**Fine-scale ecological and anthropogenic variables predict the habitat use and detectability of sloth bears in the Churia habitat of east Nepal**

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**Abstract**

Once widespread throughout the tropical forests of the Indian Subcontinent, the sloth bears have suffered a rapid range collapse and local extirpations in the recent decades. A significant portion of their current distribution range is situated outside of the protected areas (PAs). These unprotected sloth bear populations are under tremendous human pressures, but little is known about the patterns and determinants of their occurrence in most of these regions. The situation is more prevalent in Nepal where virtually no systematic information is available for sloth bears living outside of the PAs. We undertook a sign survey-based single-season occupancy study intending to overcome this information gap for the sloth bear populations residing in the Trijuga forest of southeast Nepal. Sloth bear sign detection/non-detection data and field-based covariates data were collected at the 74 randomly chosen 4-km2 grid cells using a varying number of 400m long transects in each grid cell. From our results, the model-averaged estimate of site use probability (*ψ* ± SE)was estimated to be 0.432 ± 0.039, which is a 13% increase from the naïve estimate (0.297) not accounting for imperfect detections of sloth bear signs. The presence of termite mounds and the distance to the nearest water source were the most important variables affecting the habitat use probability of sloth bears. The average site-level detectability (*p* ± SE) of sloth bear signs was estimated to be 0.195 ± 0.003 and was significantly determined by the index of human disturbances. We recommend considering the importance of fine-scale ecological and anthropogenic factors in predicting the sloth bear-habitat relationships across their range in the Churia habitat of Nepal.

**Keywords:** Churia hills, detectability, habitat use, Nepal, occupancy, sloth bear

**1. INTRODUCTION**

Among the terrestrial mammals, large-bodied species inhabiting the lowland areas of the developing regions are at greater risk of human-induced extirpation (Schipper *et al.*, 2008). One particular cause for this is the widespread habitat loss and degradation undergoing in these areas, limiting the ability of large mammals to meet their complex biological requirements (Ceballos and Ehrlich, 2002; Cardillo *et al.*, 2005). For example, in the tropics of South and Southeast Asia, increased habitat conversion has isolated most of the threatened large mammals to generally small protected reserves, whereas remaining natural habitats outside the reserves are largely fragmented and degraded (Wikramanayake *et al.*, 2004; Wong and Linkie, 2013). Population dispersal of certain charismatic species (e.g. tiger, elephant) has been facilitated through the initiation of landscape-scale habitat connectivity approaches (Brodie *et al.*, 2016). But, such single or few species-focused management approaches often come at the cost of undermining the ecological needs and threats of many other sympatric species that have important ecological and conservation value (Wang *et al.*, 2018, 2021). This is especially true for species having a less charismatic demeanor with poor representation in the network of PAs (Guan *et al.*, 2015; Wang *et al.*, 2021). The south Asian endemic sloth bear (*Melursus ursinus*) exemplifies the conservation challenges faced by such species (Puri *et al.*, 2015).

Sloth bears in general are a lowland species that were once widespread throughout the tropical forests of the Indian subcontinent (Brander, 1982). However, over the past decades they have suffered rapid range collapse and local extirpations, leading to patchy distributions in lowland habitat remnants of India, Nepal, Sri Lanka, and probably Bhutan (Garshelis *et al.*, 1999; Yoganand *et al.*, 2006). It has been estimated that more than half of the sloth bears’ remaining range is not under any forms of protection (Dharaiya, Bargali and Sharp, 2016). In these areas, sloth bears are under tremendous human pressures arising primarily from large scale habitat loss and degradation, and to a lesser extent from poaching and harvest of live cubs for use as “dancing bears” (Garshelis *et al.*, 1999; D’Cruze *et al.*, 2011; Dharaiya, Bargali and Sharp, 2016). Additionally, sloth bears are perceived as a dangerous species due to their frequent involvement in human attacks. As a result, locals support for conservation has eroded, and bears often become a subject of human persecution for retaliation or self-defense (Ratnayeke *et al.*, 2014; Garcia, Joshi and Dharaiya, 2016; Debata *et al.*, 2017). Given the lack of enforcement in abating most of these threats in the unprotected regions, greater than 30% of sloth bears’ population is projected to decline within the next few decades (Dharaiya, Bargali and Sharp, 2016).

Furthermore, the conservation of sloth bears is hindered by the lack of scientifically sound information required for effective conservation planning. There are rough estimations regarding the overall distribution and population status of sloth bears (Garshelis *et al.*, 1999). Though studies examining human-sloth bear conflict are emerging (Ratnayeke *et al.*, 2014; Garcia, Joshi and Dharaiya, 2016; Debata *et al.*, 2017; Dhamorikar *et al.*, 2017; Sharp *et al.*, 2020; Prajapati, Koli and Sundar, 2021), research on sloth bear-habitat relationships and space use patterns are very limited, mostly restricted to a few geographic landscapes of their entire range (Joshi, Garshelis and Smith, 1995; Ratnayeke, van Manen and Padmalal, 2007; Puri *et al.*, 2015; Srivathsa *et al.*, 2018). Being a relatively widespread species with observed geographic variations in the use of resources and habitats (Joshi, Garshelis and Smith, 1995; Ratnayeke, van Manen and Padmalal, 2007), understanding the fine-scale patterns and drivers of sloth bear occurrence become crucial for effective site-specific conservation planning. Such information would be especially vital in managing the populations residing in the fragmented landscapes outside of PAs (Akhtar, Bargali and Chauhan, 2004; Puri *et al.*, 2015).

Sloth bears in Nepal are a nationally endangered species (Jnawali *et al.*, 2011). They have a small estimated population (<250 adults) and a narrow range of distribution in the fragmented forests of southern lowlands and adjacent Churia hills (Garshelis, Joshi and Smith, 1999; Jnawali *et al.*, 2011). Four PAs provide formal protection to sloth bears in this range, but a large portion of their habitat remains unprotected (Garshelis *et al.*, 1999). They are fairly common and somewhat comprehensively studied in the Chitwan National Park (CNP) of central Nepal, elsewhere they are considered rare with poor ecological information available (Garshelis *et al.*, 1999; Garshelis, Joshi and Smith, 1999). Limited remnants of natural lowland habitats outside the PAs have made the Churia hills to be the last refuge for sloth bears in these areas (Garshelis *et al.*, 1999). However, there is largely inadequate information about the sloth bears inhabiting the Churia hills. Even the baseline reports on the distribution and abundance are extrapolated based on the opinions of experts and locals (Garshelis *et al.*, 1999; Jnawali *et al.*, 2011). Efforts made to ground-truth such reports and investigate the local-level habitat correlates of sloth bears are extremely rare.

We carried out this study as an effort of bridging this information gap for sloth bears inhabiting the typical Churia habitat in the Trijuga forest of east Nepal. This forest is known to shelter one of the probable strongholds of the sloth bear population (40-50 individuals) within Nepal. But, again the assessment is grounded on the anecdotal evidences provided by the locals (Jnawali *et al.*, 2011). A few recent studies corroborated the presence of sloth bears and also revealed the issue of human-sloth bear conflict in parts of this region (Pokharel and Aryal, 2020; Subedi *et al.*, 2021). This has made it essential to investigate how the sloth bears use this forest patch, so that appropriate local-level conservation and management plans can be devised. We used sign survey-based single-season occupancy modelling to reliably elucidate the patterns and determinants of habitat use by sloth bears in the Trijuga forest. The obtained findings provide baseline data with implications for the design of future studies targeted at sloth bears in the Trijuga forest as well as similar areas of the Churia hills.

**2. MATERIALS AND METHODS**

**2.1 Study Area**

The Trijuga or Triyuga forest is one of the largest remaining patches of lowland forest outside the PAs of Nepal. It is approximately 430 km2 in size and is distributed under 9 municipalities of Udayapur and Saptari districts that fall under the administration of Province 1 and Province 2 respectively (Aryal, Subedi and Aryal, 2020). This forest is a part of the Churia hills that runs east to west parallel to the Himalayas in Nepal (Subedi *et al.*, 2021). The peripheral areas of the forest are managed as community forests and the remaining is designated to national forest. Extensive agricultural lands interspersed with human settlements surround the Trijuga forest from all sides except the north-western part where the habitat is connected through a narrow patch to the Churia range moving westwards (Figure 1). The lower tropical ecological zone dominates this region with average annual temperature and precipitation falling in the range of 23 - 25.5oC and 1,159 - 2827mm respectively (Lillesø *et al.*, 2005). The elevation of the forest ranges from 104 to 430m. Soil erosion, landslides, and flash floods are frequent in this area during the peak monsoon (June-September) similar to other parts of the geologically fragile Churia hills (Ghimire, Higaki and Bhattarai, 2013). On the other hand, the summer season (March – June) is associated with extensive dryness leading to less availability of water sources and forest fires (Thapa and Kelly, 2017).

Vegetation of the Trijuga region consists of both dry as well as moist deciduous forests. *Shorea robusta* dominates much of the northern part of the forest. Progressing towards the south the vegetation is slowly replaced by mixed deciduous forest, and it becomes the dominant forest type in the southern boundary. *Dalbergia latifolia, Acacia catechu, Terminalia tementosa,* and *Semicarpous anacardium* are the commonly found trees in mixed deciduous forest. Deciduous riverine forest mostly dominated by *Dalbergia sisoo* and *Acacia catechu* is prevalent along the river banks of the study area. In addition to sloth bears, mammalian faunas, such as common leopard (*Panthera pardus*)*,* Asiaticelephant (*Elephas maximus*)*,* barking deer (*Muntiacus muntjak*)*,* wild boar (*Sus scrofa*)*,* jungle cat (*Felis chaus*)*,* golden jackal (*Canis aureus*)*,* Bengal fox (*Vulpes bengalensis*)*,* rhesus macaque (*Macaca mulatta*)*,* andTarai gray langur (*Semnopithecus hector*)*,* can be found in the Trijuga forest (Shah *et al.*, 2018; Aryal, Subedi and Aryal, 2020). Local people of the area are highly dependent on the forest resources for their livelihood and they harvest products from different plants including those that are of dietary importance to sloth bears. Some of such shared plant resources are *Bombax ceiba*, *Ficus sps.*, *Cassia fistula*, *Magnifera indica*, *Zizyphus sps.*, *Aegle marmelos, Bridelia retusa*, *Syzygium cumini,* and *Phoenix humilis* (Shah *et al.*, 2018).

**2.2 Sampling design**

The intention of this study was to investigate the site use parameters of sloth bears rather than true occupancy, thus we designed the sampling framework accordingly (Mackenzie and Royle, 2005). Our sample units consisted of 4-km2 grid cells, smaller than the known home range size of sloth bears across much of their range (Joshi, Garshelis, and Smith 1995; Yoganand, Rice, and Johnsingh 2005; *but see* Ratnayeke, van Manen, and Padmalal 2007). This sample size was chosen as a tradeoff between limited resources and manpower available for surveys and the small movement rate of sloth bears during the daylight hours (Joshi, Smith and Garshelis, 1999; Ramesh *et al.*, 2013). We overlaid 144 4-km2 grid cells on the land cover map of Trijuga forest using the Fishnet tool in ArcGIS 10.4. We eliminated 35 grid cells that fell within the forest edges and had less than 50% area (<2 km2) within its boundary, after deciding them to be less suitable for use by sloth bears through direct habitat observation and consultation with the locals. From the remaining 109 grid cells we randomly selected 78 (71%) for sampling, out of which 4 cells could not be surveyed due to difficult topographic conditions. Thus, the survey was carried out in 74 (68%) grid cells. Time spent to survey each of the grid cells ranged from 3 to 4.5 hours and the surveys were carried out between 10 AM to 4 PM.

**2.3 Field data collection**

The principal requirement of occupancy studies is the detection histories of the target species through the use of either temporal or spatial replicates (Mackenzie *et al.*, 2002; Kendall and White, 2009). Performing temporally replicated surveys over a large scale, however, often becomes unfeasible because of the associated logistical requirements (Hines *et al.*, 2010). Due to similar reasons, we opted for spatial replicates. The replicates were delineated as linear transects of 400m length that were arranged consecutively and were placed along the substrates that maximized detectability of species indirect signs (Hines *et al.*, 2010; Karanth *et al.*, 2011). Substrates such as forest roads and trails provide an important pathway for sloth bears to travel within their habitat, thus increasing the likelihood of sign detections (Puri *et al.*, 2015; Srivathsa *et al.*, 2018). But, similar substrates were less available in our study area. Thus, we concentrated our survey efforts on sandy riverbeds (78%, *n* = 355 400m transects). In the Churia habitat similar to ours, sandy riverbeds are widespread and can be instrumental in recording indirect signs, especially tracks of large carnivores (Harihar and Pandav, 2012). The remaining transects were placed along forest trails (20%, *n* = 91) and ridgelines (2%, *n* = 10). We ensured adequate spatial coverage of the sample grid cells by delineating transects prior to the field surveys (Karanth *et al.*, 2011; Harihar and Pandav, 2012). The number of transects ranged from 4 to 10 (Mean = 6.16 replicates/grid cell), which was primarily defined by the proportion of available habitat and prevailing topography in the cell (Wibisono *et al.*, 2011; Puri *et al.*, 2015). Within the 400m long transects, we collected sloth bear detection/non-detection data and field-based covariates data at each 100m segment. Only the first detected and clearly identified sign at the 100m segment was noted as “1” indicating detection and as “0” for non-detection (Mackenzie *et al.*, 2002; Karanth *et al.*, 2011). Detection histories were later constructed by aggregating the segment-level detection/non-detection data to the 400m transects, whereas the values of field-based covariates were all averaged at the grid cell-level to form site covariates.

For this study, we included the indirect signs of sloth bears in the form of pugmarks, scats, and excavated holes on termite mounds and ground (Garshelis, Joshi and Smith, 1999; Puri *et al.*, 2015; Srivathsa *et al.*, 2018). A team of 3 to 5 surveyors actively looked for these signs at the 400m transects. The surveyors were sufficiently familiarized with sloth bear signs and survey protocols through pilot training surveys prior to the actual field surveys. We only considered relatively fresh and unambiguously identified signs for the analysis to reduce biasness that could arise from sign degradation and false-positive detections (Miller *et al.*, 2011; Rhodes *et al.*, 2011). Due to the possibility of misidentifying the ground holes by sloth bears with that of other species, such as wild boars, we only consigned holes with ≥30 cm depth (Garshelis, Joshi and Smith, 1999) and containing secondary identification features (e.g. claw marks or pugmarks) to sloth bears. We carried out the field surveys in the post-monsoon season between 2nd October to 3rd December 2020 as an effort of minimizing the variation in the sign detection process due to rainfall (Karanth *et al.*, 2011; Harihar and Pandav, 2012).

**2.4 Covariates selection**

We reviewed available literatures on sloth bear ecology and devised covariates that seemed important in influencing the spatial pattern of sloth bears at the Trijuga forest (Table 1). Sloth bears are inclined towards myrmecophagy (Palei, Mohapatra and Sahu, 2014; Rather *et al.*, 2020) and studies from Nepal show their greater reliance on termites for food (Joshi, Garshelis and Smith, 1997; Khanal and Thapa, 2015). Sloth bears prefer areas with heterogeneous terrain and proximity to water sources for different purposes such as resting, denning, and feeding (Akhtar, Bargali and Chauhan, 2004; Ghimire and Thapa, 2015; Puri *et al.*, 2015). On the other hand, sloth bears tend to avoid or react aggressively during human encounters (Ratnayeke *et al.*, 2014; Sharp *et al.*, 2020) and are sensitive to overharvesting of forest products, overgrazing, poaching, and minerals extraction, especially outside the PAs (Garshelis *et al.*, 1999; Bargali, Akhtar and Chauhan, 2004; Dharaiya, Bargali and Sharp, 2016). These anthropogenic factors can have both direct or indirect consequences on the survival of sloth bears. Taking into account the available information, we hypothesized that the availability of termites and a high degree of terrain heterogeneity would positively influence the site use intensity of sloth bears. Similarly, we predicted the site use probability to be negatively influenced by the human disturbance factors and larger distances from the water sources. Detectability was also modelled as a function of the same site-level covariates because of their potential in exerting fluctuations on species abundance (Royle and Nichols, 2003). This approach was also helpful in minimizing the number of parameters to be estimated during the analysis (Jathanna *et al.*, 2015).

We used various methods to note or quantify the devised covariates. Because of the difficulty in detecting underground colonies of termites, we only considered the aboveground mound-building termites for this study. We carried out extensive searches, often deviating from the predefined transects at each grid cell, to note the presence/absence and count the number of termite mounds. Topographic heterogeneity was measured using the Terrain Ruggedness Index (TRI) developed by Riley et al (1999) by using the Shuttle Radar Topographic Mission (SRTM) Digital Elevation Model (DEM) data with 90 m resolution (downloaded from <https://srtm.csi.cgiar.org/>). The average value of TRI for each grid cell was used for the analysis (Thapa, Kelly and Pradhan, 2019). We georeferenced the majority of the perennial and permanent water sources during the field surveys with the help of local field assistants. A few water bodies that we failed to locate during the surveys were digitized using Google Earth Imagery. We calculated the distance from the centroid of the grid cells to the nearest water sources using the Euclidean distance tool in ArcGIS 10.4.

Likewise, we obtained an overview of the potential anthropogenic threats to sloth bears in the study area through interactions with the locals and forest officers. These interactions revealed six major threats *viz*. human-caused mortality (for retaliation, self-defense and presumably poaching), human-induced forest fires, vegetation disturbances (logging, cutting, and looping), livestock grazing, vehicular disturbances (mainly tractors for transporting riverbed minerals, firewood and timber), and direct human presence in bear habitat. However, bear killing was found to happen rarely and forest fires mostly occurred during the summer. Hence, we were unable to document the evidences of these threats during the course of this study. We incorporated the remaining four categories of threats to the framework of Barber-Meyer et al (2013) with necessary modifications for quantifying human disturbances (Thapa and Kelly, 2017; Thapa *et al.*, 2021). We recorded the evidence of livestock and their signs (L), human presence (HP), vehicular disturbance (VeD), and vegetation disturbances (VD) at each 100m segment. Due to the lack of published information regarding the degree of influence of these threats to sloth bears, we assigned equal weights (0.25) to each category and calculated human disturbance index at each segment as HDI = (L\*0.25) + (HP\*0.25) + (VeD\*0.25) + (VD\*0.25). We averaged the obtained value of HDI to the grid cell-level.

**2.5 Data analysis**

We performed the single-season occupancy analysis in program PRESENCE 13.10 (Hines, 2006). Akaike Information Criterion corrected for small sample size (AICc) was used for model comparison and selection of the best models that fit our data (Burnham and Anderson, 2002). We adopted a three-step modelling approach to model the parameters of our interest (Karanth *et al.*, 2011; Srivathsa *et al.*, 2018). In the first step, we compared the standard occupancy model (Mackenzie *et al.*, 2002) with the model accounting for correlated detections along the spatial replicates (Hines *et al.*, 2010). We initially predicted our data to follow the correlated detections model because of the potential spatial dependence in sign detection events along the consecutive spatial replicates. After identifying the most suitable model for our data, we modelled the detection parameter either in a constant form or as a function of individual covariates (Thapa, Kelly and Pradhan, 2019). The occupancy parameter at this step was kept in the most parameterized form. Finally, occupancy was modelled by fixing the covariate structure for detection probability from the top-ranked model in the previous step (Karanth *et al.*, 2011; Jathanna *et al.*, 2015; Puri *et al.*, 2015; Srivathsa *et al.*, 2018). We used either a single or additive combination of the covariates for investigating their influence on habitat use. Models with ΔAICc <2 were considered as competing models and the final estimates of site use probability and detectability were calculated by model averaging the competing models (Burnham and Anderson, 2002).

We computed β estimates of the covariates to understand the magnitude and direction (positive or negative) of their influence on the site use and detection probability. All the continuous covariates were normalized and checked for collinearity before the occupancy analysis. The categorical covariate indicating the presence or absence of termite mounds was coded as a binary variable represented by 1 or 0 respectively (MacKenzie *et al.*, 2006). None of the numerical covariates were strongly correlated (all rs < |0.5|) which enabled us to try covariate combinations without restrictions. The most parameterized model was tested for over-dispersion by calculating c-hat using a parametric bootstrap approach with 1000 iterations in program PRESENCE 13.10. The obtained value of c-hat (0.20) indicated no over-dispersion in the data (MacKenzie and Bailey, 2004). We incorporated the model-averaged estimates from our study to ArcGIS 10.4 and prepared a predicted habitat use map of sloth bears in the Trijuga forest.

**3. RESULTS**

We carried out 182.4 km of transect walk and recorded 59 fresh signs of sloth bears. The signs were recorded in 22 grid cells that estimated naïve site use probability of 0.297. Pugmarks were the most abundantly encountered signs (54%, *n* = 32) compared to dugout holes on mounds and ground (24%, *n* = 14) and scats (22%, *n* = 13). Sign detections occurred in sites that were closer to the georeferenced water sources and had lesser human disturbances in comparison to the sampling sites as a whole (Table 1). We documented the presence of termite mounds in 25 grid cells that had an average of 4.37 (SE = 0.61) mounds/ha. The majority of obtained signs were in mixed deciduous forests (76%, *n* = 45) followed by *Shorea robusta* forest (24%, *n* = 14), and no signs were detected in the riverine forests.

**3.1 Modelling detection and habitat use probability**

Contrary to our expectations, the standard model assuming independence among the detection events better fitted our data (AICc Weight = 0.69). Though the correlated detection model was also found to be competing (AICc =1.59), it’s model weight was relatively low (AICc Weight = 0.31). The probability of replicate-level presence depending on the non-detection or detection of sloth bear signs in the previous replicate [θ0(SE) = 0.97 (0.30) and θ1(SE) = 0.34 (0.17) respectively] also did not show the evidence of spatial autocorrelation among sign detections. Thereafter, we used the standard occupancy model for the analysis of our data.

We fitted 5 regression models for detection probability, including the model with constant detection *p*(.) (Table 2). The model with human disturbance index (HDI) as a covariate for detection probability emerged at the top (AICc Weight = 0.904). The second-best model was that with constant detection probability (ΔAICc = 5.58), but it received a very small model weight (AICc Weight = 0.05). HDI had a significant negative influence on the detection probability of sloth bear signs (βHDI = -0.602, 95% CI = -1.016 to -0.188, Figure 2). Detection probability (*p* ± SE)ranged from 0.051 ± 0.032 in grids with high human disturbances to 0.389 ± 0.081 in grids with the least disturbances. The model averaged detection probability (*p* ± SE) was estimated to be 0.195 ± 0.003.

For habitat use analysis, we constructed 11 regression models by fixing the HDI as a covariate for detectability (Table 3). We tried to keep the model parameters low by not including more than 2 covariates for *ψ*, except for the global model. All the fitted models received better support for the data compared to the constant model *ψ*(.) *p*(.) (ΔAICc = 17.66 for the constant model). Most support for the data was garnered by the model where *ψ* varied as an additive function of TMP and DW. The second-ranked model had a similar covariate structure with the addition of TRI (ΔAICc = 0.58). Estimated habitat use probability (*ψ* ± SE)increased from 0.371 ± 0.071 in model with no covariates to 0.432 ± 0.039 in the model-averaged estimates using the competing models. Based on the summed AICc weights of each variable, we found TMP to be the most important predictor of sloth bear habitat use (Summed AICc weight = 0.996). DW was the next important predictor (0.847), whereas TRI received less support (0.392) and HDI had relatively negligible influence (0.052) in predicting the habitat use.

The estimated β coefficients indicated that TMP had a strong positive influence on the habitat use probability of sloth bears (Figure 3). DW had a negative influence, indicating lower habitat use probability in sites with larger distances from the water sources (Figure 4). The 95% CIs did not overlap 0 for these covariates (βTMP =3.562, 95% CI = 0.817 to 6.308; βDW =-1.456, 95% CI = -2.902 to -0.011). As hypothesized, TRI had a positive influence and HDI had a negative influence on the site use probability though the 95% CIs for both the covariates overlapped zero (βTRI =0.330, 95% CI = -0.598 to 1.257; βHDI =-0.584, 95% CI = -1.439 to 0.272).

**4. DISCUSSIONS**

This is the first study investigating the habitat use correlates of sloth bears in the Churia habitat outside the protected areas of Nepal. Our findings shed light on the major factors influencing the distribution and site use pattern of sloth bears in this part of their range, and have provided baseline for evaluation of future trends in the site use with respect to the changes in given covariates. Furthermore, our study highlights the importance of considering the fine-scale ecological and anthropogenic factors in predicting the sloth-bear habitat relationships across their range in Nepal, and more specifically in the unprotected areas.

Space use patterns of large mammals, be it herbivore, carnivore, or omnivore, is most often determined by the availability and distribution of feeding resources (Karanth *et al.*, 2011; Kozakai *et al.*, 2011; Barber-Meyer *et al.*, 2013; Dupke *et al.*, 2017). Our findings showed consistency with this hypothesis by demonstrating a strong positive influence of termite mound presence on the habitat use by sloth bears in the Trijuga forest. Termites constitute an important part of sloth bears’ diet across much of their range, but their significance becomes more prevalent during the period of low fruit availability in the wild (Joshi, Garshelis and Smith, 1997; Bargali, Akhtar and Chauhan, 2004; Ratnayeke, van Manen and Padmalal, 2007). Because we conducted this study after the fruiting season (May - August) of most plants in the lowlands of Nepal (Joshi, Garshelis and Smith, 1997), the sloth bears could have exhibited strong dependency on termites for diet. Similar to our results, Das et al (2014) has reported the important function of termite mounds in the spatial pattern of sloth bears inhabiting the semi-arid region of northeastern Karnataka in India when fruiting resources were less available. Additionally, it has been reported that grassland habitat provides sloth bears and other bear species (e.g. Asiatic black bears *Ursus thibetanus*) important feeding ground by sheltering high density of underground termites and ants (Joshi, Garshelis and Smith, 1997; Yamazaki *et al.*, 2012). Due to the lack of natural grassland habitat and the associated food resources in the Trijuga forest, the sloth bears could have been more reliant on the aboveground mound-building termites for food. Nonetheless, our observation of some excavated ground holes indicates that they feed on the underground colonies of termites and ants whenever available.

Also, it is evident from secondary data (Shah *et al.*, 2018) and our observation that there exists some degree of competition between humans and sloth bears for the plant-based food resources in the Trijuga forest. This kind of competition leads to the lack of food materials for sloth bears, thereby forcing them to look for anthropogenic food sources around human settlements or farmlands (Bargali, Akhtar and Chauhan, 2004; Rajpurohit and Krausman, 2006; Prajapati, Koli and Sundar, 2021). However, throughout the Trijuga forest, we rarely documented the incidents of crop-raiding by sloth bears and human attacks was the prime type of conflict reported (Pokharel and Aryal, 2020). This further points out to the more myrmecophagous diet of sloth bears in this region. Little to no consumption of human food by sloth bears exhibiting strong myrmecophagy has been reported by other studies, even in areas with some level of overlap between humans and bears for the plant-based feeding resources (Joshi, Garshelis and Smith, 1995; Ratnayeke, van Manen and Padmalal, 2007; Rather *et al.*, 2020). However, since sloth bears are known for their ability to adjust to changing food and habitat conditions (Laurie and Seidensticker, 1977; Joshi, Garshelis and Smith, 1997), it is unwise to make inferences on the feeding behavior of sloth bears without reliable supporting data. Thus, we believe the obtained finding has opened up new avenues for further research on the feeding behavior of sloth bears in this and other parts of the Churia region in Nepal. It is essential to conduct multi-season occupancy studies (fruiting and non-fruiting seasons) incorporating the influence of both termites and fruits, whereas research on feeding ecology is equally important to reliably ascertain the pattern of sloth bears’ dietary resource utilization (Joshi, Garshelis and Smith, 1997; Ramesh *et al.*, 2012).

Distance to water sources was the second most important predictor of habitat use by sloth bears in the Trijuga forest, whereby sloth bears tended to prefer sites that are near to the perennial or permanent sources of water. Preference of areas near to water bodies has been documented for sloth bears and other bear species, such as the American black bears (*Ursus americanus*) and Asiatic black bears, because of their potential in providing suitable foraging and denning habitat (Akhtar, Bargali and Chauhan, 2004; Benson and Chamberlain, 2007; Bashir *et al.*, 2018; Jain *et al.*, 2021). For sloth bears in particular, termites are found to be abundant in moist soil conditions (Ratnayeke, van Manen and Padmalal, 2007), and foraging them becomes easy in the well-drained soft soils around water bodies (Akhtar, Bargali and Chauhan, 2004). Thus, in the relatively dry Churia habitat water could be an important limiting factor for sloth bears, as have been reported for other large mammals, including Bengal tigers (*Panthera tigris*) and gaurs (*Bos gaurus*; Thapa & Kelly, 2017). The importance of moist foraging sites should become more prevalent during the dry season when both termite mounds and ground soil become hard, impeding the bears’ ability to exploit them by digging (Joshi, Garshelis and Smith, 1995, 1997). In the same way, sloth bears are known to make use of rocky outcrops, tree cavities, and erosion made cavities located nearby water sources as denning sites for rearing cubs and for resting (Akhtar, Bargali and Chauhan, 2004; Baskaran *et al.*, 2015). During the field work for this study, we observed extensive dugout holes by sloth bears in some areas having moist riverbeds and on the moist walls of narrow gullies. Besides, all of the observed den sites (*n* = 5) that had evidences of sloth bears were in large tree cavities and rocky outcrops adjacent to small streams. Thus, it is somewhat evident that sites in proximity to water bodies provide sloth bears with appropriate feeding, denning, and resting habitat in the Churia hills, thereby increasing the probability of being used.

We predicted the positive influence of TRI and the negative influence of HDI on the habitat use by sloth bears. The results obtained were in congruence with the predicted direction of influence, even though the strength of association was weak and the 95% CIs of the estimates included zero for both the variables. High terrain heterogeneity is associated with complex topographic conditions that are less accessible to humans and also provide important denning and resting space for sloth bears (Puri *et al.*, 2015). The weaker degree of support for TRI could be due to the small spatial scale (4 km2) of our study. Different studies have reported a weak influence of TRI in describing the fine-scale species-habitat relationships including for sloth bears (Srivathsa *et al.*, 2018), American black bears (Gould *et al.*, 2019) and Asiatic elephants (Thapa, Kelly and Pradhan, 2019). Hence, if we would expand the size of our sample unit to include the area larger than the home range of sloth bears (e.g. Puri et al., 2015), there is a possibility that TRI could exhibit strong predictive power on the landscape-scale occurrence pattern of sloth bears in our study area.

On the other hand, HDI had a moderate but significant effect on the detectability of bear signs, but very little support was obtained in explaining the habitat use pattern. It could be due to the widespread nature of human disturbances in the Trijuga forest, especially during daylight hours. And sloth bears might have been forced to live with disturbances through the adoption of some spatiotemporal mechanisms of habitat segregation to minimize the degree of impact. Temporal ways of segregation, such as increased nocturnal activity, and spatial mechanisms, such as restriction of movement to certain areas of limited disturbances during times of high human activities, has been reported for several wildlife species, including bears, throughout the world (Martin *et al.*, 2010; Carter *et al.*, 2012; Gaynor *et al.*, 2018). Typically, the sloth bears are crepuscular or nocturnal species (Yoganand, Rice and Johnsingh, 2005; Ramesh *et al.*, 2013), which might have facilitated some level of coexistence between humans and sloth bears in the Trijuga region. Yet, during sub-adulthood and motherhood sloth bears are more likely to remain active during the daytime to avoid the risks of predation and aggressive encounters with adult conspecifics (Joshi, Smith and Garshelis, 1999). This could have promoted human attacks by bears mostly during daylight as documented in parts of the Trijuga forest (Pokharel and Aryal, 2020).

Additionally, in our case, the design of spatial replicates should have contributed to the significant influence of HDI on the detectability. The replicates were predominantly placed on the dry riverbeds and trails (>90%), which were also frequently used by the local people, often accompanied by livestock or vehicles, to travel in the forest. This must have caused the destruction of sloth bear signs such as pugmarks and scats, thus limiting our ability to detect them during surveys. We suggest future studies be targeted in understanding the spatial as well as temporal variations in sloth bear use of habitat in response to human disturbances. The use of new technologies, such as camera traps can be instrumental in collecting data necessary for such analysis (Carter *et al.*, 2012), while it also has the potential to minimize biasness arising from sign degradation by human activities that may occur along the spatial replicates.

**5. CONCLUSIONS AND MANAGEMENT IMPLICATIONS**

The conservation policies and practices in Nepal are largely biased towards the large mammals. Yet, species like the sloth bears have never garnered special conservation interest, and are not listed as the protected species of Nepal (Heinen and Yonzon, 1994). Our study demonstrates the significance of protecting the sloth bear populations outside the PAs, where they are under intense anthropogenic pressures and have their distribution minimally overlapped with the conservation-focused species. Our results indicate that the fine-scale space use patterns of sloth bears in the Trijuga forest is determined by the availability and distribution of basic ecological resources. Hence, the long-term survival of sloth bears in this area can only be ensured given their foraging, denning, and resting habitat are maintained in good quality. In this regard, we suggest that the predictors of sloth bears’ site use identified in this study (i.e. the presence of termite mounds and the proximity to water sources) should also be applicable to other areas of their distribution in the Churia range of Nepal.

Moreover, given the high rate of habitat conversion, encroachment, and other anthropogenic disturbances undergoing in the Churia hills (Subedi *et al.*, 2021), it is obvious that these basic resources are being depleted at a faster rate. Though not evident in our study, the degree of human disturbances can have a profound impact on the occurrence probability of sloth bears (Puri *et al.*, 2015), and the major priority should be to regulate human activities in the probable areas of bear occurrence in a way that has a minimal impact on the long term conservation of the species. The predictive map prepared in this study has prioritized sites based on their probability of being used by sloth bears (Figure 5). For example, the sites in the eastern and west-central part of the Trijuga forest have higher use probability. Conservation and habitat-management interventions should, therefore, be targeted to these areas through minimization of human disturbances. Expanding similar assessments to other parts of the Churia range can help us identify major distribution hotspots of sloth bears outside the PAs of Nepal. In addition, studies like ours could act as a starting point for carrying out human-sloth bear conflict investigation and mitigation interventions by predicting probable areas where conflicts could occur.

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**CONFLICT OF INTEREST**

The authors declare no conflict of interests.

**AUTHOR CONTRIBUTIONS**

**Manoj Pokharel:** Conceptualization (Lead), Data curation (Lead), Formal analysis (lead), Funding acquisition (Lead), Investigation (Equal), Methodology (Equal), Project administration (Lead), Resources (Equal), Validation (Equal), Visualization (Lead), Writing-Original draft preparation (Lead), Writing-Review & Editing (Equal). **Asmit Limbu:** Investigation (Equal), Methodology (Equal), Resources (Equal), Validation (Equal), Writing-Review and Editing (Equal). **Dipa Rai:** Investigation (Equal), Resources (Equal), Validation (Equal), Writing-Review and Editing (Equal). **Simrik Bhandari:** Validation (Equal), Writing-Original draft preparation (Supporting), Writing-Review & Editing (Equal). **Yadav Ghimirey:** Formal analysis (Supporting), Funding acquisition (Supporting), Validation (Equal), Supervision (Lead), Writing-Review & Editing (Equal).

**DATA ACCESSIBILITY STATEMENT**

Data associated with this manuscript will be archived in Dyrad data repository if the paper is accepted for publication.

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**Tables**

Table 1: Covariates devised to test their influence on the habitat use of sloth bears at the Trijuga forest, their predicted direction of influence, and the descriptive statistics of the numerical covariates at all the sampling sites (n = 74) and at sites where sloth bear signs were detected (n = 22)

|  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- |
| Covariate | Predicted direction of influence | All sampling sites | | Detection sites | |
| Mean | SE | Mean | SE |
| Terrain ruggedness index (TRI) | Positive | 100.67 | 3.58 | 99.96 | 7.78 |
| Distance to the nearest water source (DW) (m) | Negative | 1300 | 100 | 1029.37 | 112.91 |
| Human disturbance index (HDI) | Negative | 0.49 | 0.02 | 0.39 | 0.04 |
| Termite mounds presence (TMP) | Positive | - | - | - | - |

Table 2: Comparison of different models to identify the covariates influencing the detection probability of sloth bear signs in the Trijuga forest using global model ψ(TMP+DW+TRI) for occupancy

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
| Model | AICc | ΔAICc | AICc weight | Model Likelihood | *K* | Deviance |
| *ψ*(Global), *p*(HDI) | 245.96 | 0 | 0.904 | 1 | 6 | 233.06 |
| *ψ*(Global), *p*(.) | 251.84 | 5.88 | 0.0478 | 0.0529 | 5 | 241.2 |
| *ψ*(Global), *p*(DW) | 253.96 | 8 | 0.0166 | 0.0183 | 6 | 241.06 |
| *ψ*(Global), *p*(TRI) | 254.03 | 8.07 | 0.016 | 0.0177 | 6 | 241.13 |
| *ψ*(Global), *p*(TMP) | 254.07 | 8.11 | 0.0157 | 0.0173 | 6 | 241.17 |

TMP = Termite mounds presence; DW = Distance to the nearest water source, TRI = Terrain ruggedness index, HDI = Human disturbance index, *K* = Number of parameters

Table 3: Comparison of different models to identify the covariates influencing the habitat use probability of sloth bears in the Trijuga forest using the spatially replicated sign surveys

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
| Model | AICc | ΔAICc | AICc weight | Model Likelihood | *K* | Deviance |
| *ψ*(DW+TMP), *p*(HDI) | 245.63 | 0 | 0.4834 | 1 | 5 | 234.82 |
| *ψ*(TMP+DW+TRI), *p*(HDI) | 246.21 | 0.58 | 0.3617 | 0.7483 | 6 | 233.06 |
| *ψ*(TMP), *p*(HDI) | 249.47 | 3.84 | 0.0709 | 0.1466 | 4 | 240.94 |
| *ψ*(HDI+TMP), *p*(HDI) | 250.14 | 4.51 | 0.0507 | 0.1049 | 5 | 239.33 |
| *ψ*(TRI+TMP), *p*(HDI) | 251.21 | 5.58 | 0.0297 | 0.0614 | 5 | 240.4 |
| *ψ*(HDI), *p*(HDI) | 258.12 | 12.49 | 0.0009 | 0.0019 | 4 | 249.59 |
| *ψ*(DW), *p*(HDI) | 258.21 | 12.58 | 0.0009 | 0.0019 | 4 | 249.68 |
| *ψ*(DW+HDI), *p*(HDI) | 258.59 | 12.96 | 0.0007 | 0.0015 | 5 | 247.78 |
| *ψ*(TRI+HDI), *p*(HDI) | 260.18 | 14.55 | 0.0003 | 0.0007 | 5 | 249.37 |
| *ψ*(TRI), *p*(HDI) | 260.31 | 14.68 | 0.0003 | 0.0006 | 4 | 251.78 |
| *ψ*(TRI+DW), *p*(HDI) | 260.46 | 14.83 | 0.0003 | 0.0006 | 5 | 249.65 |

TMP = Termite mounds presence; DW = Distance to the nearest water source, TRI = Terrain ruggedness index, HDI = Human disturbance index, *K* = Number of parameters

**Figure legends**

Figure 1: Map of Trijuga forest with the selected grid cells (2 x 2 km) for sign-based occupancy surveys. Inset map shows the location of the study area in reference to the predicted distribution range of sloth bears in Nepal, adapted from IUCN Red List 2020

Figure 2: Relationship between human disturbance index (HDI) and detection probability of sloth bear signs in the Trijuga forest. The dashed lines represent the 95% confidence intervals of the detection probability

Figure 3: Relationship between the presence or absence of termite mounds and the habitat use probability of sloth bears in the Trijuga forest. The error bars represent the 95% confidence intervals of the habitat use probability

Figure 4: Relationship between the distance to the nearest water source (DW) and the habitat use probability of sloth bears in the Trijuga forest. The shaded area represents the 95% confidence intervals of the habitat use probability

Figure 5: Patterns of site use by sloth bears in the Trijuga forest based on the sign surveys. (a) Naive site use (b) Estimated probabilities of site use