and Maximum Entropy Modeling (MaxEnt). This detection/non-
detection sampling framework is robust because it accounts for imperfect detection and
allows for several predictor variables to be included, providing inferences about the
factors influencing sun bear habitat use across multiple scales (Gray et al., 2010).

Almost all of the sun bear distribution studies, except those that used basic
statistics, addressed the issues of multicollinearity and variable selection, and clearly
described the model settings, packages used, and algorithms. However, few studies
included model averaging in their analyses. Zurell et al., (2020) strongly encouraged the
use of model averaging in species distribution models. However, model averaging can lead to more uncertainty of the prediction. Therefore, models might be averaged using an unweighted consensus method or using weighted averaging based on the cross-validation or resampling approaches (Dormann et al., 2018). Among 22 papers that used occupancy models and niche models, only three papers continued to model projection or prediction outputs, and none of the studies tested their model’s prediction performance. Instead, they provided the performance statistics, such as the AUC (area under receiver operating curve – ROC) and AIC (Akaike information criterion). Shabani et al. (2018) reported that AUC is an appropriate performance statistic to use when true-presence absence data are available. However, most of the models in sun bear distribution studies used pseudo-absence data; thus, the AUC statistic may not be the appropriate statistic to determine performance of these models.

**Covariates**

Anthropogenic variables examined in sun bear studies, such as distance to reserve boundary, distance to roads, distance to settlements, and human activities reflect that these studies were done in protected areas surrounded, in part, by humans and their infrastructure. Tee et al. (2021) reported that distance to reserve boundary affected sun bear distribution in Borneo where development activities, such as oil palm plantations and road construction around the reserve make sun bear habitat more accessible by humans. Wong et al. (2013) reported that roads increase habitat fragmentation and accessibility to sun bear habitat in Sumatra, and that the presence of sun bears is strictly related to the distance to roads. Similarly, Wong and Linkie (2013) reported that the presence of villages around sun bear habitat can significantly reduce habitat suitability for
bears. In Myanmar, human settlements and associated human activities, such as poaching, mining and non-wood-forest-products collection, negatively affected sun bear presence (Cremonesi et al., 2021). In Borneo, Guharajan et al. (2018) suggested that avoidance of human disturbance was a more important factor than food abundance for sun bears in the Lower Kinabatangan Wildlife Sanctuary, when they observed fewer sun bear signs in corridor areas with more human disturbance even when food appeared abundant. Yet, in the Deramakot Forest Reserve and Tangkulap in Borneo, Guharajan et al. (2021) reported that sun bears utilized oil palm plantations at night for food. These observations suggest that some sun bears are highly adaptive to some kinds of human disturbance.

Some studies reported on the activity patterns of sun bears in mainland Southeast Asia and insular Asia. In a telemetry study of four bears in Sabah Malaysia, Norma et al. (2004) reported that 100% of bear activity was nocturnal, and they usually shifted their activities to nocturnal behaviors in areas with higher human activity. In Malaysia, Wong et al. (2004) reported that male sun bears were primarily diurnal, but a few individuals were active at night for short periods. In the Dampa Tiger Reserve in India, Gouda et al., (2020) reported no bear activity during the mid-day, and that bears were more active during twilight.

Environmental factors, including climatic, topographic, vegetation and other ecological variables, are also important in determining the habitat use of sun bears. Few studies included climatic factors, such as precipitation and temperature, in their sun bear habitat models (Nazeri et al., 2012). Yet, these climatic factors directly affect vegetation
structure; thus, presumably sun bear distribution, which may be more apparent at a regional scale.

Among the topographic variables, elevation was most frequently used as a covariate in sun bear habitat models where slight elevation changes can affect vegetation structure and, consequently, food availability for sun bears (Tee et al., 2021). Slope, terrain roughness index (TRI) and aspect were not identified as important variables in sun bear studies, but this could be related to survey effort. For example, camera traps are usually positioned along trails, roads, ridge lines and areas with less slope to maximize the detection probability of target species. Consequently, the sampling data points already avoid steep slopes with higher TRI. Similarly, transect survey studies are frequently conducted in areas of reduced slope, potentially masking the importance of slope in sun bear studies, but radio telemetry studies can avoid those kinds of biases.

As a forest-dependent species that feeds mainly on fruits and insects, sun bears favor interior mature and heterogeneously structured primary forests (Augeri, 2005). Yet, sun bears are generalist omnivores, feeding on termites, ants, beetle larvae, stingless bee larvae, honey, and a large variety of fruit species such as figs (*Ficus* spp.) (Wong et al., 2002). They usually spend most of their time on the ground, preferring to feed on fallen fruits (Wong et al., 2013), and males usually spend more time foraging than females (Normua et al., 2004). In western Thailand, fruits of Lauraceae, Fagaceae, Leguminosae, Labiatae, and Sapindaceae are the most commonly consumed by sun bears (Steinmetz et al., 2013), while fruits of the families Moraceae, Burseraceae, and Myrtaceae are primarily consumed in Borneo (Frederick et al., 2012). Wong et al. (2004) reported that sun bears use fallen hollow logs, tree cavities and tree branches high above the ground as
bedding sites. Steinmetz et al. (2013) reported that forest types, such as bamboo forest, secondary forest, and primary forest determine food availability and affect sun bear distribution. In their regional study, Scotson et al. (2017b) reported that sun bear detection probability is positively correlated with the percent of tree cover.

Studies in Borneo and Sumatra by Wong and Linkie (2013) reported that the presence of sun bears is strictly related to the presence of primary forests, and they avoid secondary forests of all ages. In contrast, Guharajan et al. (2021) reported that sun bears in Borneo foraged in palm plantations. Gouda et al. (2020) noted that sun bears utilize secondary forest/ bamboo forest because the abundance of old logs for bedding sites as well as food sources. Tee et al. (2021) reported higher sun bear presence in areas with lower percent natural forest cover. But in general, sun bears there are probably trying to avoid human disturbance and prefer primary forest when oil palm plantations bound the habitat of sun bear (but see Guharajan et al. 2021). In the Namdapha Tiger Reserve, India, sun bears used both disturbed and undisturbed habitats at different elevations (Sethy and Chauhan, 2016). In Dampa Tiger Reserve, India, Gouda et al. (2020) reported more bear sign for both Asiatic black bears and sun bears in bamboo forests compared to moist deciduous forests. They reported that that these secondary forests provide termite mounds and large wooden logs that provide important food resources for both bear species. Similarly, Hwang et al. (2021) reported that sun bears were more likely to be observed in areas with termites or bees, which are usually more common in logged areas or secondary forest, but sun bear occurrence was low in newly logged forests compared to old-logged forest. As an indicator for forest degradation and regeneration, Guharajan et al. (2021) used the Normalized Difference Moisture Index (NDMI) as a covariate in their
sun bear model, and they suggested that sun bears are highly adaptive to various habitat types and changes. In his review, Garshelis (2022) suggested that sun bears may care more about food availability than human interactions.

In Myanmar, Gaffi et al. (2020) reported sun bears in various environments, including primary forests, degraded forests, and bamboo breaks, demonstrating their flexibility in habitat use. Similarly, in Myanmar, sun bears occur in the primary evergreen forests of Htamanthi Wildlife Sanctuary (Htike & Naing, 2017), as well as in the mixed bamboo and evergreen forests of Rahine Yoma Elephant Range (Cremonesi et al., 2021). Despite these studies in Myanmar, the Myanmar Sun Bear Conservation Action Plan (2020) and Gaffi et al. (2020) suggest additional research is needed on sun bear occupancy and the effects of water sources, human settlements, and habitat types on the distribution of sun bears. Further, the effect of predators on sun bears is unknown. From a camera trap survey in Myanmar, Naing et al. (2020) reported an Indian leopard (*Panthera pardus*) carrying a sun bear cub by the throat. Bengal tigers and Indian leopards may have a significant effect on the distribution of sun bears where they co-occur.

Overall, sun bear studies indicate that the species utilize a wide variety of habitats and there is much variation in the factors affecting their distribution. However, given the various study approaches and variety of environmental and anthropogenic variables examined, it is difficult to identify the factors most affecting sun bear distribution. Undoubtedly, the factors affecting sun bear distribution may vary between study sites; however, standardizing the variables examined across studies could allow for more direct comparisons across sun bear range. Based on this literature review, we outline a suite of variables for researchers to consider using in future sun bear distribution studies (Table
1.3). We also encourage researchers to: 1) utilize camera traps rather than transect sampling, 2) fortify model construction by assembling different models and incorporate model averaging, 3) incorporate model testing, and 4) expand sun bear distribution studies to regions where there is little information, particularly in Myanmar and Thailand.

Conclusions

Based on an extensive literature review, the most common methods of collection of sun bear presence points are camera trapping and transact sampling. Camera traps are more reliable than other methods of quantifying or retrieving presence/absence data (Bisi et al., 2019). Our findings indicate that sun bear distribution studies that fortified model construction could be more insightful; for example, assembling different models and model averaging. Ensemble modeling is most often used in the context of making forecasting and helps explore the range of predictions given the different uncertainties (Zurell et al., 2020). Moreover, the model outputs are tested in only very few studies; for example, Wong & Linkie (2013) used an additional data set collected from 10 camera traps to test/validate the model prediction (see Fig. 1.1). Nazeri et al. (2012) also validated their model outputs in Peninsular Malaysia by comparing range maps created by IUCN experts and they found that their model prediction was within the range maps, but it was not statistically significant or seen the prediction performance statistics.

Many environmental variables are useful among the models, such as elevation, roads, and biogeographical information, but modelers should also consider ecological data such as tree species (for example, fruiting trees) and predator occurrence may also be informative. A good predictive model should account for the biological, topographic, and climatic aspects associated with species occurrence data (Abidin et al., 2019). In
many habitat models, the prediction of species’ habitats does not cover biotic factors such as species interactions, which could influence habitat preference (Nazeri et al., 2012; Guisan et al., 2002). For example, Naing et al. (2020) reported that an Indian leopard carrying a sun bear cub in Northern Myanmar, and (Fredriksson, 2005) also reported female sun bear in Borneo was swallowed by a reticulated python.

The IUCN Bear Specialist Group mapped the current range-wide distribution of sun bears in 2014 (Fig. 1.1), based on more than 2,000 presence points (Scotson et al. 2017a). However, the veracity of the probable range depicted by experts remains unclear. The Myanmar National Red list of threatened species (2020) also described the range map of sun bears based on more than 100 presence points (WCS, 2021). Even though Scotson et al. (2017a) had done a range-wide habitat map, their studies used only one variable (percent of tree cover) to predict sun bear presence in both insular and mainland Southeast Asia. As described above, sun bear distribution relates to tree cover and many other environmental variables.

Habitat modeling approaches (e.g., Maxent) produce maps of the potential distribution of bear status locally (Nazeri et al., 2012). However, those models are often generated with bear presence data from a small area and are extrapolated broadly. In addition, previous modeling studies may be weak because models’ inference was extrapolated beyond the temporal and/or spatial extent of the data. Large areas remain to be researched where sun bear status is uncertain, particularly in Myanmar and Thailand. Moreover, connectivity analysis for bear habitats, considering the habitat fragmentation variable, may provide a higher resolution of species distribution and population viability.
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Table 1.1: Frequency of modeling approaches, data gathering methods, and geographic scale used to assess Malayan sun bear (*Helarctos malayanus*) occurrence, as tabulated from a review of 33 peer-reviewed papers published between 2003 and 2022.

<table>
<thead>
<tr>
<th>Research topic</th>
<th>Approach</th>
<th>References using ≥1 approach&lt;sup&gt;a&lt;/sup&gt;</th>
</tr>
</thead>
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<tr>
<td></td>
<td></td>
<td>No.</td>
</tr>
<tr>
<td>Data collection methods</td>
<td>Camera Trap</td>
<td>19</td>
</tr>
<tr>
<td></td>
<td>Transect</td>
<td>14</td>
</tr>
<tr>
<td></td>
<td>Radio Collar</td>
<td>4</td>
</tr>
<tr>
<td></td>
<td>Barrel Traps</td>
<td>1</td>
</tr>
<tr>
<td></td>
<td>Interview</td>
<td>4</td>
</tr>
<tr>
<td></td>
<td>GPS (secondary presence points)</td>
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<tr>
<td></td>
<td>Literature Review</td>
<td>2</td>
</tr>
<tr>
<td></td>
<td>Genetic Sampling</td>
<td>1</td>
</tr>
<tr>
<td>Modeling approach</td>
<td>Niche Modeling</td>
<td>12</td>
</tr>
<tr>
<td></td>
<td>Occupancy Modeling</td>
<td>10</td>
</tr>
<tr>
<td></td>
<td>Basic Statistics</td>
<td>6</td>
</tr>
<tr>
<td></td>
<td>Home Range Analysis</td>
<td>3</td>
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<tr>
<td></td>
<td>Other (no model)</td>
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<tr>
<td>Scale</td>
<td>Local</td>
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<td></td>
<td>Regional</td>
<td>5</td>
</tr>
<tr>
<td></td>
<td>Other (at zoo)</td>
<td>1</td>
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</tbody>
</table>

<sup>a</sup> For Data collection methods, some references used more than one, thus percentages in that topic area add up to >100.
Table 1.2: Variables in peer-reviewed studies with varying modeling approaches and scale used to investigate Malayan sun bear (*Helarctos malayanus*) distribution and reported as having no significance or importance in influencing the species’ distribution.

<table>
<thead>
<tr>
<th>General variable type</th>
<th>Variable codes</th>
<th>Definition of variables</th>
<th>Modeling approach$^a$</th>
<th>Scale$^b$</th>
<th>No. of references</th>
</tr>
</thead>
<tbody>
<tr>
<td>Human disturbance</td>
<td>HuDen</td>
<td>Human density</td>
<td>NM/OM</td>
<td>L</td>
<td>3,2,12</td>
</tr>
<tr>
<td></td>
<td>HuFp</td>
<td>Human footprint</td>
<td>OM</td>
<td>L</td>
<td>2,3,4</td>
</tr>
<tr>
<td></td>
<td>HUNT</td>
<td>Hunting density</td>
<td>OM</td>
<td>L</td>
<td>1,5</td>
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<td></td>
<td>LandClr</td>
<td>Land clearance</td>
<td>OM</td>
<td>L</td>
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<tr>
<td></td>
<td>Access</td>
<td>Accessibility</td>
<td>OM</td>
<td>L</td>
<td>1,5</td>
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<td>Topographic</td>
<td>Slp</td>
<td>Slope</td>
<td>NM/OM/O</td>
<td>L/R</td>
<td>6,10</td>
</tr>
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<td></td>
<td>10,2,1,8,2,7</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Asp</td>
<td>Aspect</td>
<td>NM/OM</td>
<td>L/R</td>
<td>3,2,8,9</td>
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<tr>
<td></td>
<td>TRI</td>
<td>Terrain ruggedness index</td>
<td>NM</td>
<td>L</td>
<td>2,12</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>1,9,10</td>
</tr>
<tr>
<td>Climate</td>
<td>Bio18</td>
<td>Precipitation of warmest quarter</td>
<td>NM</td>
<td>L/R</td>
<td>2,1,9</td>
</tr>
<tr>
<td></td>
<td>Bio4</td>
<td>Seasonality temperature</td>
<td>NM/OM</td>
<td>L/R</td>
<td>2,8,9</td>
</tr>
<tr>
<td></td>
<td>Bio7</td>
<td>Temperature annual range</td>
<td>NM</td>
<td>R</td>
<td>2,9</td>
</tr>
<tr>
<td></td>
<td>Bio9</td>
<td>Mean temperature of driest quarter</td>
<td>NM</td>
<td>R</td>
<td>2,9</td>
</tr>
<tr>
<td></td>
<td>Bio3</td>
<td>Isotermality</td>
<td>NM</td>
<td>R</td>
<td>1,9</td>
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<td></td>
<td>Bio16</td>
<td>Precipitation of Wettest Quarter</td>
<td>NM</td>
<td>R</td>
<td>1,9</td>
</tr>
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<td>Bio17</td>
<td>Precipitation of driest quarter</td>
<td>NM</td>
<td>R</td>
<td>1,9</td>
</tr>
<tr>
<td>Vegetation</td>
<td>AF</td>
<td>Available forage</td>
<td>NM/OM</td>
<td>L</td>
<td>2,2</td>
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<tr>
<td></td>
<td>CH</td>
<td>Canopy height</td>
<td>NM</td>
<td>L</td>
<td>2,11,6</td>
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<tr>
<td></td>
<td>N.bigtree</td>
<td>No. of big trees</td>
<td>NM</td>
<td>L</td>
<td>2,7,6</td>
</tr>
<tr>
<td>Variable</td>
<td>Description</td>
<td>Model</td>
<td>Local/Regional</td>
<td>Site 1</td>
<td>Site 2</td>
</tr>
<tr>
<td>-----------</td>
<td>--------------------------------------------------</td>
<td>-------</td>
<td>----------------</td>
<td>--------</td>
<td>--------</td>
</tr>
<tr>
<td>CO</td>
<td>Canopy openness</td>
<td>NM</td>
<td>L</td>
<td>1</td>
<td>6</td>
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<tr>
<td>DBH</td>
<td>Mean diameter at breast height</td>
<td>NM</td>
<td>L</td>
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<td>6</td>
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<td>N.climber</td>
<td>No. of climbers</td>
<td>NM</td>
<td>L</td>
<td>1</td>
<td>6</td>
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<tr>
<td>N.rattans</td>
<td>No. of rattans</td>
<td>NM</td>
<td>L</td>
<td>1</td>
<td>6</td>
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<tr>
<td>N.saplings</td>
<td>No. of saplings</td>
<td>NM</td>
<td>L</td>
<td>1</td>
<td>6</td>
</tr>
<tr>
<td>NDVI</td>
<td>Normalized differential vegetation index</td>
<td>NM</td>
<td>R</td>
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<tr>
<td>Ucover</td>
<td>Understory cover</td>
<td>NM</td>
<td>L</td>
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<td>6</td>
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<tr>
<td>Vol.Deatree</td>
<td>Volume of dead trees</td>
<td>NM</td>
<td>L</td>
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<table>
<thead>
<tr>
<th>Other Ecological Variable</th>
<th>Description</th>
<th>Model</th>
<th>Local/Regional</th>
<th>Site 1</th>
<th>Site 2</th>
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<tbody>
<tr>
<td>CP</td>
<td>Competitor presence</td>
<td>NM/OM</td>
<td>L</td>
<td>2</td>
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<tr>
<td>D.hab</td>
<td>Distance to habitat patch edge</td>
<td>NM/O</td>
<td>L</td>
<td>2</td>
<td>11, 13</td>
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<tr>
<td>OSC</td>
<td>Other species of concern</td>
<td>NM/OM</td>
<td>L</td>
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<td>2</td>
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<td>PP</td>
<td>Predator presence</td>
<td>NM/OM</td>
<td>L</td>
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<td>2</td>
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<tr>
<td>ST</td>
<td>Soil texture</td>
<td>OM</td>
<td>L</td>
<td>1</td>
<td>8</td>
</tr>
<tr>
<td>N.intertrail</td>
<td>No. of intersecting trails</td>
<td>NM</td>
<td>L</td>
<td>1</td>
<td>7</td>
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</tbody>
</table>

*a NM = Niche Modeling, OM = Occupancy Modeling, O = other/no model. (BS = Basic Statistics, and HR = Home Range Analysis were not used with covariate in Malayan sun bear habitat studies. Therefore, those models were not included in this table.)

b L = Local, R = Regional.

Table 1.3: Variables in peer-reviewed studies with varying modeling approaches and scale used to investigate Malayan sun bear (*Halaartos malayanus*) distribution and reported as significant or important (Sig.) in influencing the species’ distribution.

<table>
<thead>
<tr>
<th>General variable type</th>
<th>Variable codes</th>
<th>Definition of variables</th>
<th>Modeling approach&lt;sup&gt;a&lt;/sup&gt;</th>
<th>Scale&lt;sup&gt;b&lt;/sup&gt;</th>
<th>No. of references</th>
<th>References&lt;sup&gt;c&lt;/sup&gt;</th>
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</thead>
<tbody>
<tr>
<td>Human disturbance</td>
<td>D.Sett</td>
<td>Distance to settlements</td>
<td>NM/OM/O</td>
<td>L</td>
<td>7</td>
<td>1 (14)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>R</td>
<td>3</td>
<td>1 (33)</td>
</tr>
<tr>
<td></td>
<td>HuAct</td>
<td>Human activities/disturbance</td>
<td>NM/OM</td>
<td>L</td>
<td>4</td>
<td>1 (25)</td>
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<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>R</td>
<td>1</td>
<td>0 (0)</td>
</tr>
<tr>
<td></td>
<td>D.Rd</td>
<td>Distance to roads</td>
<td>NM/OM/O</td>
<td>L</td>
<td>7</td>
<td>1 (14)</td>
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<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>R</td>
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<td>0 (0)</td>
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<tr>
<td></td>
<td>D.Rb</td>
<td>Distance to reserve boundary</td>
<td>NM</td>
<td>L</td>
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<td>1 (100)</td>
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<td></td>
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<tr>
<td>Total</td>
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<td></td>
<td></td>
<td>23</td>
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<td>Topographic</td>
<td>Elev</td>
<td>Elevation</td>
<td>NM/OM/O</td>
<td>L</td>
<td>12</td>
<td>7 (58)</td>
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<tr>
<td></td>
<td>2,10,14,16,17,18</td>
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<td>R</td>
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<td>0 (0)</td>
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<tr>
<td>Total</td>
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<td></td>
<td></td>
<td></td>
<td>14</td>
<td>7 (50)</td>
</tr>
<tr>
<td>Climate</td>
<td>Bio1</td>
<td>Annual temperature</td>
<td>NM/OM</td>
<td>L</td>
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<td>0 (0)</td>
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<tr>
<td>Variable</td>
<td>Description</td>
<td>Level</td>
<td>Left</td>
<td>Right</td>
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</tr>
<tr>
<td>Bio12</td>
<td>Annual precipitation</td>
<td>NM/OM</td>
<td>L</td>
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<td>Bio14</td>
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<td>NM/OM</td>
<td>L</td>
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<td></td>
<td></td>
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<tr>
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<td>Above ground carbon density</td>
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<tr>
<td>FCover</td>
<td>Forest, vegetation, or canopy cover</td>
<td>NM/OM/O</td>
<td>L</td>
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<td>L</td>
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<tr>
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<td>Enhanced vegetation index</td>
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<td>N (100%)</td>
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<td>Refugia (escape cover)</td>
<td>NM/OM</td>
<td>L</td>
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<td>NM</td>
<td>L</td>
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<td></td>
<td></td>
<td></td>
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*a NM = Niche Modeling, OM = Occupancy Modeling, O = other/no model. (BS = Basic Statistics, HR = Home Range Analysis, were not used with covariate in Malayan sun bear habitat studies. Therefore, those models were not included in this table.)*

*b L = Local, R = Regional.

Figure 1.1: Range map of Malayan sun bear (*Helarctos malayanus*); map modified from Scotson et al. 2017a) and locations of Malayan sun bear, distribution research (see Appendix S1).
Figure 1.2: A) Country/region-specific number of peer-reviewed publications and B) annual number of publications within the period of 2000 to 2023, that assessed distribution of Malayan sun bears, (*Helarctos malayanus*).
Figure 1.3: Survey seasons of Malayan sun bear, \textit{(Helarctos malayanus)} distribution studies as reported in 33 peer-reviewed publications. Most studies took place during wet seasons (Jun-Oct) vs. dry seasons (Dec-Apr).
CHAPTER 2

FACTORS INFLUENCING THE DISTRIBUTION OF MALAYAN SUN BEARS IN A PORTION OF THE HTAMANTHI WILDLIFE SANCTUARY IN NORTHERN MYANMAR

Abstract – Across its range, the Malayan sun bear (*Helarctos malayanus*) occurs in various habitat types ranging from dense natural forests to degraded forests, and also forages in agricultural land such as palm oil plantations. Yet, the species is classified as threatened with more than a 50% population decline within the last 30 years due to habitat loss, hunting, and other threats. In Myanmar, where the bear’s distribution is not well-known, we conducted three annual surveys using 120 camera-trap stations in a portion of the Htamanthi Wildlife Sanctuary (HWS) in northern Myanmar during 2016-17 to 2018-19 to identify factors influencing bear distribution. From a total effort of 15,315 trap nights, we obtained 47 independent photo events of sun bears at 16%, 13%, and 9% of the stations each year. We analyzed eight factors potentially influencing bear distribution and found that the top three ranked models for sun bear habitat use were a combination of elevation and slope, normalized difference vegetation index (NDVI) and slope, and distance to water and slope. The presence of tigers (*Panthera tigris*) in the area had a positive relation with mean sun bear occupancy. The combination of NDVI and slope was the most effective predictor for both occupancy and detection of sun bears. We also found that sun bear occupancy was negatively related to distance to water, elevation, slope, and NDVI, while detection probability was positively related to tiger detection. Our findings can inform future research and conservation efforts for sun bears by
highlighting the importance of specific habitat features and the potential role of tigers in affecting sun bear occupancy.

**Key words** – elevation, habitat, *Helarctos malayanus*, modeling, NDVI, *Panthera tigris*, slope, tiger, water.

**Introduction**

Understanding the factors affecting the distribution of a species plays a critical role in conservation, as it provides information on the presence and distribution of species, which is essential for land planning and management (Swan et al., 2021). Moreover, a better understanding of a species’ biology and its relationship with the environment can inform conservation efforts to protect and manage species and their habitats (Villero et al., 2017). The development of robust distribution models has revolutionized the way we approach conservation, as they are useful to predict and map the occurrence of species, both at local and global scales (Austin, 2007). This is particularly important for threatened species, as distribution models can help identify areas that are most important for their conservation and prioritize efforts to protect these areas (Jones, 2011).

The sun bear (*Helarctos malayanus*) is considered to be a keystone species in Southeast Asian forests, playing an important role in seed dispersal and maintaining the structure of the forest understory (Scotson et al., 2017). They are also culturally and economically valuable for local communities and are listed as a vulnerable species by the IUCN due to habitat loss and hunting pressures (Scotson et al., 2017). As a result, it is
crucial to understand their distribution and habitat requirements to inform conservation efforts and ensure their survival.

Malayan sun bears are one of the least studied and understood bear species in the world, despite being listed as vulnerable by the International Union for Conservation of Nature (IUCN) (Scotson et al., 2017). While studies on the distribution of sun bears have increased in recent years, providing important insights into their ecological needs and the threats they face, these studies have used a variety of methods and variables, making it difficult to identify factors most affecting sun bear distribution (see Chapter 1).

In Myanmar, tropical forests provide critical habitat for sun bears and other bear species (Htun, 2006). Yet, there is a scarcity of information on sun bear distribution and ecology (Gaffi et al., 2020), which makes it difficult to design effective conservation strategies for this species in Myanmar. As a result, there is an urgent need for additional research on sun bear populations and their habitat requirements in Myanmar.

The specific objectives of this study were to: (1) assess the current distribution of sun bears within a portion of the Htamanthi Wildlife Sanctuary (HWS) in northern Myanmar; (2) evaluate the importance of different anthropogenic and environmental variables in predicting sun bear presence; and (3) generate predictions of sun bear occurrence to guide conservation efforts within the larger HWS. To achieve these objectives, we used camera trap data collected from 120 camera trap stations within a 240-km² subunit of the HWS, in combination with other ecologically related environmental covariates. The results of this study provide new insights into the ecology of sun bears in Myanmar and inform conservation efforts aimed at preserving this important species in the region.
Methods

Study Area

The 2,151-km² Htamanthi Wildlife Sanctuary (HWS), located in the northwestern part of Myanmar and between the Chindwin and Uyu Rivers (Fig. 2.1), is one of the largest protected areas in Myanmar (Naing et al., 2019). In general, the sanctuary is covered by tropical evergreen forest, with mixed deciduous forest in the western part (Htike & Naing, 2017), and elevations range from 152-539 m. Four main streams flow from the eastern side to the western part of the sanctuary and can be navigated by boat through portions of the year. Three main footpaths/former logging roads cross the sanctuary, connecting 27 villages on the eastern and western sides of the wildlife sanctuary (Fig. 2.1). The sanctuary has a humid subtropical climate; there is a winter dry season (Nov-Apr when precipitation averages <1 cm and temperature averages 5°C, and a summer wet season (May-Oct) characterized by an average rainfall of 42 cm and an average temperature of 19°C (Forest Department & WCS, 2019).

We conducted this study on four adjacent 60-km² sample plots (total of 240 km²) in the northwest portion of the Sanctuary (see Plots A, B, C, and D; Fig. 1). We selected plots that were accessible by researchers and had habitat features known to be important for sun bears (e.g., topography, vegetation cover, water sources; Fig. 2.2). Earlier studies showed that home range data for sun bears in Malaysia measured by VHF telemetry were 6-21 km² (Wong et al., 2002) and 4-21 km² in Borneo (Meijaard E. et al., 2005); therefore, we ensured that our plots were large enough to potentially support 5-10 bears, a population necessary to assure higher data quality and accuracy. Plot A was close to human settlement and intervention (e.g., old gold mining and frequent trespassing by
local people) and, as a result, had degraded habitat, including old gold mining sites; plots B and C had no human settlement and intervention. Additionally, law enforcement was more intensive in the area adjacent to Plot C, presumably resulting in more wildlife compared to the other parts of the study area and sanctuary. Given these characteristics, we assumed that our four sampling plots were representative of almost the entire sanctuary. However, our plots differed from the larger sanctuary area with all four plots occurring in nearly flat areas (elevation ranging from 142 to 271 m above sea level (asl), while the sanctuary has a highland area ranging from 300 to 512 m asl in the southern part.

**Data collection**

We deployed camera traps (Acorn Camera, Model LTL-5210A with wide-angle Trail Camera) at a single location in each of the 120 (2-km²) grid cells in a plot. We chose the 2-km²-sized grid based on reported sun bear home range sizes reported by (Wong et al., 2004) (kernel home range is 6-20 km² with core area of 0.6-2 km²) Wong et al., 2004). Within each grid cell, we set up one single camera trap at selected location, but we avoided areas with cliffs, dense vegetation like bamboo forests, and deep and wide streams due to logistical challenges. Moreover, the camera traps were deployed as much as possible on natural animal trails, old logging roads, and dry stream beds where tracks and signs of sun bears were observed to maximize the detection probability. We used a single camera trap at each station, set at ~45-60 cm height from ground level. Camera sites were located, on average, 1.5 km apart (range = 1-2 km). Camera traps recorded 20 seconds videos and were active for 45 consecutive days. We chose this 45-day period because our preliminary surveys indicated that sun bears typically first appeared in front
of our cameras two to three weeks after we set cameras up, presumably because they are sensitive to human sign and disturbance in the area immediately around the camera trap. Each year we surveyed Plots A and B first and then moved cameras to Plots C and D.

The same survey was conducted for three consecutive years, using the same camera trap positions, during the dry seasons (Nov-Apr) of 2016-2017 through 2018-2019. We surveyed during dry seasons because of logistical challenges; in our study area, field work during the rainy seasons is difficult because transportation to and from the survey sites is problematic and because of health concerns for survey teams. Moreover, during the rainy season the probability of false triggers by camera traps is high due to the motion detection sensors.

**Data Analysis**

Species identification – We referred to previous studies and field guidebooks for sun bear identification. To confirm the species identification, we sought the assistance of local wildlife experts and members of the Myanmar Forest Department who were familiar with the wildlife species found in the region (Naing et al., 2019). We considered an image as an independent sun bear event when a single sun bear was captured within a 30-minute interval at the same camera trap station.

Model covariates – To analyze the habitat selection and preferences of sun bears, we collected various covariate data reported as important by other sun bear studies (see Chapter 1). Covariate data, including distances to patrol stations, roads, water bodies, and villages (Table 2.1), were generated using the Euclidean Distance tool in the Spatial Analyst extension of ArcMap 10.2. The Euclidean Distance tool calculates the straight-line distance between each cell and the nearest feature, in this case, from camera trap
stations to the patrol station, road, water body, and village, respectively. The raster files with a resolution of 30 x 30 m were produced and used in the analysis.

Moreover, we also utilized additional covariates such as elevation, Normalized Difference Vegetation Index (NDVI), and slope. The elevation raster was generated using the Digital Elevation Models (DEM) – Digital Elevation – Global 30 Arc-Second Elevation (GTOPO30) dataset, which was downloaded from USGS Earth Explorer (https://earthexplorer.usgs.gov/) on July 15, 2022. For the NDVI raster, we obtained the data from NASA Earth Data (https://search.earthdata.nasa.gov/search) on July 15, 2022. Careful consideration was given to the acquisition dates to ensure that the data accurately represented the survey period and had minimal cloud coverage. The slope raster was calculated using the DEM layer in the “slope” tool of ArcMap 10.5. To ensure that our raster data were accurate and relevant to our study area, we masked the elevation raster using a 1-km buffer of the HWS boundary and extracted by the sampling plots.

In addition to these covariates, we also included the co-occurrence of tigers or leopards (Panthera pardus) in our analysis. This information was obtained from our camera trap photo detection history. These variables, along with the sun bear presence data, were used in our analysis to better understand the distribution and habitat use of sun bears in the HWS. However, we did not consider annual temperature and annual precipitation in the analysis, as these climatic variables are more useful for regional scale studies (see Chapter 1).

Data modeling – In the data modeling process, we used single-season (dry) occupancy to analyze the habitat use of sun bears based on camera trap data. The R package “Unmarked” (Fiske & Chandler, 2011) was utilized to model the effects of
covariates (distance to villages, distance to water, distance to road, distance to patrol station, elevation, NDVI, and slope as site covariates, and tiger detection history as observational covariate) at all camera trap locations. To calculate sun bear detection history, we considered a full 45-day annual sampling period to be one sampling occasion. Because data were collected for three years, our detection history had three sampling occasions. Our occupancy analysis made the following assumptions (MacKenzie et al. 2002):

a) During the intensive small grid cell occupancy survey period, the site occupancy of species does not change, i.e., the site occupancy of targeted species during November 2016 and May 2009 is closed.

b) Detection probability is assumed to be constant unless the site covariates vary in the model.

c) Species observations at different sites are independent of each other and unbiased.

d) Throughout the survey season, we assume that the defined study area is closed that is the occupancy of species at the site level has not increased or decreased because of immigration, emigration, or local extinction.

e) Detection in each sampling unit of the site is independent and has no effect on the outcome of detection in other sampling units.

f) Response variables (species observation) are influenced by the predictor variables (i.e., environmental, and anthropogenic characteristics).

g) Among predictor variables, there is no multicollinearity such as land cover and distance to stream.
Standardizing the data is a crucial step in data analysis, helping to ensure that variables with different units of measurement are on the same scale. In our analysis, we standardized the data by converting all the variables to standard scores (mean = 0, standard deviation = 1) using the “scale” function in R. To examine potential multicollinearity among the predictor variables, pair plots and variance inflation factor (VIF) values were calculated (Zuur et al., 2007). Pair plots were used to visually assess the relationship between each pair of predictor variables and identify any strong linear relationships that could lead to multicollinearity (Gelman and Hill, 2007). According to the pair plots (Fig. 2.3), distance to water (d.water) and elevation (elev) had a significant Spearman's rank correlation coefficient of 0.612, as did distance to village (d.vlg) and distance to patrol station (d.patrol; 0.565). Therefore, we did not include those combinations of variables in the model fitting process. Furthermore, we also removed the combination of variables that resulted VIF values > 10. The final candidate models included the combination of elevation, NDVI, and distance to water with a slope variable (Table 2.2).

To ensure that the results of our analysis were interpretable and to minimize multicollinearity, we fit our models with no more than two covariates (Zurell et al., 2020). This approach is commonly used in species distribution modeling and ecological research, as it reduces the complexity of the models and allows for a more clear interpretation of the relationships between the predictor variables and the response variable (Guisan et al., 2002). The selection of the most important predictor variables was based on the Akaike Information Criterion (AIC) and model selection using stepwise regression (Fletcher & Fortin, 2018).
After model fitting and selection, we included the MacKenzie and Bailey goodness-of-fit test to assess the goodness of fit of our candidate occupancy models with 1,000 simulation samples test (MacKenzie & Bailey, 2004). This test compares the observed and predicted values of the detection/non-detection process to evaluate the accuracy of the model, and it is well-established in the literature and has been widely used in occupancy modeling studies (MacKenzie & Royle, 2005).

To project the habitat use of sun bears, we utilized the best six models selected based on their AIC values and used the “predict” function in R (Fiske & Chandler, 2011). The prediction was made using a raster stack of the 240-km² survey area within the sanctuary with a resolution of 30 x 30 m. The raster stack was created using the relevant covariates (distance to villages, distance to water, distance to road, distance to patrol station, elevation, NDVI, slope, and tiger detection history). The predictions from the best six models were then ensembled to produce an AIC-weighted sun bear occupancy prediction map. The use of AIC-weighted assembling approach provides a more robust and reliable prediction of habitat use compared to using a single model (Fletcher & Fortin, 2018). To make ensembles, we used the weighted average of AIC values to make ensemble models to compare hypotheses.

**Results**

During the 2016-2017 survey, we obtained 19 independent photographs of sun bears over 5,172 trap nights and, they were detected at 18 (16%) of 114 sites (each year, data were not obtained from some sites due to camera malfunction or theft). In 2017-2018 survey, we obtained 16 independent photographs of sun bears over 4,977 trap nights, and they were detected at 14 (13%) of 111 sites. During the 2018-2019 survey, we
obtained 12 independent photographs of sun bears over 5,166 trap nights, and they were detected at 9 (8%) of 115 sites (Fig. 2.4).

Combining the unmarked data frames for the three survey periods (2016-2019), there was a total 47 sun bear photos obtained at 35 different sites (29% of the 120 sites) during 15,315 trap nights (Appendix S1). For the combined three survey periods, sun bears were detected at 9 stations in Plot A, with only one occasion per station. In Plot B, sun bears were detected at 11 stations, with two occasions at one station. In Plot C, sun bears were detected 8 stations with two occasions at one station. In plot D, 7 stations detected sun bears with only one occasion per station (Fig. 2.5).

The top six ranked models for sun bear habitat use were a combination of elevation and slope, NDVI and slope, and distance to water and slope with and without tiger presence as observation covariates (Table 2.2). In each of these combinations, the model that included tiger detection history had a lower AIC value compared to the model without tiger detection history (Table 2.2). This suggests that the presence of tigers in the area had a positive effect on our obtaining photos of sun bears at stations. When the presence of tigers was included as a covariate, the mean occupancy was slightly higher compared to the models without tigers.

The model prediction showed that the mean occupancy of sun bears in the sample plots was highest in the (elevation + slope) model, both with and without the presence of tigers (Table 2.3). The mean occupancy with tigers was 0.975 (sd = 0.002), while the mean occupancy without tigers was 0.975 (sd = 0.009). The (NDVI + slope) model showed a mean occupancy of 0.926 (sd = 0.210) with the presence of tigers and 0.904 (sd = 0.200) without the presence of tigers. The (distance to water + slope) model showed a
mean occupancy of 0.951 (standard deviation = 0.203) with the presence of tigers and
0.909 (sd = 0.207) without the presence of tigers.

The mean detection probability of sun bear in the (NDVI + slope) model was
0.121 (sd = 0.030), while the mean detection probability in the (distance to water + slope)
model was 0.115 (sd = 0.032). On the other hand, the (elevation + slope) model had the
lowest mean detection probability of 0.112 (sd = 0.022). In contrast to the occupancy
models above, the mean detection probability of sun bear was slightly lower when tigers
were present compared to when they were not. For example, in the (NDVI + slope)
model, the mean detection probability of sun bear with tiger presence was 0.118 (sd =
0.0331), while the mean detection probability without tiger presence was 0.121 (sd =
0.0297). Similarly, in the (distance to water + slope) model, the mean detection
probability with tiger presence was 0.115 (sd = 0.0318), while without tiger presence was
0.120 (sd = 0.0305). The (elev + slope) model also showed a similar trend, with a mean
detection probability of 0.112 (sd = 0.0174) without tiger presence and 0.118 (sd = 0.022)
with tiger presence. However, when we consider the sd values, the effect of tigers
presence is not strong for sun bear detection. Further, the estimated beta coefficients (log-
odds scale) were positive for the "tiger" variable (0.9628 to 0.9809), suggesting that when
tigers are present, the probability of detecting sun bears becomes higher around 0.4 to 0.5
times.

The Mackenzie and Bailey goodness-of-fit test for the top models yielded a chi-
squared statistics of 21 to 30, with a bootstrap sample size of 1000. The p-values ranged
from 0.241 to 0.328 (if p-value is larger than 0.05, the model is not significantly lacking
fit [Mackenzie & Bailey, 2004]), and the c-hat values ranged from 0.97 to 1.2
(approximate to 1 means the variation of data are acceptable for a good model [MacKenzie & Bailey, 2004]). These results suggest that the models had a good fit to the observed data (Fig. 2.8).

**Discussion**

The top three ranked two-variable models for habitat use were a combination of elevation and slope, NDVI and slope, and distance to water and slope. In these top models, sun bear occupancy appeared to be negatively correlated with distance to water, elevation, slope, and NDVI. This suggests that sun bears are less likely to occupy habitats that are far from water sources, at higher elevations, steeper slopes, or have lower NDVI values. The inclusion of tiger detection history in each of these combinations resulted in a lower AIC (Akaike Information Criterion) value compared to the models without tiger detection history, indicating that the presence of tigers has a positive effect on sun bear habitat use. This sun bear-tiger relation was surprising because we expected an antagonistic relationship between the species. This highlights the importance odor future studies to consider the interaction between sun bears and tigers in assessing the habitat use of sun bears.

The proportion of sites where sun bears were detected (sun bear detection rate) was relatively low in our study area compared to other sun bear studies (Cremonesi et al., 2021; Guharajan et al., 2023; Tee et al., 2021), but similar to the detection rate reported for sun bears in the Rakhine Yoma Elephant Range (RYER) in western Myanmar (Cremonesi et al., 2021). Using camera traps and a similar sampling effort, Cremonesi et al., (2021) reported sun bear trapping rates of 0.7 to 1.0 versus 0.2 to 0.4 in our HWS study (Appendix S1). For sun bear detection probability, we calculated 0.11-0.12 in our
study versus 0.020-0.16 in the RYER study. However, the two studies used different survey periods: we used one year occasion while RYER used 14 day occasion period. Tee et al. (2021) reported that the sun bear capture rate in Tabin Wildlife Reserve, Borneo, Malaysia was 3.4 with detection at 57% of 83 sites, and Guharajan et al. (2023) reported that 33% of 1,151 camera trap stations detected sun bear in Sabah Malaysia vs. 8-15% of our sites in HWS.

Although anthropogenic variables were not variables in our top AIC ranked models for habitat use of sun bears in HWS, anthropogenic variables were reported important in other studies in both mainland Southeast and insular Asia (Guharajan et al., 2018; Tee et al., 2021). Wong et al. (2013) reported that roads increased habitat fragmentation and accessibility to sun bear habitat in Sumatra, and that the presence of sun bears was strictly related to the distance to roads. Similarly, Wong and Linkie (2013) reported that the presence of villages around sun bear habitat can significantly reduce habitat suitability for bears. However, HWS has been completely protected and well-managed by the forest department of the Myanmar government since 2013, although it was designated as a timber extraction area before 2013 (Htike & Naing, 2017). As a result, there were almost no poaching activities in the nearby villages and there was no use after timber extraction was halted in 2013, and we did not find any impact of villages on sun bear habitat use. So, large mammals, including the sun bears, may use these old logging roads for movement. Moreover, when we included human covariates in our models, the VIF values were high (>10), indicating less precision in the models. This might be because the sampling plots were away from villages, and patrol stations and
camera trap stations were too distant (> 4 km) from villages for anthropogenic variables to affect sun bear distribution. thereby

Sun bear occupancy was positively related to elevation in RYER (Cremonesi et al., 2021). Similarly, sun bear presence in Borneo also increased with higher elevations (Tee et al., 2021). In additions, Augeri, 2005; Guharajan et al., 2018; Nazeri et al., 2014; W.-M. Wong et al., 2013; and Wong & Linkie, 2013 reported that elevation is an important geographical covariates for the sun bear habitat occupancy. In contrast, sun bears were negatively related to higher elevations in our study, perhaps because in HWS lower elevations may provide more food and water availability.

No other sun bear studies considered slope as an important covariate for sun bear habitat studies in both local and region-wide scale studies (Chapter 1). But, in HWS, slope contributed to all our top models, and thus could be important when combined with other covariates such as elevation, NDVI, and distance to water. Sun bears in HWS do not prefer the steep slopes with higher elevation, lower NDVI, and sites farther away from water sources.

Our findings agreed with Scotson et al. (2017b) who reported that sun bear probability of occupancy is positively correlated with the percent of tree cover; the more tree cover, the higher NDVI value. Steinmetz et al. (2013) reported that forest types, such as bamboo forest, secondary forest, and primary forest determine food availability and affect sun bear distribution. Gouda et al. (2019) reported that sun bears were found to use both disturbed (lower NDVI) and undisturbed habitats (higher NDVI) to a varying extent in their study area, the Dampa Tiger Reserve, Mizoram, India. Nevertheless, in the HWS, we found that sun bears appeared to avoid areas with lower NDVI value.
We also tried to include the tiger information in our models. We assumed that tigers and leopards are potential predators of sun bears (Kawanishi & Sunquist, 2004; Naing et al., 2020). We could not, however, include leopards as a variable in our models because no leopards were recorded in our sampling plots. We found that the detection probability of sun bears is positively related to tiger detection, i.e., sun bears are more likely to be detected in areas where tigers are also present. Sun bear occupancy being positively related to tiger detection can have multiple ecological reasons. One is that the presence of tigers may create a deterrent effect on other potential predators such as leopards (Naing et al., 2020), leading to an increase in sun bear occupancy in those areas. Another possibility is that areas where tigers are present may be high quality habitats for sun bears, as well as for other wildlife species, and sun bears may be more likely to be detected in these areas as a result of their higher food abundance and habitat suitability.

Our findings agreed with the report of Widodo et al., (2022) who reported that sun bears did not avoid tigers in Riau Province of central Sumatra. However, these are just hypotheses, and more research is needed to establish the extent of interaction between sun bears and tigers, as well as other potential sun bear predators. It is also important to note that this relationship may vary in different habitats and regions and may be influenced by multiple biotic and abiotic factors.

Regarding modeling challenges, according to the coefficient tables of our top-ranked models, we noticed no significant relationship between the sun bear habitat use prediction and covariates. Our sampling locations and the wildlife sanctuary do not have a significant deviation in environmental conditions (Fig. 2.2). Elevation, NDVI values and canopy coverage varied little across our study plots. While there were large
differences in the distance-related parameters (distance to roads, patrol stations, villages, and water) in our study, none of these variable were important for estimating habitat use by sun bears, probably because the minimum distance from the patrol stations (4.1 km) and villages (4.2 km). In addition, all of our sampling plots occurred inside the completely protected wildlife sanctuary. Therefore, it was not surprising that anthropogenic effects were low in our study.

Our ability to model only one season was a limitation of our study, thus potentially affecting the accuracy of our model predictions. Bear species in general are mobile and opportunistic feeders who move their foraging habitat based on the seasonal distribution of food (Ghimire & Thapa, 2015). Thus, the seasonality and variety of habitats surveyed can influence detectability (Jones, 2011). Additionally, the location of our camera traps on trails, ridgelines, and roadsides where slopes are low probably influenced the low significance of the slope variable in our model results.

Due to the nature of rare species, most sun bear studies resulted low detection probability of ~0.1. For rare species, more camera trap stations in place for longer time periods might produce more precise estimates of occupancy (Mackenzie & Royle, 2005). However, our study with 120 stations over 3 annual occasions still produced low detection of sun bears (~0.1) with unstable occupancy estimates (high standard deviation). For HWS, we suggest that future sun bear studies should cover more area (>240 km²), perhaps with a higher density of stations, and carefully choose cameras locations to better represent the different habitat types so as to obtain a more accurate estimate of occupancy by maximizing the detection probability. Moreover, it would be
more accurate and informative if the future studies could cover the different seasons including both wet and dry seasons.

Based on our results, sun bears seem to prefer habitats that are closer to water sources, at lower elevations, on gentler slopes, with higher NDVI values. These habitats likely provide sun bears with resources such as food and water and may also offer some protection from predators.

Conclusions

The habitat use of sun bears in HWS mainly depends on environmental covariates such as canopy coverage, distance to water, elevation slope, and NDVI values rather than anthropogenic impacts. Sun bears usually avoid hilly areas, where NDVI value is lower due to different vegetation structure in the HWS. Although sun bears usually avoid roads in other studies (Tee et al., 2021), they did not respond to the roads in our HWS study.

During our survey, 41 photographs over 15,315 trap nights probably represent a reasonable trapping rate for sun bears, a rare, shy species which is difficult to detect in HWS. However, we found a relatively high occupancy rates with lower detection probability. Mackenzie & Royle (2005) recommend increasing the number of replicates for common species and increasing the number of sites for rare species. Thus, we recommend that that future sun bear studies in HWS increase sampling effort over a wider range of habitats. Additionally, future sun bear research needs to be conducted in all seasons throughout the years to fully understand the habitat uses, including the habitat shift of sun bears in tiger conservation areas. More research is also needed on the ecological factors that influence the co-occurrence of sun bears and tigers, such as resource availability, habitat quality, and predation risk. Additionally, it would be
valuable to understand how sun bear behavior and demographics may vary in areas where tigers are present versus areas where they are not.

We also recommend that future research expand the geographical range of the study to include additional areas with different biotic and abiotic conditions in HWS. This would provide a more comprehensive understanding of the habitat use and occupancy patterns of sun bears and help to identify the key ecological drivers that shape their distribution. Finally, it would be worthwhile to consider incorporating the effects of anthropogenic factors, such as deforestation, hunting, and habitat degradation, into future studies. This would help inform the potential impacts of human activities on sun bear populations and inform conservation efforts aimed at mitigating these effects.

References


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Table 2.1: Variables used to estimate the habitat use of sun bear in the Htamanthi Wildlife Sanctuary, Myanmar, with categories, abbreviation, names, units, and the data sources.

<table>
<thead>
<tr>
<th>Variable categories</th>
<th>Abbreviation</th>
<th>Names</th>
<th>Range</th>
<th>Unit</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Vegetation</td>
<td>NDVI</td>
<td>Normalized Difference Vegetation Index</td>
<td>-0.37 - 0.821</td>
<td>index</td>
<td><a href="https://www.earthengine.app/">https://www.earthengine.app/</a></td>
</tr>
<tr>
<td>Human</td>
<td>d.patrold</td>
<td>Distance to the nearest patrol station of the wildlife sanctuary</td>
<td>4,080 - 17,377</td>
<td>m</td>
<td>WCS (Wildlife Conservation Society)</td>
</tr>
<tr>
<td></td>
<td>d.rd</td>
<td>Distance to the nearest roads (old-logging roads) inside and around the wildlife sanctuary</td>
<td>0 – 7520</td>
<td>m</td>
<td>WCS (Wildlife Conservation Society)</td>
</tr>
<tr>
<td></td>
<td>d.vlg</td>
<td>Distance to the nearest villages around the wildlife sanctuary</td>
<td>4,230 – 20,821</td>
<td>m</td>
<td>WCS (Wildlife Conservation Society)</td>
</tr>
<tr>
<td>Topographic</td>
<td>elev</td>
<td>Elevation in the wildlife sanctuary</td>
<td>142 - 271</td>
<td>m asl</td>
<td>Shuttle Radar Topography Mission (SRTM) Digital Elevation Models (DEM), 30*30 m resolution by USGS</td>
</tr>
<tr>
<td>Variable</td>
<td>Description</td>
<td>Range</td>
<td>Type</td>
<td>Source</td>
<td></td>
</tr>
<tr>
<td>----------</td>
<td>------------------------------------------------------------------------------</td>
<td>---------</td>
<td>--------</td>
<td>------------------------------------------------------------------------</td>
<td></td>
</tr>
<tr>
<td>slp</td>
<td>Slope in the wildlife sanctuary</td>
<td>0 – 36.5</td>
<td>Degree</td>
<td>Shuttle Radar Topography Mission (SRTM) Digital Elevation Models (DEM), 30*30 m resolution by USGS</td>
<td></td>
</tr>
<tr>
<td>Ecological tiger</td>
<td>The presence absence data of tiger (<em>Panthera tigris</em>) in the sampling area</td>
<td>0 - 1</td>
<td>Binary</td>
<td>Camera trap data from the sampling plots</td>
<td></td>
</tr>
<tr>
<td>d.water</td>
<td>Distance to the nearest water bodies such as rivers, streams, and lakes inside the wildlife sanctuary</td>
<td>0 – 3,995 m</td>
<td></td>
<td>WCS (Wildlife Conservation Society)</td>
<td></td>
</tr>
</tbody>
</table>
Table 2.2: Top-ranked and null single-season occupancy models explain sun bear (*Helarctos malayanus*) habitat use (pi) and detection probability (p) in the Htamanthi Wildlife Sanctuary, Myanmar, with the number of parameters (k), Akaike's information criterion (AIC), change in AIC (delta), and Akaike weight (AICwt).

<table>
<thead>
<tr>
<th>Model Description</th>
<th>k</th>
<th>AIC</th>
<th>delta</th>
<th>AICwt</th>
<th>cumltvWt</th>
</tr>
</thead>
<tbody>
<tr>
<td>p(.)pi(.)</td>
<td>2</td>
<td>237.72</td>
<td>0</td>
<td>0.363</td>
<td>0.36</td>
</tr>
<tr>
<td>p(tiger)pi(elev+slp)</td>
<td>5</td>
<td>239.64</td>
<td>1.92</td>
<td>0.139</td>
<td>0.5</td>
</tr>
<tr>
<td>p(.)pi(elev+slp)</td>
<td>4</td>
<td>239.85</td>
<td>2.13</td>
<td>0.125</td>
<td>0.63</td>
</tr>
<tr>
<td>p(tiger)pi(ndvi+slp)</td>
<td>5</td>
<td>240.1</td>
<td>2.37</td>
<td>0.111</td>
<td>0.74</td>
</tr>
<tr>
<td>p(.)pi(ndvi+slp)</td>
<td>4</td>
<td>240.27</td>
<td>2.54</td>
<td>0.102</td>
<td>0.84</td>
</tr>
<tr>
<td>p(tiger)pi(d.water+slp)</td>
<td>5</td>
<td>240.65</td>
<td>2.93</td>
<td>0.084</td>
<td>0.92</td>
</tr>
<tr>
<td>p(.)pi(d.water+slp)</td>
<td>4</td>
<td>240.85</td>
<td>3.12</td>
<td>0.076</td>
<td>1</td>
</tr>
</tbody>
</table>

*d.water* distance to nearest water bodies such as rivers and streams, *elev* Elevation at camera trap location, *ndvi* Normalized vegetation index, *slp* slope at camera trap location, *d.vlg* distance to villages, *d.rd* distance to nearest road, *tiger* tiger presence-absence at camera trap locations.
Table 2.3: Mean and standard deviation (SD) of occupancy estimates and detection probability of each model explain sun bear in the Htamanthi Wildlife Sanctuary, Myanmar, generated by each model.

<table>
<thead>
<tr>
<th></th>
<th>Occupancy</th>
<th>Detection</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Mean</td>
<td>SD</td>
</tr>
<tr>
<td>$p(tiger)pi(elev_{slp})$</td>
<td>0.9262</td>
<td>0.2101</td>
</tr>
<tr>
<td>$p(.)pi(elev+slp)$</td>
<td>0.9747</td>
<td>0.0095</td>
</tr>
<tr>
<td>$p(tiger)pi(ndvi+slp)$</td>
<td>0.9262</td>
<td>0.2101</td>
</tr>
<tr>
<td>$p(.)pi(ndvi+slp)$</td>
<td>0.9037</td>
<td>0.2004</td>
</tr>
<tr>
<td>$p(tiger)pi(d.water+slp)$</td>
<td>0.9510</td>
<td>0.2026</td>
</tr>
<tr>
<td>$p(.)pi(d.water+slp)$</td>
<td>0.9093</td>
<td>0.2074</td>
</tr>
</tbody>
</table>

$d_{water}$ distance to nearest water bodies such as rivers and streams, $elev$ Elevation at camera trap location, $ndvi$ Normalized vegetation index, $slp$ slope at camera trap location, $d_{vlg}$ distance to villages, $d_{rd}$ distance to nearest road, $tiger$ tiger presence-absence at camera trap locations.
Figure 2.1: Location of the 2,151-km² Htamanthi Wildlife Sanctuary and the 240-km² sampling plots in northwestern Myanmar.
Figure 2.2: Variables used to estimate the habitat use of sun bears in corresponding sampling plots of the Htamanthi Wildlife Sanctuary.
Figure 2.3: Plot showing the correlation between the covariates used in model fitting process for sun bears in the Htamanthi Wildlife Sanctuary, Myanmar. Corr values represent the Pearson’s correlation coefficient. The black thin lines represent the distribution pattern of covariates. $d_{water}$ distance to nearest water bodies such as rivers and streams, $elev$ Elevation at camera trap location, $ndvi$ Normalized vegetation index, $slp$ Slope at camera trap location, $d_{vlg}$ distance to villages, $d_{rd}$ distance to nearest road
Figure 2.4: Detection history of sun bear and tiger during a three-year camera survey in the Htamanthi Wildlife Sanctuary, Myanmar. Red small dots represent no data due to camera trap damage, white medium dots represent no detection of both species, and large green dot represent the detection of target species.
Figure 2.5: Location of camera traps, and summary of tiger and sun bear detection during a three-year camera survey in the Htamanthi Wildlife Sanctuary, Myanmar. A) sampling grid and locations of camera trap, B) detection of sun bear, C) detection of tigers, and D) stations that detected both species.
Figure 2.6: Map showing the model prediction of sun bear occupancy probability in the Htamanthi Wildlife Sanctuary, Myanmar. The gradient color represents the probability – the bright yellowish green represents zero and dark green represents the 100% sun bear occupancy. A) Elevation + Slope, B) NDVI + Slope, C) Distance to water + Slope, D) Model ensembles with sampling plots, and E) Model ensembles based on AIC weight.
Figure 2.7: Response of sun bear habitat use to covariates a) distance to nearest water bodies, b) elevation, c) NDVI, and d) slope.
Figure 2.8: The results of MacKenzie and Bailey goodness-of-fit for single-season occupancy models for sun bears during a three-year camera survey in the Htamanthi Wildlife Sanctuary, Myanmar. This approach simulates detection history sets, assuming the detection model fits appropriately, and then compares the binomial distribution of simulated data to the real data. If p value is greater than 0.05, it means the model is not
significant lack of fit. The red dotted lines are observed real data, and the histogram represent the distribution of simulated data.
CHAPTER 3

TESTING THE PREDICTION PERFORMANCE OF SINGLE-SEASON OCCUPANCY MODELS: A CASE STUDY OF MALAYAN SUN BEARS IN MYANMAR

Abstract – Testing the accuracy of occupancy models is commonly based on pseudo absence points and simulated presence points rather than on independent data. Thus, in this study, we tested the prediction performance of the single-season occupancy model for Malayan sun bears (*Helarctos malayanus*) using independent camera trap data from a larger area in northwest Myanmar. We used camera trap survey data to develop habitat use models for sun bears from an intensive study area (240 km$^2$) using an unmarked occupancy R package. We then applied these habitats models to assess the accuracy of our models using independent camera trap data for the entire Htamanthi Wildlife Sanctuary (HWS) (2,151 km$^2$). We tested the prediction performance of the top six models in the PresenceAbsence Package and calculated the AUC (Area under receiver curved), TSS (True skill statistics), and Kappa scores. The AUC score ranged from 0.5 to 0.6, while the TSS score ranges from -0.001 to 0.28. None of the top six models’ predictions agreed well with the sanctuary-wide survey data. The discrepancies may be due to the limited sample size, the temporal scale of the prediction, and the presence of other ecological factors (e.g., predators, competitors, or food availability) not accounted for in the habitat use models. The AIC-weighted ensemble outputs were more reliable than the predictions from the single models. To improve the prediction performance of occupancy models, we recommend that future sun bear surveys increase the number and size of sampling effort and include ecological covariates such as potential predators, as
much as possible; to better accommodate for the seasonal and long-term habitat use patterns of sun bears.

**Keywords:** Occupancy model prediction, camera trap survey, sun bear habitat, AUC score, ensemble model

**Introduction**

Species distribution models (SDMs) and occupancy models are increasingly used in wildlife conservation and management (Gould et al., 2019) in combination with model evaluation and model selection procedures (i.e. Akaike's Information Criterion (Fletcher & Fortin, 2018). However, model selection alone is insufficient to investigate a model's predictive performance. Moreover, despite the common use of occupancy models and SDMs, studies often fail to assess model fit and the predictive performance of models (Gould et al., 2019). Few studies have used spatially-independent data to validate habitat use relationships (Babu et al., 2015), largely because obtaining independent validation data is logistically difficult.

Malayan sun bears (*Helarctos malayanus*) are highly mobile species that possess adaptative behaviors to different environments (Scotson et al., 2017) and, for example, can be found in both undisturbed natural forests as well as palm oil plantations (Hwang et al., 2021). Additionally, their seasonal behaviors and daily activity patterns are dynamic, based on availability of food and shelter (Guharajan et al., 2018). Consequently, studying and modeling the habitat use of sun bears is challenging (Chapter 1), and there are no universal distribution or habitat use models of sun bears in their natural range. This may be due to the highly variable biogeographic features of study areas where sun bears have
been studied but may also result from poorly predictive models (often a result of limited sample size) that are not well-tested.

Our study objective was to test the predictive performance of a sun bear distribution model we derived from intensive field studies in a portion of Htamanthi Wildlife Sanctuary (HWS), Myanmar (Chapter 2). In this paper, we applied our sun bear model predictions to another independent data set obtained from a concurrent, more widespread study within HWS that overlapped our sun bear study area. In the analysis, we removed data from the overlapped area between the intensive survey area and independent data set. From this comparison, we assessed the predictive performance of our sun bear distribution models, and the effects of sampling effort and study area differences on model performance.

Methods

**Study Areas**

The 2,151-km² Htamanthi Wildlife Sanctuary (HWS), located in the northwestern part of Myanmar between the Chindwin and Uyu Rivers (Fig. 3.1), is one of the largest protected areas in the country (Naing et al., 2019). The area is covered by tropical evergreen forest, with mixed deciduous forest in the western part of the sanctuary (Htike & Naing, 2017), and elevation ranges from 152-539 m. Four main streams flow from east to west across the sanctuary and can be navigated by boat through portions of the year. Three main footpaths/former logging roads cross the sanctuary and connect 27 villages on the eastern and western sides (Fig. 3.1). The sanctuary has a humid subtropical climate with a winter dry season (Nov-Apr) with an average rainfall of <1 cm and an average
temperature of 15°C, and a summer wet season (May-Oct) with an average rainfall of 42 cm and an average temperature of 19°C (Forest Departement & WCS, 2019).

We had two different study areas within HWS. We used data from our 3-year intensive camera study to model sun bear distribution within a 240 km² study area in the northwest portion of HWS (Chapter 2). We selected this study area based on topography, vegetation cover, water sources, and accessibility for long-term survey and monitoring (Fig. 1), and each of the four plots was large enough to potentially support 5-10 bears (based on home range estimates in the literature). The other study area encompassed the entire 2,151-km² HWS (Fig. 3.2). Although the intensive survey (240-km²) overlapped with some areas of entire HWS (2151-km²), we removed the camera trap stations that overlapped, to ensure independence. However, there were a couple of differences between our smaller sampling area and the larger sanctuary. First, our cameras in the smaller study area were located in mostly low elevation areas, ranging from 152 to 300 m, but elevations in other areas of HWS range from 300 to 512 m in the southern part. Secondly, more intensive law enforcement efforts typically occur in the central part of HWS compared to the smaller survey area that may affect wildlife abundance.

Camera trapping

In our intensive survey area (240 km²), we collected sun bear data for three consecutive years, using the same camera trap positions, during the dry seasons (Nov-Apr) of 2016-2017 through 2018-2019. We deployed single camera trap in each of the 120 (2-km²) grid cells in 4 different plots (Fig. 3.1), but avoided areas with cliffs, dense vegetation like bamboo forests, and deep and wide streams due to logistical challenges. Moreover, to the extent possible, we deployed camera traps on natural animal trails, old
logging roads, and dry stream beds where tracks and signs of sun bears were observed to
maximize detection probability. We mounted the single camera trap at each station at
~45-60 cm height from ground level. Camera sites were located, on average, 1.5 km apart
(range = 1-2 km). Camera traps recorded videos and were active for 45 consecutive days,
and each year we surveyed Plots A and B first and then moved cameras to Plots C and D.

We used camera trap data from the sanctuary-wide survey to test our model
results developed from the intensive survey above. The larger sanctuary survey occurred
simultaneously in the third year (2018-2019) of our intensive survey and covered the
entire 2,151-km² HWS (Fig. 2). In this larger HWS survey, we set 282 camera traps
during December 2018-March 2019 in 141 grid cells (16 km² grid cells) (Fig. 3.2); tigers
(Panthera tigris) were the primary target for this survey. The minimum distance between
camera traps in adjacent grids was at least 4 km. Two paired cameras were set in each
grid with a single camera on each side of the trail with at least 7 m between the left and
right cameras. The paired cameras were used at each site to aid in identifying individual
tigers by using unique stripe patterns on each animal's side (Karanth & Nichols, 1998).
The camera traps were set 45–50 cm above ground level in flat terrain, with camera
height adjusted on sloping terrain. The camera traps were deployed along natural animal
trails near waterholes or salt licks, streams, and old logging roads, assuming these are the
main travel routes used by many large mammal species. There were two survey periods
during the study, each lasting 45 days, from December 2018-March 2019. All camera
traps operated for 24 hours per day using photo capture mode with normal motion
sensitivity, 5 MP resolution, and a 15-second delay between continuously recorded
photographs.
**Data analyses**

For the intensive camera trap survey, we calculated the number of independent events of the sun bear occurrence for each camera trap station. Then, we analyzed the habitat use by examining the influencing factors on the distribution of sun bears in the "unmarked" package (Fiske & Chandler, 2011). The best-supported models were identified based on Akaike's information criterion (AIC) and model weight (Zurell et al., 2020). The goodness of model fit was accessed through parametric bootstrap goodness of fit tests using the model sum of squared errors (MacKenzie & Bailey, 2004; Chapter 2).

For the projection of habitat use of sun bears for our intensive study area, we used the "predict" functions along with the raster stack (30 x 30 m resolution) of covariates. After predicting each best six models, we ensembled the prediction to produce AIC weighted sun bear occupancy prediction map (Dormann et al., 2018).

After generating species distribution predictions based on intensive sampling plots, we tested the models using sanctuary-wide camera trap data. Since the sanctuary-wide survey included some cameras of that year’s intensive sun bear sampling plots, we removed the data from those cameras (n=12) when applying our model to the larger area. We used the parameter structure for occupancy from the top six models to predict the distribution of sun bears for the entire wildlife sanctuary excluding the four study plots used for developing the habitat models from our intensive sampling area. Next, we validated these projected occupancy prediction raster layers by comparing them to observed habitat use based on the species presence-absence points obtained from the sanctuary-wide tiger survey data.
We used the PresenceAbsence package (Freeman & Moisen, 2008) to assess model validation based on the prospective sampling dataset from the 2018-19 sanctuary-wide tiger camera trap data. For model validation, we calculated Area under the Receiver Operating Characteristics (ROC) Curve (AUC) (Lawson et al., 2014). We also calculated four binary metrics taken from the confusion matrix: sensitivity, specificity, kappa, and the true skill statistic (TSS). To maximize the sum of specificity and sensitivity, we selected optimal thresholds of 3 (Liu et al., 2016). We then created the model ensembles with the predicted maps of each top 6 models. A weighted average of model predictions was based on AUC for each model evaluation metric (Marmion et al., 2009).

Results

**Sun bear detection**

In the intensive survey (Malayan sun bear survey), we obtained 47 independent photos of sun bears at 16%, 13%, and 9% of the stations each year (2016-17 to 2018-19), respectively, with a total effort of 15,315 trap nights. The sanctuary-wide tiger survey (2018-2019) with a total of 11,768 trap nights yielded 39 independent photos of sun bears at 18 stations of 104 camera traps (17%).

**Model validation**

Overall, our model’s projection (top 6 and combined) predicted 87-100 stations in the larger sanctuary-wide survey where we expected bear photos should occur, and 4-17 stations where we predicted that no bear photos would be recorded (Table 3.1). Yet, we obtained photos at only 15-17 of the sites where photos were predicted (16-18%), and we obtained bear photos at 1-3 sites where no photos were predicted (9-25%).
When we evaluated the top six models individually, the top model (slope + elevation) with tiger presence and the second model without tiger presence had the highest AUC value (~0.62). However, AUC values for the other four models were 5.3 to 5.5 (i.e., random prediction). In contrast, the kappa threshold (threshold) indicated that the other four models, excluding (elevation + slope) with and without tigers, were in best agreement with the model prediction and tested data. The log-likelihood (ll) was the highest (-318.63) for the (distance to water + slope) model, indicating the best fit of the model to the data. The maximum specificity of 0.6961 was produced by the (NDVI + slope) model, indicating this model had a low rate of false positives, and that it should correctly identify negative samples with high accuracy compared to other models. Kappa values ranged from -0.0029 to 0.0712, also suggesting all model projections were random predictions. (Table 3.2). True scale statistics (TSS) of models 1 to 6 were small, ranging from -0.0098, -0.0098, 0.1176, 0.1078, 0.0490, and 0.0784, respectively, indicating that all models are not strong predictors of sun bear habitat use. Overall, the model (elevation + slope) with and without a tiger appeared to be the best model as measured by AUC and the sensitivity indicators. Yet, other indicators suggested that other models were a better fit.

The final ensemble map (Fig. 3.3) derived from the AUC-weighted prediction of our top six models produced different occupancy probabilities over the same area.

Discussion

Our intensive study identified six top models with multiple factors affecting the presence of sun bears, including elevation, slope, NDVI, proximity to water, and the presence of tigers (Chapter 2). We confirmed the accuracy of these models through
parametric bootstrap tests using the model's sum of squared errors that indicated the models were robust and fit the data for our intensive survey area. However, when we tested them with the sanctuary-wide camera trap survey using the model validation statistics, we obtained a relatively low fit for the models.

It is important to note that the AUC score of 0.62 for the first and second models was not a strong indicator of predictive accuracy. Similarly, the low kappa values of the other models indicated that the agreement between the prediction and the actual data was poor and may not be reliable. Yet, the high log-likelihood value for the model (distance to water + slope) indicated that it was a good fit for the data. Similarly, the high specificity of 0.6961 for the model (NDVI + slope) suggested that this model can accurately identify negative samples. Similarly, we also evaluated the models’ ability to identify positive cases, producing a 0.92 sensitivity value for the model (distance to water + slope) and the model (elevation + slope), meaning that the model correctly identifies 92% of the positive cases. The ensemble model prediction provided an average, perhaps minimizing both the over-estimation and under-estimation of the occupancy probability for sun bears (Fig. 3.3).

This poor performance of our models may be due to a variety of factors. First, AUC values can vary depending on the spatial extent considered (Fletcher & Fortin, 2018). In our case, our training data set came from a relatively small area (240 km²), and when we tested the same models across the larger area (2,151 km²), the average AUC value was substantially lower (~0.62). Yet, Fletcher & Fortin (2018) report that application of models across larger spatial extents typically result in higher AUC values, which is the opposite of what we found in our study. The reasons for this discrepancy are
unknown. Further, the low detection probability of sun bears (~0.1) in our intensive study (Chapter 2) in HWS was a limiting factor in determining the accuracy of the habitat use predictions. MacKenzie & Bailey (2004) reported that low detection probabilities can lead to biased estimates of a species’ true distribution and abundance, affecting the accuracy of habitat use predictions. Additionally, the low detection probability of sun bears in the sanctuary-wide camera trap survey may have contributed to inaccuracies in the habitat use predictions (Guillera-Arroita et al., 2014; MacKenzie & Bailey, 2004; Zurell et al., 2020).

TSS scores of all our models were around 0 (-0.0098 to 0.1176), suggesting there was little agreement between the model predictions and observed values. We believe these TSS scores are a better measure of model performance than the AUC scores, considering that only 17 of 104 camera stations yielded accurate occupancy records for sun bears. Similarly, Shabani et al., (2018) concluded that TSS scores are a more intuitive method for measuring the performance of species distribution models compared with the area under the ROC curve (AUC), sensitivity, and specificity indices.

The modeling method used can also affect species distribution predictions (Shabani et al., 2018b). For example, Guillera-Arroita et al., (2014) reported that hierarchical occupancy models can lead to an estimate nearly equal to one when the sample size (number of detection) is small. Considering that our sun bear occupancy estimates were close to 1 for > 90% of our stations in HWS, we suspect that our small detection (one or two detection per station) for our intensive study plots may have been too small to use occupancy modeling.
Other factors not accounted for in our study, such as food availability, mating behaviors, and human disturbance, may also affect the performance of our models. Sun bears are known to be opportunistic feeders, and their habitat use can be influenced by food availability (Tee et al., 2021). Additionally, species may have different realized niches due to spatial variation in predators, competitors, or other biotic factors (Mouton et al., 2010). Although sun bears reportedly have strong home range fidelity, they may also move in response to mating opportunities and potential predation risks (Augeri, 2005). In our study, sun bear detection probability was higher when tiger presence occurred (Fig. 3.4), and no leopards were detected at stations where sun bears were detected. This suggests that tigers may provide protection from other potential sun bear predators, such as leopards (Chapter 2). While our models did not take into account the potential distribution of leopards (as no leopard was recorded in our 240 km² sampling plots that were used as training data), future habitat studies should include relevant ecological variables such as predators, competitors, food availability, and habitat safety when predicting the distribution of sun bears.

Forest types such as evergreen forests, dipterocarp forests, and bamboo forests can affect the habitat preference of sun bears based on food availability. However, we did not include forest types in our modeling process because we did not have detailed assessments of forest types in our study areas. Additionally, our field observations indicated that the the forest types in our intensive sampling plots (240 km²) differed from the other parts of the sanctuary, especially the southeastern hilly parts of the larger wildlife sanctuary (Htike, pers. obs.). The intensive sampling plot is dominated by evergreen forest types, a few grasslands, and wetland areas, while the southeastern parts...
of the sanctuary are dominated by dipterocarps forests. This may have affected the accuracy of our models, as we suspect that forest type can play a critical role in determining the habitat use and preference of sun bears. For instance, evergreen forests are known to have higher plant diversity and productivity, which can support a greater abundance and diversity of prey species and provide better shelter and cover for sun bears. On the other hand, dipterocarps forests may have a lower plant diversity, but they can provide a rich source of food in the form of nuts and fruits, which can be an important food source for sun bears during certain seasons. Therefore, the variation in forest types between our intensive sampling plot and other parts of the wildlife sanctuary could have affected the predictive performance of our models in these areas. We recommend that future studies include forest types into the modeling process to improve the accuracy and reliability of habitat use predictions for sun bears in different areas.

Our model is based on data collected during a single season, and the survey periods did not cover the entire year. This means that our model may not fully capture the seasonal variation in habitat use patterns of sun bears, which can be influenced by factors such as food availability, climate, and breeding behavior. For instance, during the fruiting season, sun bears may have a higher preference for forest areas with abundant fruit trees, while during the dry season, they may prefer areas with reliable water sources. In addition, breeding behavior can also influence habitat use, as females with cubs may require more secluded areas for protection and shelter. Therefore, we recommend that future studies collect data across multiple seasons and years to better assess sun bears' seasonal and long-term habitat use patterns.
The application of the models from Chapter 2 in predicting the habitat use of sun bears was not very effective, as highlighted by the results of this study. We suggest that obtaining higher detection probabilities on the order of 0.3 would improve prediction performance of the models. Although logistically challenging, we recommend increasing sampling periods by deploying camera traps for more than 45 days in the study area. We also suggest including information on potential predators, such as leopards, in the models if they are known to impact the distribution and behavior of the target species significantly.

**Conclusion**

Overall, our attempts to apply the sun bear habitat models we developed from our smaller study area to the whole of HMS resulted in low predictability. Although several performance metrics provided us insight into sun bear habitat use, there were inconsistent results between the AUC scores and several performance metrics. To better understand sun bear habitat use to inform conservation efforts in HWS, we recommend that future sun bear surveys 1) increase the number and size of sampling sites to provide a more representative sample of the ecological factors that affect sun bears distribution and habitat use; 2) expand information on potential predators, such as leopards; 3) incorporate additional information on forest types; 4) extend data collection across multiple seasons and years to better accommodate for the seasonal and long-term habitat use patterns of sun bears.
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Table 3.1: Model performance at camera trap stations for Malayan sun bears in the Htamanthi Wildlife Sanctuary, contrasting number of stations with predicted and observed sun bear occurrence. (Table of Confusion matrix or error matrix).

<table>
<thead>
<tr>
<th>Model</th>
<th>Predicted probability of occupancy</th>
<th>No. of stations</th>
<th>% of stations with photograph recorded</th>
</tr>
</thead>
<tbody>
<tr>
<td>tiger.elev.slp</td>
<td>≥ 90</td>
<td>98</td>
<td>17</td>
</tr>
<tr>
<td></td>
<td>&lt; 90</td>
<td>6</td>
<td>17</td>
</tr>
<tr>
<td>elev.slp</td>
<td>≥ 90</td>
<td>98</td>
<td>17</td>
</tr>
<tr>
<td></td>
<td>&lt; 90</td>
<td>6</td>
<td>17</td>
</tr>
<tr>
<td>tiger.ndvi.slp</td>
<td>≥ 90</td>
<td>98</td>
<td>16</td>
</tr>
<tr>
<td></td>
<td>&lt; 90</td>
<td>11</td>
<td>9</td>
</tr>
<tr>
<td>ndvi.slp</td>
<td>≥ 90</td>
<td>87</td>
<td>17</td>
</tr>
<tr>
<td></td>
<td>&lt; 90</td>
<td>17</td>
<td>18</td>
</tr>
<tr>
<td>tiger.d.water.slp</td>
<td>≥ 90</td>
<td>100</td>
<td>17</td>
</tr>
<tr>
<td></td>
<td>&lt; 90</td>
<td>4</td>
<td>25</td>
</tr>
<tr>
<td>d.water.slp</td>
<td>≥ 90</td>
<td>92</td>
<td>17</td>
</tr>
<tr>
<td></td>
<td>&lt; 90</td>
<td>12</td>
<td>17</td>
</tr>
<tr>
<td>Ensemble</td>
<td>≥ 90</td>
<td>96</td>
<td>18</td>
</tr>
<tr>
<td></td>
<td>&lt; 90</td>
<td>8</td>
<td>14</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Model</th>
<th>Predicted occupancy probability</th>
<th>Observed</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Yes</td>
</tr>
<tr>
<td>tiger.elev.slp</td>
<td>≥ 90</td>
<td>17</td>
</tr>
<tr>
<td></td>
<td>&lt; 90</td>
<td>1</td>
</tr>
<tr>
<td>elev.slp</td>
<td>≥ 90</td>
<td>17</td>
</tr>
<tr>
<td></td>
<td>&lt; 90</td>
<td>1</td>
</tr>
<tr>
<td>tiger.ndvi.slp</td>
<td>≥ 90</td>
<td>16</td>
</tr>
<tr>
<td></td>
<td>&lt; 90</td>
<td>1</td>
</tr>
<tr>
<td>ndvi.slp</td>
<td>≥ 90</td>
<td>15</td>
</tr>
<tr>
<td></td>
<td>&lt; 90</td>
<td>3</td>
</tr>
<tr>
<td>tiger.d.water.slp</td>
<td>≥ 90</td>
<td>17</td>
</tr>
<tr>
<td></td>
<td>&lt; 90</td>
<td>1</td>
</tr>
<tr>
<td>d.water.slp</td>
<td>≥ 90</td>
<td>16</td>
</tr>
<tr>
<td></td>
<td>&lt; 90</td>
<td>2</td>
</tr>
<tr>
<td>Ensemble</td>
<td>≥ 90</td>
<td>17</td>
</tr>
<tr>
<td></td>
<td>&lt; 90</td>
<td>1</td>
</tr>
</tbody>
</table>
Table 3.2: Model validation statistics; AUC, corr (Correlation), threshold, sens (sensitivity), spec (specificity), tss (true scale statistics), and kappa (Kappa) for the top models of Malayan sun bears distribution in the Htamanthi Wildlife Sanctuary, validated by independent sanctuary-wide data set.

<table>
<thead>
<tr>
<th>model</th>
<th>auc</th>
<th>corr</th>
<th>ll</th>
<th>threshold</th>
<th>sens</th>
<th>spec</th>
<th>tss</th>
<th>kappa</th>
</tr>
</thead>
<tbody>
<tr>
<td>P(tiger) pi(elev_slp)</td>
<td>0.6176</td>
<td>0.0239</td>
<td>-</td>
<td>0.6600</td>
<td>0.9412</td>
<td>0.0490</td>
<td>-0.0098</td>
<td>-0.0029</td>
</tr>
<tr>
<td>P(.) pi(elev_slp)</td>
<td>0.6142</td>
<td>0.0238</td>
<td>-</td>
<td>0.5000</td>
<td>0.9412</td>
<td>0.0490</td>
<td>-0.0098</td>
<td>-0.0029</td>
</tr>
<tr>
<td>P(tiger) pi(ndvi_slp)</td>
<td>0.5479</td>
<td>0.0376</td>
<td>-390.7846</td>
<td>0.9700</td>
<td>0.7647</td>
<td>0.3529</td>
<td>0.1176</td>
<td>0.0467</td>
</tr>
<tr>
<td>P(.) pi(ndvi_slp)</td>
<td>0.5502</td>
<td>0.0470</td>
<td>-333.1941</td>
<td>0.9800</td>
<td>0.4118</td>
<td>0.6961</td>
<td>0.1078</td>
<td>0.0712</td>
</tr>
<tr>
<td>P(tiger) pi(d.water_slp)</td>
<td>0.5381</td>
<td>0.0167</td>
<td>-441.7506</td>
<td>0.9500</td>
<td>0.9412</td>
<td>0.1078</td>
<td>0.0490</td>
<td>0.0153</td>
</tr>
<tr>
<td>P(.) pi(d.water_slp)</td>
<td>0.5381</td>
<td>0.0339</td>
<td>-318.6287</td>
<td>0.9500</td>
<td>0.7059</td>
<td>0.3725</td>
<td>0.0784</td>
<td>0.0321</td>
</tr>
</tbody>
</table>
Table 3.3: Glossary of measures and index values used to evaluate performance of sun bear occurrence models in the Htamanthi Wildlife Sanctuary, Myanmar

<table>
<thead>
<tr>
<th>Definition</th>
<th>Interpretation</th>
</tr>
</thead>
<tbody>
<tr>
<td>AUC</td>
<td>The area under (a ROC) curve is a measure of the accuracy of a quantitative diagnostic test. It is a scalar value that ranges between 0 and 1. AUC value of 1 represents a perfect model that accurately classifies all cases, while a value of 0 represents a model that has no predictive power.</td>
</tr>
<tr>
<td>corr</td>
<td>Correlation between Observed and estimate</td>
</tr>
<tr>
<td>ll</td>
<td>The log-likelihood (LL) value represents the goodness of fit of a model to the observed data. A higher LL value indicates a better fit of the model to the data, while a lower LL value indicates a poor fit.</td>
</tr>
<tr>
<td>threshold</td>
<td>The kappa threshold is calculated by comparing the observed agreement between the raters to the agreement that would be expected by chance. The value of kappa can range from 0, indicating complete disagreement between the raters, to 1, indicating perfect agreement. A kappa value of 0.8 or higher is generally considered to indicate very good agreement, while a value of 0.6 or higher is considered to indicate substantial agreement.</td>
</tr>
<tr>
<td>Definition</td>
<td>Interpretation</td>
</tr>
<tr>
<td>------------</td>
<td>----------------</td>
</tr>
<tr>
<td><strong>sens</strong></td>
<td>Sensitivity is a statistical measure that describes the ability of a test or model to correctly identify positive cases. In the context of binary classification, sensitivity refers to the proportion of true positive cases that are correctly identified.</td>
</tr>
<tr>
<td></td>
<td>if a model has a sensitivity of 0.8, it means that the model correctly identifies 80% of the positive cases. A high sensitivity value is desirable for a diagnostic test, as it indicates that the test has a low false negative rate, i.e., it is able to identify most of the positive cases.</td>
</tr>
<tr>
<td><strong>spec</strong></td>
<td>Specificity (or true negative rate) is a measure of the accuracy of a binary classification test that indicates how well the test correctly identifies negative samples, that is, samples that do not belong to a particular class of interest.</td>
</tr>
<tr>
<td></td>
<td>A high specificity indicates that the test has a low rate of false positives, and therefore is able to correctly identify negative samples with high accuracy.</td>
</tr>
<tr>
<td><strong>tss</strong></td>
<td>TSS (True Skill Statistic) is a measure of the overall accuracy of a binary classification model. It is used to compare the performance of different models or the same model under different conditions.</td>
</tr>
<tr>
<td></td>
<td>TSS ranges from -1 to 1, with values close to 1 indicating a highly accurate model and values close to -1 indicating a highly inaccurate model.</td>
</tr>
</tbody>
</table>
### Definition

| kappa | Kappa value is a statistic used to measure inter-rater reliability (agreement) between two or more raters in a categorical data classification task. It is a measure of agreement beyond chance, taking into account the number of agreements and disagreements that are expected by chance. |

### Interpretation

| The Kappa value ranges from -1 to 1, where values close to 1 indicate high agreement and values close to 0 indicate agreement that is equivalent to chance. Negative Kappa values indicate less agreement than expected by chance. |
Figure 3.1: Location of camera trap plots for Malayan Sun Bear (*Halartos malayanus*) survey in the Htamanthi Wildlife Sanctuary (HWS) in Northern Myanmar.
Figure 3.2: Camera trap locations for the sanctuary-wide tiger (*Panthera trigris*) survey in the Htamanthi Wildlife Sanctuary (HWS), Northern Myanmar.
Figure 3.3: The probability of Malayan sun bear (*Halartos malayanus*) occupancy predicted in 240 - km² sampling plots for the Htamanthi Wildlife Sanctuary (HWS). Sun bear-detected station points were derived from an independent sanctuary-wide survey. A) prediction from the elevation and slope model, B) prediction from distance to water and slope model, C) prediction from NDVI and slope model, and D) Model ensemble prediction based on the AIC weight of each model.
Figure 3.4: The projection of probability of Malayan sun bear (*Halartos malayanus*) occupancy in 240 - km² sampling plots for the Htamanthi Wildlife Sanctuary (HWS). Sun bear, tiger, and leopard detected stations points were derived from independent sanctuary-wide survey. A) sun bear detected stations, B) stations detected leopard, C) stations detected tigers, and D) stations that detected sun bear, tiger, and leopard in HWS.