ASPECTS OF ECOLOGY OF SLOTH BEAR (*MELURSUS URSINUS*) IN SANJAY TIGER RESERVE, MADHYA PRADESH

Thesis submitted for the award of the degree of
Doctor of Philosophy

in

WILDLIFE SCIENCE

by

SANKARSHAN CHAUDHURI

to

Saurashtra University
Rajkot - 360005 (Gujarat)

Under the supervision of

Dr. S. Sathyakumar, Scientist-G

भारतीय वन्यजीव संस्थान
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Wildlife Institute of India

DECLARATION

I hereby declare that the work conducted under this thesis titled "**Aspects of ecology of sloth bear (*Melursus ursinus*) in Sanjay Tiger Reserve, Madhya Pradesh**" is a record of original and independent research work done by me and subsequently submitted for the award of the degree of **Doctor of Philosophy in Wildlife Science** to the **Saurashtra University, Rajkot (Gujarat)**. This research work has been carried out under the guidance and supervision of Dr. S. Sathyakumar, Scientist-G of Wildlife Institute of India, Dehradun. The work has not formed the basis for the award of any other degree, diploma or any other qualification. I also declare that the thesis embodies my own work, analysis, observation, understanding and the particulars given in it are true to the best of my knowledge.

Sankarshan Chaudhuri
(Sankarshan Chaudhuri)
Place: Wildlife Institute of India
Date: 15/9/22

15/9/22
(Dr. S. Sathyakumar)
Supervisor

पत्रपेटी सं. 18, चन्द्रबनी, देहरादून-248 001, उत्तराखण्ड, भारत
Post Box No. 18, Chandrabani, Dehradun - 248 001, Uttarakhand, INDIA
ई.पी.ए.बी.एक्स : +91-135-2640100, 2640114, 2640115, फैक्स : 0135-2640117
EPABX : +91-135-2640100, 2640114, 2640115, फैक्स : 0135-2640117
ई-मेल/E-mail : wii@wii.gov.in, वेब/Website : www.wii.gov.in

भारतीय वन्यजीव संस्थान
Wildlife Institute of India

CERTIFICATE

This is to certify that the thesis by **Mr. Sankarshan Chaudhuri** titled "**Aspects of ecology of sloth bear (*Melursus ursinus*) in Sanjay Tiger Reserve, Madhya Pradesh**" is an original and independent research work submitted to the **Saurashtra University, Rajkot (Gujarat),** for the award of the degree of **Doctor of Philosophy in Wildlife Science.**

Mr. Sankarshan Chaudhuri has put more than six semesters of research work embodied in this thesis under my guidance and supervision. The work presented in this thesis has not been submitted to any other University or Institute for the award of any degree, diploma or distinction.

S. Sathyakumar 15/9/22

(Dr. S. Sathyakumar)
Supervisor

Ruchi

Dean
Faculty of Wildlife Science

संकायाध्यक्ष / Dean
भारतीय वन्यजीव संस्थान
WILDLIFE INSTITUTE OF INDIA
देहरादून / Dehradun

पत्रपेटी सं. 18, चन्द्रबनी, देहरादून-248 001, उत्तराखण्ड, भारत
Post Box No. 18, Chandrabani, Dehradun - 248 001, Uttarakhand, INDIA
ई.पी.ए.बी.एक्स : +91-135-2640100, 2640114, 2640115, फैक्स : 0135-2640117
EPABX : +91-135-2640100, 2640114, 2640115, फैक्स : 0135-2640117
ई-मेल/E-mail : wii@wii.gov.in, वेब/Website : www.wii.gov.in

SAURASHTRA UNIVERSITY
P.G.T.R Section
Main Office, First Floor
University Road
Rajkot - 360 005 (Gujarat)
Phone No. : 2578501
Fax : (0281)2856983
www.saurashtrauniversity.edu

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I certify that the research work was appreciated by all who were present, and the comments made by the faculty and researchers have been appropriately included in the thesis.

15/9/22

(Dr. S. Sathyakumar)
Supervisor

Ruchi

Dean
Faculty of Wildlife Science

संकायाध्यक्ष / Dean
भारतीय वन्यजीव संस्थान
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Librarian

(Sunita Agarwal)
Librarian
Wildlife Institute of India
Dehradun

(Dr. S. Sathyakumar)
Supervisor

पत्रपेटी सं. 18, चन्द्रबनी, देहरादून-248 001, उत्तराखण्ड, भारत
Post Box No. 18, Chandrabani, Dehradun - 248 001, Uttarakhand, INDIA
ई.पी.ए.बी.एक्स : +91-135-2640100, 2640114, 2640115, फैक्स : 0135-2640117
EPABX : +91-135-2640100, 2640114, 2640115, फैक्स : 0135-2640117
ई-मेल/E-mail : wii@wii.gov.in, वेब/Website : www.wii.gov.in

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CHAPTER 1: INTRODUCTION

1.1 Introduction

The sloth bear (*Melursus ursinus*) is one of the four bear species found in the Indian subcontinent and endemic to this region (Johnsingh, 2003; Yoganand et al., 2013). Unlike the other three species (Sun bear, Asiatic black bear and Himalayan brown bear), sloth bears are the most widely distributed ursid, found in the tropical and subtropical regions of the Indian sub-continent (Johnsingh et al., 2003; Sathyakumar et al., 2012; Yoganand et al., 2013; Dharaiya et al., 2016). Sloth bears possess specialised morphological and physiological characteristics, which aid them in feeding primarily on ants and termites (myrmecophagy; Yoganand et al., 2013). The sloth bear is a solitary and elusive ursid with a low basal metabolic rate (McNab, 1992), which could be attributed to its energy-related deficiencies due to its specialised niche (Yoganand et al., 2013). Such characteristics could easily make sloth bears vulnerable to large-scale anthropogenic pressure on their habitats and several other stochastic events.

Sloth bears are endemic to the Indian subcontinent. In India, they are widely distributed, ranging from the Himalayan foothills to the southernmost part of the Western Ghats (Yoganand et al., 2006; Dharaiya et al., 2016). Arid regions of Rajasthan and Gujarat limit its westernmost distribution. Sloth bears avoid completely arid areas, but they can be found in lower altitudinal forest areas (except for the Western Ghats) of the subcontinent. Despite the wide distribution of sloth bears, India harbours roughly 10% of good quality sloth bear habitat, and most (~90%) of the sloth bear population is broadly confined to moist and dry deciduous forests (Yoganand et al., 2006). Out of the moist and dry deciduous forest, it is expected that dry deciduous forest alone holds ~50% of India's remaining sloth bear population (Yoganand et al., 2006). A substantial level of human disturbance in these forest types may pose a severe threat to the future of sloth bears' survival.

Degradation and loss of habitat are the primary threats to the declining status of the sloth bear population (Dharaiya et al., 2016). However, diminished food resources, negative interactions with humans, and poaching (to some extent) are also considered to be the limiting factors to the long-term survival of sloth bears (Bargali, 2012; Bargali & Sharma, 2013; Gomez et al., 2021; Yoganand et al., 2006). Ensuring the long-term conservation of the species, the existing population of sloth bears and their habitats need active management interventions. However, ecological information on sloth bears in human-dominated landscapes becomes essential for planning efficient conservation measures.

1.2 Biology and conservation of sloth bear

Taxonomy and evolutionary history

Due to its shared characteristic features with the sloth (*Bradypus* sp.), the sloth bear was initially named *Bradypus ursinus* or bear-like sloth (Shaw & Nodder, 1791). However, Meyer recognised it as a species of bear in 1793 (Erdbrink, 1953). Later it was included under the genus *Ursus* and named *Ursus labiatus* by Blainville (1817). Subsequently, the sloth bear was placed at a sub-generical level of the genus *Ursus* and named *Ursus* (*Melursus*) *ursinus* Shaw (Erdbrink, 1953). Sloth bears evolved as a sister taxon of other bear species except for the Andean/spectacled bear and the giant panda, as confirmed by mitochondrial DNA analyses (Waits et al., 1999). However, due to genetic and morphological differences, the sloth bear was placed under the genus *Melursus* instead of *Ursus* (Wozencraft, 1989; Waits et al., 1999). There are two subspecies of sloth bears found in the Indian subcontinent: *Melursus ursinus ursinus* occurring in the main lands of India and *M. u. inornatus*, which inhabits Sri Lanka (Pocock, 1932).

The evolution of sloth bears goes back to the mid-Pliocene and perhaps continues to the early Pleistocene, reaching its present form (Erdbrink, 1953; Goldman et al., 1989; Kurten, 2017; Talbot & Shields, 1996). Subsequently, they evolved to live in tropical and subtropical regions with associated behavioural and morphological features. However, according to Talbot & Shields (1996), the greater morphological divergence of sloth bears could be attributed to recent adaptive changes rather than molecular evolution. Arguably, sloth bears possess certain peculiar morphological traits and behavioural features (extended parental care and carrying of the young) similar to other insectivorous (predominantly ant/termite eating) mammals that make them unique amongst other ursids [but see Joshi et al., (1999)].

Physical description

Being medium-sized bears, adult male sloth bears can weigh up to 140kg (range: 80-150kg), and adult females are smaller than adult males, ranging from 60 to 100kg (Prater, 1965; Garshelis et al., 1999a; Yoganand, 2005). They can stand up to 65-85cm at the shoulder, and the length between the nose and tail varies between 140-170cm (Yoganand, 2005). The coat colour is typically black with a “V” shaped buff or yellowish-white coloured breast patch. The body is covered with coarse, long, shaggy hairs of varying lengths, lacking underfur. Minute and pale-coloured hairs cover the elongated snout, and short black hairs cover the head region. The long tufts of hair are seen around the neck and beside the ears. Being primarily adaptive to an insectivorous diet, the sloth bear possesses the following suite of morphological features:

1) absence or reduced maxillary incisors, which might facilitate sucking ants and termites, 2) reduced post-canine dentition with small carnassial teeth, which may be used to crush the chitinous exoskeleton of insects, 3) protrusible mobile lips and mobile snout facilitating feeding on ants and termites, 4) nostrils that can be closed voluntarily to protect from insects while digging and feeding, 5) long (~7cm) claws of forelimbs that are predominantly used for digging.

Apart from those features above, several physiological and behavioural traits of sloth bears closely resemble mammalian ant-eaters (Laurie and Seidensticker, 1977), such as low basal metabolic rate (McNab, 1992), solitary habit, extended parental care and low reproductive rate. However, it was argued that most of the physiological and behavioural features of sloth bears are typical for other members of the ursid family, except for the extended parental care as a form of carrying young on the back (Joshi et al., 1999). The latter is a distinctive feature of sloth bears and is likely to be an outcome of coexisting with large carnivores, particularly leopards and tigers (Joshi et al., 1999). Sloth bears' low basal metabolic rate and comparatively high thermal conductance could be attributed to their occurrence in hot and tropical regions where heat loss and reduced heat production are necessary for sustenance (Yoganand, 2005).

Extent of occurrence and distribution

Sloth bears are widely distributed in the Indian subcontinent, and India covers most (~90%) of the sloth bear’s current distributional range (Dharaiya et al., 2016; Figure 1.1). Its extent of occurrence starts from the southernmost part of the Western Ghats to the Himalayan foothills (south to north) and from hills (Naga Hills) of northeastern India to the periphery of the Thar Desert, where forest cover remains (east to west) (Yoganand et al., 2013; Dharaiya et al., 2016). In the past, the overall distribution of sloth bears appeared to be similar to the present; however, it was shrunk with concurrent reduction of forest cover due to growing anthropogenic activities (Garshelis et al., 1999b). Apart from India, sloth bears are also found in three other countries, i.e., Nepal, Bhutan and Sri Lanka. In Nepal, sloth bears mainly occur in the Terai-Arc landscape, a narrow strip of forest-grassland mosaic habitat in the Himalayan foothills (Garshelis et al., 1999b). Sloth bears also occur in Bhutan, but their occurrence status remains obscure due to the extreme rarity of this species (Garshelis et al., 2015). However, in Bangladesh, sloth bears have become extinct (Islam et al., 2013). In Sri Lanka, sloth bears occupy ~18% of the total geographical area of the island and are primarily found in the contiguous forest patches (Ratnayeke et al., 2007b).

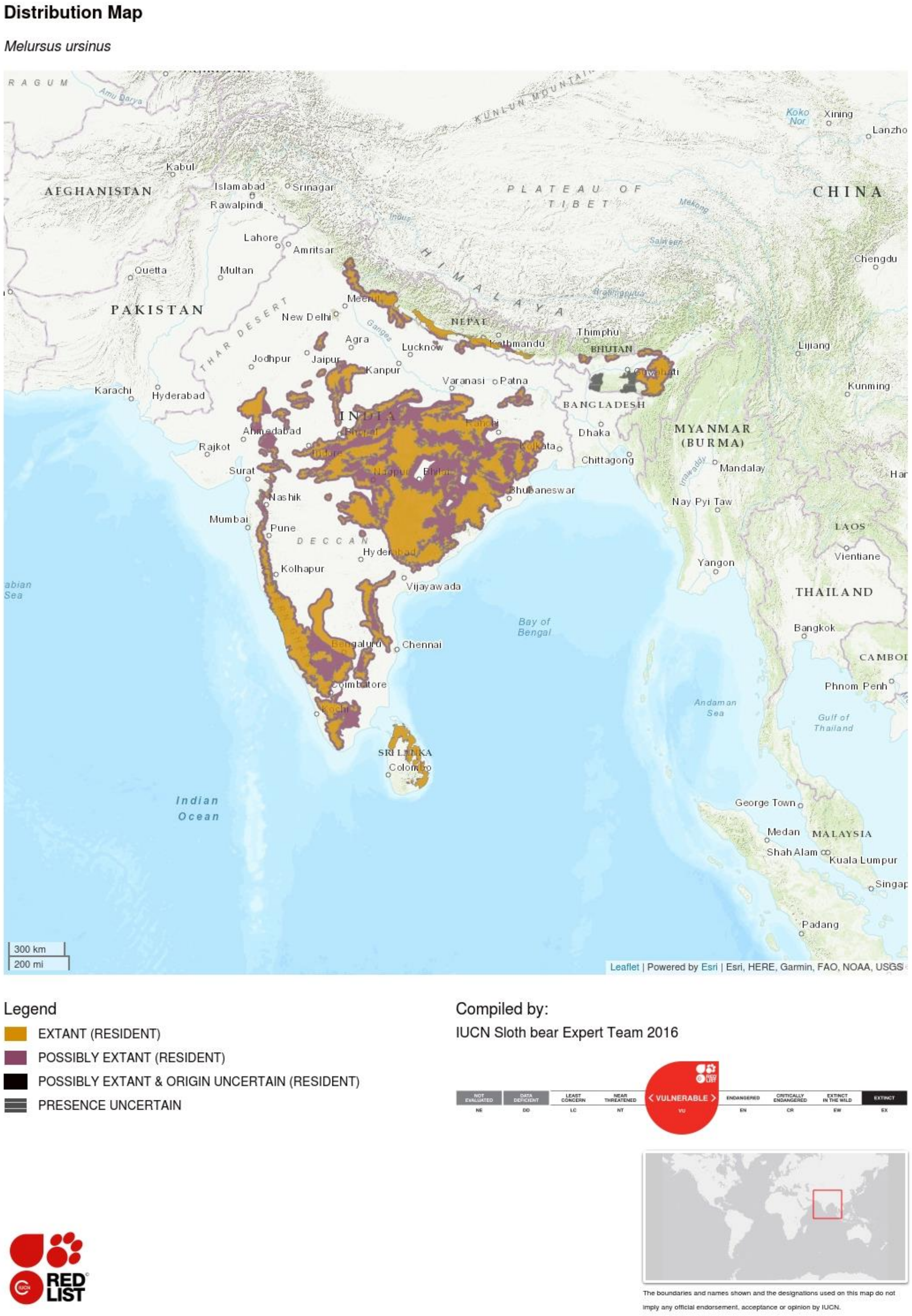

Figure 1.1. Global distribution of sloth bear

In India, central Indian highlands and the Western Ghats currently harbour substantial populations of sloth bears (Yoganand et al., 2013). However, in comparison to other forest types, sloth bears predominantly occur (~90% of the total population) in deciduous (dry and moist) forest types and more specifically, they occur in higher densities in dry deciduous forests (Yoganand et al., 2006). Broadly, sloth bears are found at an elevation of <1000m in India, though they can be found as high as 2000m in the Nilgiris and Anamalai hills of Western Ghats (Seidensticker et al., 2011). Sloth bears occur at relatively low densities in wet evergreen forest patches of India (the Western Ghats and northeastern hills) either due to the unavailability of year-round food resources or interspecific competition with Asiatic black bears and sun bears. Such ecological reasons perhaps cause the low/obscure distribution of sloth bears in northeastern hills (Seidensticker et al., 2011). Sloth bears coexist with two large predators, i.e. tiger and leopard, throughout their distributional range; however, the occurrence of large predators does not seem to limit the distribution of sloth bears (Seidensticker et al., 2011).

Nation-level assessments have been carried out to estimate the potentially occupied area by sloth bears. Several authors reported an estimated ~2,00,000 km^2 area to be occupied by sloth bears in India (Johnsingh, 2003; Chauhan, 2006). During the country-wide assessment of tigers, co-predators and prey, ~32% of the total forest area of India was found occupied by this species (Jhala et al., 2008). Sathyakumar et al. (2012) reported potential area occupied by sloth bears could be as large as ~4,00,000 km^2. A more recent study by Puri et al. (2015) estimated that sloth bears could occupy ~52% of India’s total geographical area based on a multi-scale occupancy analysis.

However, the latter was criticised due to the dissimilarity between model prediction on a coarse scale and ground reality of sloth bear distribution which, as a result, was believed to inflate the occupied area (Dharaiya et al., 2016).

Population status of sloth bear

There is a gap in knowledge of the population status of sloth bears, as very few studies with a scientifically robust framework have been conducted to estimate the population size. Most of the available studies on population estimates were based on expert opinions from various sloth bear-ranging areas in the country. Yoganand et al. (2006) extrapolated a population size of 6,000-11,000 sloth bears in India from an expert-opinion-based survey. Range-wide population estimates were found to be highly variable (from less than 10,000 to greater than 20,000 bears), perhaps as a result of different habitats and methodology-related variations (Chauhan, 2006; Garshelis, Joshi, & Smith, 1999; Sathyakumar et al., 2012 Yoganand et al., 2006). However, these estimates are largely unreliable in tracking population change over time (Dharaiya et al., 2016; Sathyakumar et al., 2012).

A few studies related to the sloth bears’ density and abundance estimation in various Protected Areas (PAs) of the Indian subcontinent indicated that sloth bears could occur at relatively high densities compared to other bear species (Dharaiya et al., 2016). However, density estimates varied considerably across habitats due to different methodologies applied. In Panna National Park (PNP), located in central Indian highlands, Yoganand (2005) reported a density of 6-8 individuals/100 km^2 based on observations of radio-collared and non-collared sloth bears. Contrastingly, much higher density estimates (25-72 individuals/100 km^2; extrapolated to 27 individuals/100 km^2) with marked seasonal and habitat-related variations were reported from Nepal's Royal Chitwan National Park (RCNP) than in PNP (Garshelis et al., 1999a). In North Bilaspur Forest Division (NBFD), a highly fragmented and disturbed forest area of the central Indian landscape, Akhtar et al. (2008) reported a density estimate of 23 bears/100 km^2, based on their den site observations. Ratnayeke et al. (2007) reported density estimates of sloth bears ranging from 24 to 38 bears/100 km^2 in Wasgomuwa National Park, Sri Lanka, based on home range estimates obtained from radio-collared individuals. However, except for density estimates reported by Garshelis et al. (1999) following the statistically robust (mark-resight) method, other density estimates should be treated as a relative measure of abundance due to the lack of a statistically rigorous framework.

Ecology of sloth bear

The sloth bear is a myrmecophagous solitary ursid, though no apparent territoriality was observed (Laurie and Seidensticker, 1977; Joshi et al., 1999; Yoganand, 2005). Unlike other bears, the sloth bear has a remarkably small home range (but varies with different habitats), a characteristic feature of other ant and termite-eating mammals (Joshi et al., 1995, 1999; Yoganand, 2005; Ratnayeke et al., 2007a). Annual home range estimates of sloth bears were high (ranging from 12.4 km^2 for a female to 84 km^2 for a male) in PNP compared to other areas (Yoganand, 2005). In RCNP, the average annual home range varied between 9.4 km^2 (for females) to 14.4 km^2 (for males) (Joshi et al., 1995). However, much smaller annual home range estimates (2.2 km^2 and 3.8 km^2 for females and males, respectively) were reported from Sri Lanka (Ratnayeke et al., 2007a). Home ranges of adult sloth bear overlap extensively, and frequent observations of non-collared unrelated bears with radio-collared individuals indicated high mutual tolerance (Joshi et al., 1995, 1999; Yoganand, 2005; Ratnayeke et al., 2007a). Broadly, sloth bears are crepuscular and nocturnal in activity (Yoganand, 2005; Ramesh et al., 2013), but they could be active during daylight hours in forests with less anthropogenic pressure, perhaps to avoid confrontation with predators and other adult bears (Joshi et al., 1999). However, in a human-dominated landscape, sloth bears were strictly nocturnal and crepuscular (Bargali et al., 2012).

Throughout its distributional range, sloth bears occur in a diverse array of habitats such as dry and moist deciduous forests, tropical evergreen forests, alluvial grasslands and scrublands (Joshi et al., 1995; Akhtar et al., 2004; Yoganand et al., 2006; Ratnayeke et al., 2007a; Seidensticker et al., 2011; Ramesh et al., 2012). However, most of the sloth bear populations survive in the moist and dry deciduous forests of the Indian subcontinent (Yoganand et al., 2006). Sloth bears are generally known to avoid areas prone to anthropogenic pressure (Ratnayeke et al., 2007a; Puri et al., 2015). Contrastingly, sloth bears living in areas with high anthropogenic pressure were also known to raid crops and fruits and often attracted to human-produced food wastes (Akhtar et al., 2004; Bargali et al., 2004; Prajapati et al., 2021). Sloth bears tend to rest during the daytime at natural cavities and crevices of escarpment habitat, dug-out holes and dense thickets of shrubs to avoid excessive heat and emerge from den sites during early evening hours (Yoganand, 2005; Akhtar et al., 2007; Baskaran et al., 2015). The thermal environment of central Indian forests and other parts of its distributional range perhaps influenced the activity patterns and the use of day-resting den sites by sloth bears (Yoganand et al., 2013).

Sloth bears’ diet is mainly comprised of insects and seasonally/year-round available fruits (Desai et al., 1997; Joshi et al., 1997; Bargali et al., 2004). However, sloth bears’ diet profile indicates that they consume ants and termites consistently (but with high seasonal variation, ≥80% except for fruiting season) throughout the year, perhaps to maintain the constant protein requirement, but also consume sugar-rich fruits (70%-90% of the diet) depending on the seasonal availability (Joshi et al., 1997; Bargali et al., 2004; Yoganand, 2005; Seidensticker et al., 2011; Sukhadiya et al., 2013; Baskaran et al., 2015; Mewada, 2015; Mewada et al., 2019; Palei et al., 2020; Rather et al., 2020).

One of the physiological characteristics of sloth bears is their low reproductive rate, probably due to extended parental care (Joshi et al., 1999). The breeding season generally starts in May and continues till July. Cubs are born during winter (November to January) in secure dens, i.e. natural caves in rocky outcrops or dug-out holes by the mother bears. Litter size varies between 1 and 3; however, two is the most frequently observed one (Laurie and Seidensticker, 1977; Joshi et al., 1999; Yoganand, 2005). After parturition, female bears leave den sites when cubs are about two months of age; however, during this period, they periodically go out for foraging at night (Joshi et al., 1999). Carrying cubs on the back (at least for six months) is typical behaviour shown by female sloth bears, perhaps as a defence strategy from predators and other adult sloth bears (Laurie and Seidensticker, 1977; Joshi et al., 1999). Association between a mother bear and cubs lasts till 1.5-2.5 years, and cubs leave before the onset of the breeding season, making adult female bears sexually receptive once every 2-3 years (Joshi et al., 1999; Yoganand et al., 2013).

Threats and conservation issues

Human-related threats in the form of poaching, retaliatory killings of sloth bear due to human-bear conflict, degradation and fragmentation of habitat, extraction of food resources of sloth bear and livestock-grazing are the limiting factors for the survival of sloth bears in long-term (Johnsingh, 2003; Bargali et al., 2005; Yoganand et al., 2006; Bargali et al., 2012; Sathyakumar et al., 2012; Bargali and Sharma, 2013; Ratnayeke et al., 2014; Gomez et al., 2021). In some parts of its distribution, especially in human-dominated landscapes, sloth bears are often involved in conflicts with humans, mostly in the form of human injury as well as the retaliatory killing of the species (Rajpurohit and Krausman, 2000; Bargali et al., 2005; Ratnayeke et al., 2014; Debata et al., 2016; Dhamorikar et al., 2017; Singh et al., 2018; Sharp et al., 2020). Poor quality of habitat and competition for food resources with humans prompted sloth bears to forage on certain crops (maise, ground nuts etc.) and other fruits (*Ziziphus mauritiana*, *Syzygium cumini* etc.) that are abundant in villages (Bargali et al., 2004). However, due to better protection and less human pressure inside PAs, human-sloth bear conflict is less likely to occur. Unfortunately, outside of PAs, conflict scenarios are increasing, and local people’s negative perceptions and attitudes toward sloth bears in these areas pose difficulties to the conservation of sloth bears (Bargali et al., 2005; Dharaiya and Ratnayeke, 2009; Mardaraj, 2014; Garcia et al., 2016).

Several cases of sloth bear poaching have been reported in many parts of its distribution (Garshelis, Joshi, Smith, et al., 1999; Sathyakumar et al., 2012); however, commercial trades for body parts of sloth bears were found to be low in comparison to other Asian bear species (Burgess et al., 2014). Despite the low demand for body parts of sloth bears, a recent (2009-2019) rise in poaching incidents was observed in India (Gomez et al., 2021).

As a novel threat, capturing bear cubs (likely after killing the mother) for street performance as “dancing bears” could be one of the reasons for the decreasing population trend of sloth bears in India, though, in the recent past, this illegal activity has been dropped down significantly (D’Cruze et al., 2011).

Sloth bear comes under Schedule I of the Wildlife (Protection) Act, 1972, in India and globally included in Appendix I of CITES (Convention on International Trade in Endangered Species of Wild Fauna and Flora). The global population trend of sloth bears is decreasing; hence, it has been included under the category “Vulnerable” by the IUCN (International Union for Conservation of Nature and Natural Resources) for the past three decades (Dharaiya et al., 2016).

1.3 Justification of the study

It has become evident that the lack of long-term studies related to the ecology and sloth bears’ conservation throughout their distributional range has hindered drawing inferences on large-scale assessment. Conservation planning of any species requires prior in-depth knowledge of the ecology and behaviour of the concerned species. Despite its widespread distribution and vulnerable conservation status, the sloth bear is one of the least studied bear species globally. In this view, Sanjay Tiger Reserve (STR), Madhya Pradesh, a human-dominated landscape with sub-optimal habitat conditions which comes under Level III Tiger Conservation Landscape (TCL)

(Wikramanayake et al., 2011), was chosen to study the aspects of the ecology of sloth bear. I conducted the fieldwork for this study for three years (2017-2019). This study was a part of the (then) ongoing project related to tiger ecology and conservation conducted by the Wildlife Institute of India, and it broadly follows the methodology of All India Tiger Estimation (Jhala et al., 2015).

1.4 Objectives of the study

The broad objectives of this study were:

1. To estimate the occupancy and abundance of sloth bears and identify environmental and anthropogenic factors which influence the same
2. To assess the space use patterns by sloth bears in terms of habitat use and den selection
3. To investigate the seasonal diet patterns of sloth bears
4. To assess the human-sloth bear conflict based on spatio-temporal interaction and identification of the conflict risk-prone areas

1.5 Organisation of thesis

The thesis was organised into seven chapters. The general introduction, the biology and conservation of sloth bears were briefly mentioned in chapter 1 (present chapter). This chapter includes the evolutionary history, distribution, various ecological aspects and conservation challenges of sloth bears. Chapter 2 describes the study area, with special reference to its physiognomic characteristics, land use land cover types, forest types, biodiversity and anthropogenic pressures. Chapter 3 deals with the first objective of the study, i.e. occupancy and abundance of sloth bears. Space use patterns of sloth bears were described in chapter 4, which consists of two parts, i.e., habitat use (Part A) and den selection by sloth bears (Part B). Chapter 5 presents an account of the food habits of sloth bears with respect to seasons. In chapter 6, the seasonal spatio-temporal interaction between sloth bears and humans was assessed, and the hotspots of the human-sloth bear conflict were predicted. Lastly, in chapter 7, I summarised the entire work, mentioned management recommendations, and discussed the further scope of research related to the ecology and conservation of sloth bears. Each chapter (Chapters 3,4,5, and 6) consists of an introduction, methodology, results, discussion and conclusion.

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CHAPTER 2: STUDY AREA

2.1 Constitution of Sanjay Tiger Reserve

The Sanjay Tiger Reserve (STR) is situated in the eastern part of Madhya Pradesh (Figure 2.1). The core zone of STR encompasses Sanjay National Park (SNP) and Dubri Wildlife Sanctuary (DWLS) (Table 2.1 & Figure 2.1), which are situated in the Sidhi district. The buffer zone covers parts of the Shahdol Forest Division (Beohari Tehsil of District Shahdol) and the Sidhi Forest Division (Majhauli and Kusmi Tehsil of District Sidhi). STR is situated between 24.125122°N 81.546009°E towards the north-west and 23.822641°N 82.175139°E towards the south-east. The total geographical area of STR includes 812.58 km^2 as a core zone (Table 2.1) and 861.93 km^2 as a buffer zone (Table 2.2), out of which only the core area was considered the intensive study area. Details of forest Ranges, sub-ranges/Circles, forest Beats and Compartments of the core zone were mentioned in Table 2.3.

Table 2.1. Area (in km^2) description of the core zone of Sanjay Tiger Reserve (STR), Madhya Pradesh, India.

Protected Area	**Reserved Forest (km^2)**	**Protected Forest (km^2)**	**Revenue Land (km^2)**	**Total (km^2)**
Sanjay National Park	428.949	11.327	24.367	464.643
Dubri Wildlife Sanctuary	244.583	60.030	43.325	347.938
1.1.1 Grand Total	**673.532**	**71.357**	**67.692**	**812.581**

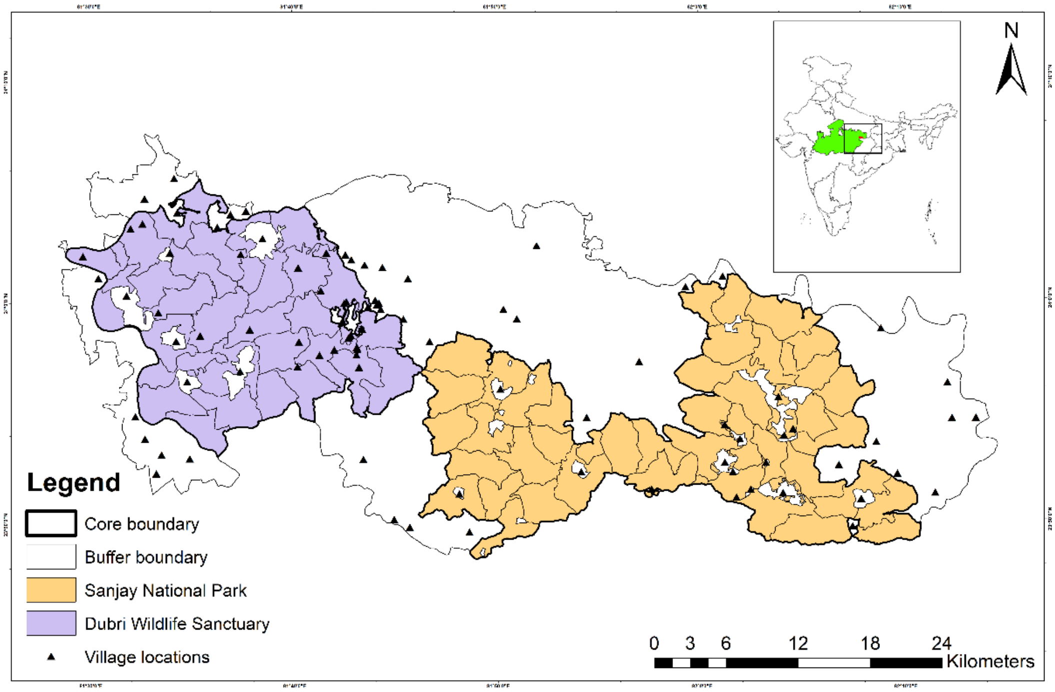

Figure 2.1. Location and administrative map of Sanjay Tiger Reserve, Madhya Pradesh, India.

Table 2.2. Buffer area (Range-wise, in km^2) of Sanjay Tiger Reserve, Madhya Pradesh, India.

Sl. No.	**Name of Range**	**Forest Area (km^2)**	**Revenue Land (km^2)**	**Total (km^2)**
1	Beohari	100.620	114.834	215.454
2	Madwas	198.948	75.719	274.667
3	Tamsar	108.337	109.046	217.383
4	Bhuimad	96.741	57.686	154.427
Total		**504.646**	**357.285**	**861.931**

Table 2.3. Details of compartments, forest Beats, sub-ranges/circles and forest Ranges of the core zone of Sanjay Tiger Reserve, Madhya Pradesh, India.

Sl.No.	**Name of Sub Division**	**Name of Range**	**No. of Sub Ranges/Circles**	**No. of Beat**	**No. of Compartment**	**Total Forest Area (km^2)**	**Revenue Area (km^2)**	**Grand Total**
1	Sanjay National Park	Mohan	03	14	101	264.502	18.555	283.057
2		Pondi	03	10	60	175.774	5.812	181.586
3	Dubri Wildlife Sanctuary	Dubri	02	10	69	159.434	27.831	187.265
4		Bastua	03	11	83	145.179	15.494	160.673
Total			**11**	**45**	**313**	**744.889**	**67.692**	**812.581**

2.2 Vegetation

According to “Revised Classification of Forest type of India” by Champion & Seth (1968), the forest type of the STR comes under sub-group 3C- “North Indian moist deciduous forests”, with subtype (C2) “Moist sal-bearing forest” with subdivision (2e) “Moist peninsular sal forest”. Sal (*Shorea robusta*) forest is a dominant plant community in STR, occupying about 80% of the entire area. This sal forest occurs mainly in the Gondwana system and can grow along with bamboo. Recurring fires, heavy grazing, illicit felling, and lopping appear to be the causal factors that significantly reduced sal regeneration. The sal crop present in STR is primarily middle-aged with scattered matured trees. Apart from sal forest, other vegetation/habitat types are sal mixed, mixed, bamboo mixed and grasslands. I prepared a forest type map (consisting of the forest types mentioned above) based on the supervised classification technique (Figure 2.2), which was detailed in Chapter 3.

1. **Sal and sal mixed forests:** Sal is a dominant plant species in the SNP and DWLS. Sal is considered a climax species on low undulating plains with loamy soils and forms more pure crops than its associates. The proportion of sal varies widely, with some patches bearing ≥80% of sal trees. However, in STR, sal is mainly associated with other (miscellaneous) tree species common in the central Indian landscape. If a forest patch consists of at least 50% miscellaneous species of trees and the remaining proportion (50%) of sal, it would be identified as a sal mixed forest type. In STR, sal mixed forest is found in the largest proportions (Table 2.4).
2. **Mixed Forests:** Mixed crop occurs interspersed with sal on lower slopes mainly due to edaphic factors. In STR, frequently found species in mixed forests are tendu (*Diospyros melonoxylon*), sedha (*Lagerstroemia parviflora*), mahua (*Madhuca longifolia*), char (*Buchanania cochinchinensis*), dhawa (*Anogeissus latifolia*), aola (*Emblica officinalis*), bharhi (*Chloroxylon swietenia*), palas (*Butea monosperma*), amaltas (*Cassia fistula*), salai (*Boswellia serrata*) etc. Mixed forests are the second most abundant forest types in STR and comprise miscellaneous species of trees mentioned above and sal, in more or less equal proportions.
3. **Bamboo mixed:** Bamboo (*Dendrocalamus strictus*), sal, and other mixed tree species occur mainly in SNP, especially in the Pondi and Mohan range. In DWLS, due to heavy lopping and cutting, the presence of bamboo is rare.
4. **Scrub:** The important species in this habitat type are *Acacia* sp., *Ziziphus* sp. (less abundant), dhawai (*Woodfordia fruticosa*) etc. Along with these species,

the occurrence of the *Lantana camera* is also common in some parts of STR. Scrubs are mainly found near the habitations and western parts of STR.

5. **Grasslands:** Grassland is the least found habitat/forest type in STR. These grasslands are under the constant pressure of livestock grazing. However, a few grassland patches are created and managed as a habitat restoration tool at relocated village sites in the core zone. Commonly found species of grasses are lampa (*Heteropogon contortus*), bhurbhusi (*Eragrostis tenella*), bhor (*Themeda quadrivalvis*), doob (*Cynodon dactylon*), bahari (*Thysanolaena latifolia*) etc.

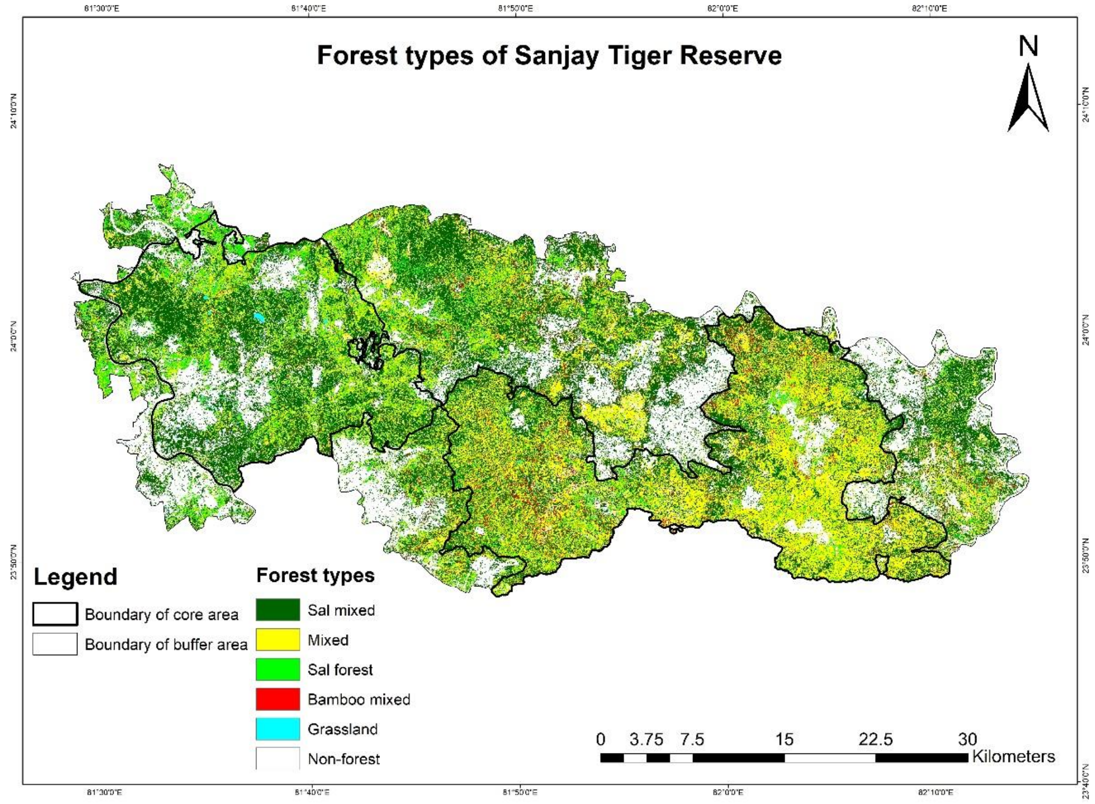

Figure 2.2. Map showing different forest types (n=5) of Sanjay Tiger Reserve, Madhya Pradesh, India.

Table 2.4. Area availability of different forest types in Sanjay Tiger Reserve, Madhya Pradesh, India.

Sl. No.	**Forest type**	**Area (km^2)**
1	Sal mixed	538.06
2	Mixed	457.54
3	Sal forest	150.03
4	Bamboo mixed	55.65
5	Grassland	7.28

Land use and land cover types: In STR, I identified six land use land cover (LULC) classes based on the supervised classification of satellite imagery (Figure 2.3), which was detailed in Chapter 3. Dense forest occurs in the highest proportion, followed by open forest, agricultural land, barren land, scrubland and water bodies (Table 2.5).

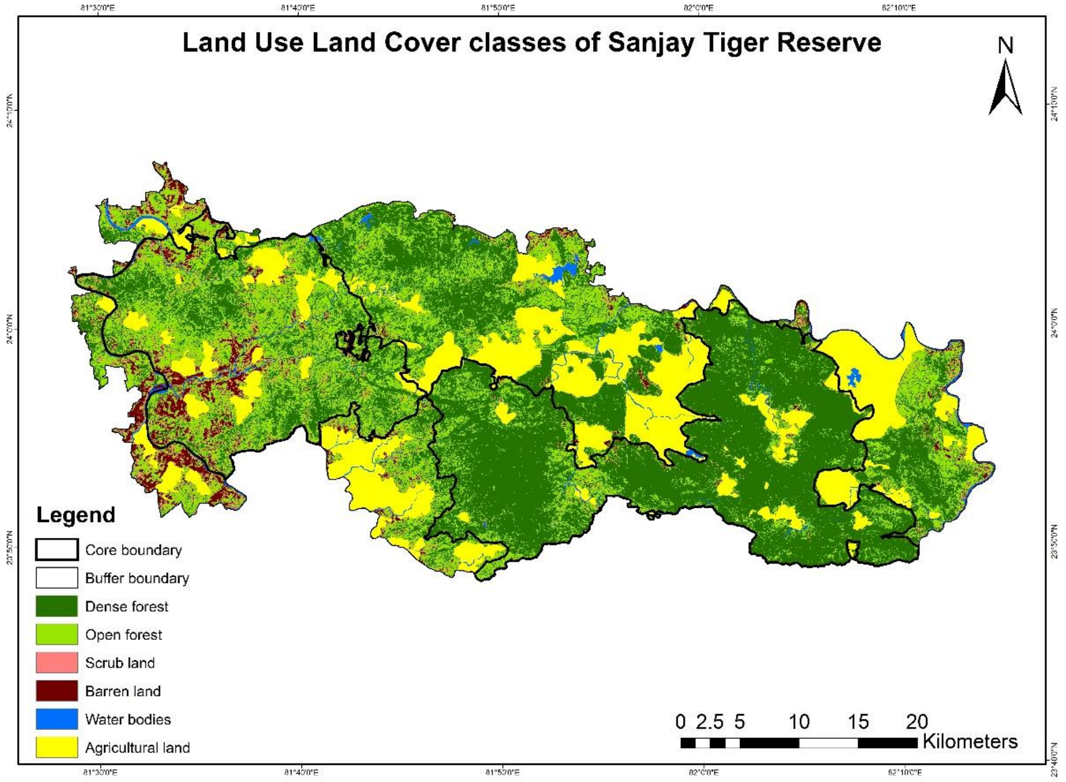

Figure 2.3. Map showing Land Use Land Cover (LULC) classes (n=6) of Sanjay Tiger Reserve, Madhya Pradesh, India.

Table 2.5. Area availability of different Land use land cover (LULC) classes (n=6) of Sanjay Tiger Reserve, Madhya Pradesh, India.

Sl. No.	**LULC classes**	**Area (km^2)**
1	Dense forest	706.66
2	Open forest	447.20
3	Scrubland	41.81
4	Barren land	66.77
5	Agricultural land	303.25
6	Water bodies	37.95

2.3 Topography and geology

The general topography of STR varies from hilly and undulating to moderate slope and plain areas. The terrain of SNP is characterised by hilly and rugged along with steep gorges cutting through the plateaus. Unlike SNP, the topography of DWLS and the eastern part of STR is mostly plain interspersed with gentle undulating areas. The slope varies between 0° to 61°, with a steeper slope towards the northeastern and eastern parts of STR (Figure 2.4).

The elevation of STR varies from 239m to 770m above mean sea level (MSL), with higher elevated eastern part (SNP and adjacent buffer) and lower elevated western part (DWLS and adjacent buffer) (Figure 2.5).

The local geological sequence of STR is the Mahadeva series of Gondwana formations. The typical Mahadeva beds are considered ferruginous and often break into small pieces. The rocks in STR are quartzite, sandstone, mica schists, horn blends, schist shales and limestone (least frequent). These rocks give rise to immature soils of less variety and fertility.

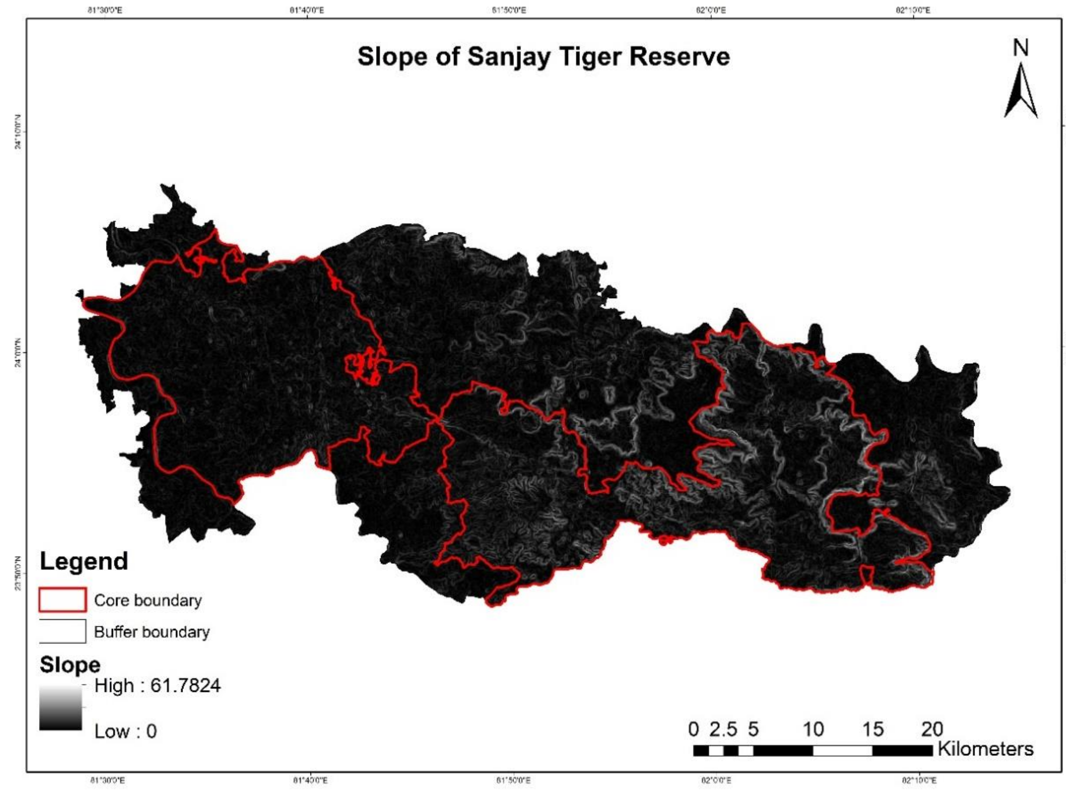

Figure 2.4. Map showing slope (in degree) of Sanjay Tiger Reserve, Madhya Pradesh, India.

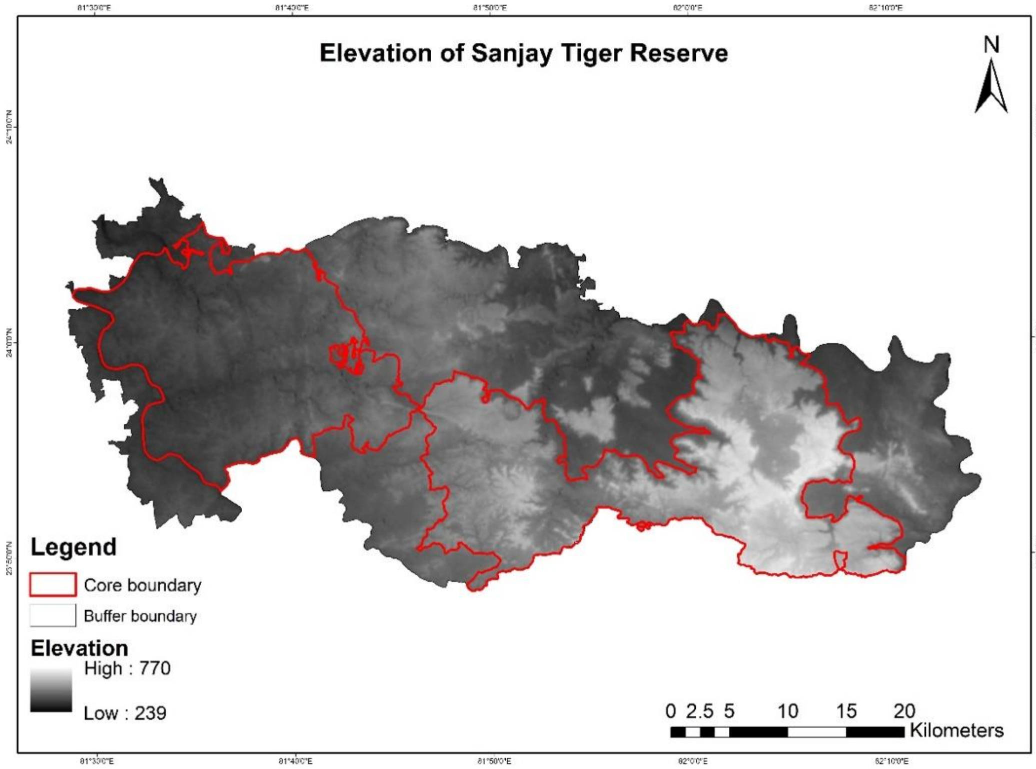

Figure 2.5. Map showing elevation (in metre) range of Sanjay Tiger Reserve, Madhya Pradesh, India.

2.4 Climate

Data loggers (n=4; HOBO U23 Pro v2) were deployed from 2017 to 2018 in four forest Ranges of the core zone of STR to record the daily temperature and relative humidity. Based on the result obtained from data loggers, two different seasons in STR were considered: summer and winter. Although three distinct seasons (summer, monsoon and winter) exist in STR, the classification of seasons was restricted to only summer and winter, as the entire data collection was conducted during these seasons. I categorised summer and winter if the daily mean air temperature was >20°C and <20°C, respectively. Thus, March to June was classified as summer and December to February was classified as winter. The average minimum and maximum temperature varied between 13°C to 24°C in winter and 25°C to 37° in summer (Figure 2.6). Several dust squalls generally characterise the onset of monsoon with high-velocity winds or storms. The mean annual rainfall of STR is 1303.0 mm. The usual fluctuations of the mean annual rainfall are expected to range between 1059.70 mm. to 1277.10 mm. However, the mean annual rainfall was 873.5mm during the study period (2017-2018), and an average annual rainfall of 1045.56mm was recorded for five years (2016-2020) (Available at Customized Rainfall Information System, India Meteorological Department). Rainfall (month-wise) varied between 0 to 439mm during 2017-2018, with heavy rain (>50mm) from June to September and less rainfall (≤20mm) during winter between January to February (Figure 2.7). The river banks, valleys and other low-lying depressions generally experience frost during severe cold.

In STR, the relative humidity varies between 72% to 95% in monsoon, 46% to 83% in winter and 24% to 55% in summer (Figure 2.8). The relative humidity is higher in plateaus, ridges, high elevations, and areas close to perennial water sources than in open and plain lands.

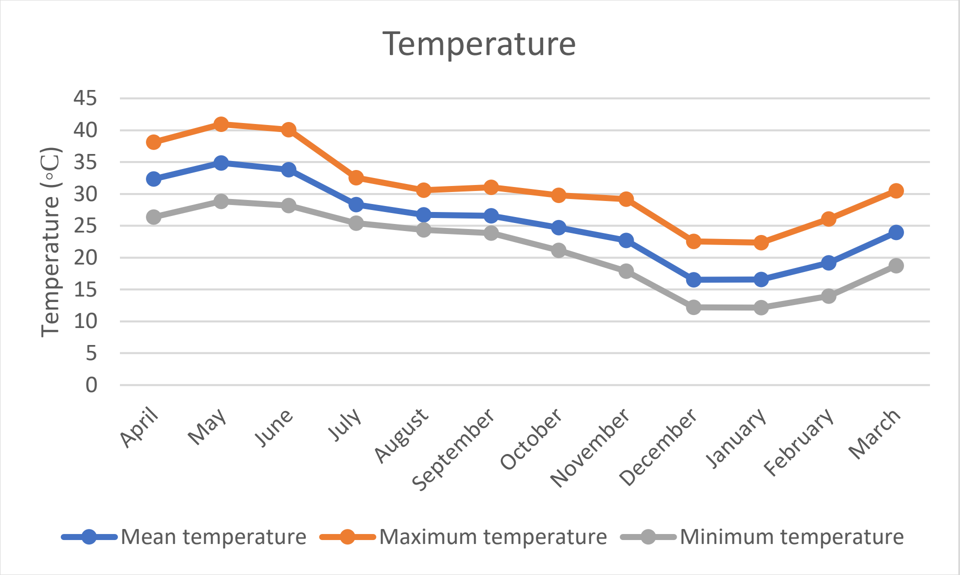

Figure 2.6. Month-wise mean temperature (◦C), obtained from data loggers (n=4) from 2017 to 2018 in Sanjay Tiger Reserve, Madhya Pradesh, India.

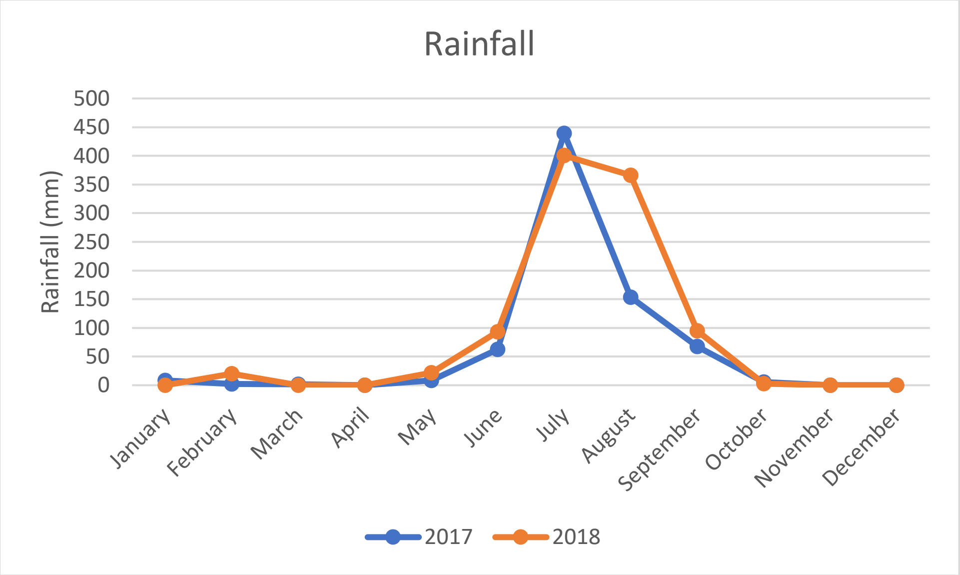

Figure 2.7. Month-wise mean rainfall (in millimetres) from 2017 to 2018 in Sanjay Tiger Reserve, Madhya Pradesh, India. (Source: IMD)

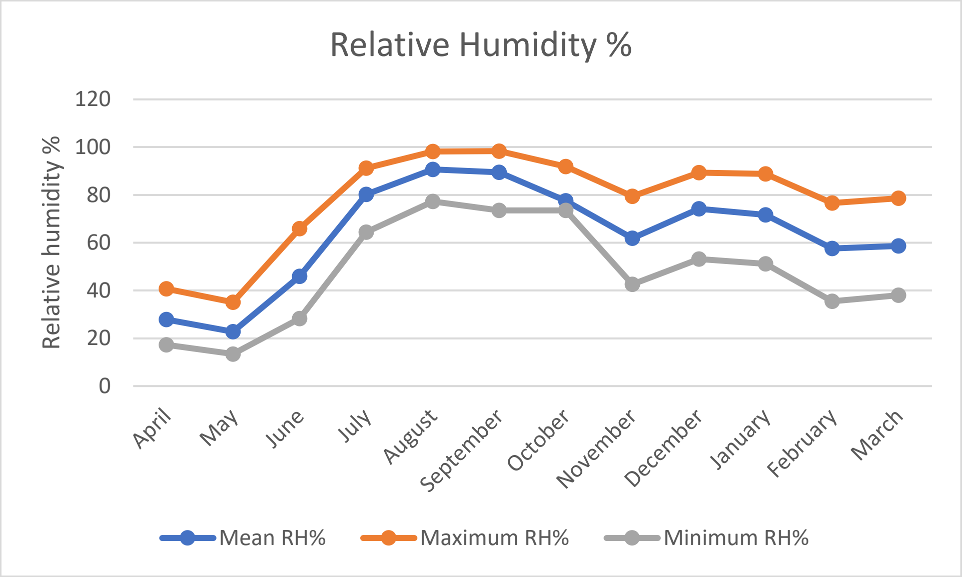

Figure 2.8. Month-wise mean relative humidity (RH) (in %) from 2017 to 2018, obtained from data loggers (n=4) in Sanjay Tiger Reserve, Madhya Pradesh, India.

2.5 Water source

The plain area of STR lies in the catchments of several rivers/streams such as Banas, Gopad, Son, Mawai, Kodmar, Umrari and their tributaries. However, major perennial rivers flow along the periphery of the core zone. Network of seasonal (water lasts only for 4-6 months after monsoon) and perennial nullas also exist in SNP and DWLS. Water shortage becomes particularly acute in those years when the wet season finishes much early and when the total rainfall of the year is much less than the optimum. Apart from the natural water sources, there are also artificial water bodies which the forest department has created to ensure water availability uniformly throughout the Tiger Reserve, especially during pinch periods. However, most artificial (perennial) water bodies are located at DWLS. There is a need to create and manage artificial water bodies at SNP, as water availability in these areas is insufficient during summer.

2.6 Fauna

STR harbours a diverse assemblage of fauna. The major carnivores of the STR are the tiger (*Panthera tigris*), leopard (*Panthera pardus*), hyena (*Hyaena hyaena*), sloth bear (*Melursus ursinus*), Indian wild dog (*Cuon alpinus*), Indian wolf (*Canis lupus*), golden jackal (*Canis aureus*), Indian fox (*Vulpes bengalensis*), jungle cat (*Felis chaus*), Asian palm civet (*Paradoxurus hermaphroditus*), small Indian civet (*Viverricula indica*), ruddy mongoose (*Herpestes smithii*), Indian grey mongoose (*Herpestes edwardsii*) and ratel or honey badger (*Mellivora capensis*). In addition to this, Asiatic wild cat (*Felis silvestris*) and rusty-spotted cat (*Prionailurus rubiginosus*) were also reported from STR in the recent past (Chaudhuri et al., 2021). Seven species of wild ungulates, such as chital (*Axis axis*), nilgai (*Boselaphus tragocamelus*), sambar (*Rusa unicolor*), four-horned antelope (*Tetracerus quadricornis*), chinkara (*Gazella bennettii*), barking deer (*Muntiacus muntjak*), and wild pig (*Sus scrofa*), are found in STR. Indian crested porcupine (*Hystrix indica*) and black-naped hare (*Lepus nigricollis*) are the two large-bodied rodent and lagomorph, respectively, found in STR. Two species of primates, i.e. grey langur (*Semnopithecus entellus*) and rhesus macaque (*Macaca mulatta*), are found in STR. Based on a camera trap survey conducted in 2018 (part of the All India Tiger Estimation program), 30 species of animals (mammals, birds and reptiles) were found in STR (Jhala et al., 2020). STR comes under the level III Tiger Conservation Landscape (TCL) (Wikramanayake et al., 2011), characterised by sub-optimal habitat for tigers, co-predators and prey. Five unique individual tigers were photo-captured in STR, which yielded a density of 0.23 tigers/100km^2 (Jhala et al., 2020).

2.7 Anthropogenic pressure

There were 39 villages inside the core area of STR during the present study period, with a sizable human population and substantial livestock holding. Approximately 81,259 people (54/km^2, 132 villages, and 19,076 households; Census, 2011) and ~1,40,044 livestock (93/ km^2) live in and around STR. Sidhi is one of the poorest districts, as indicated by the multidimensional poverty index (MPI; 52.68% of the total population are multidimensionally poor; NITI Aayog, 2021). Hence, people living in and around STR are expected to rely more on forest resources for their sustenance (Reddy & Chakravarty, 1999; Sunderlin et al., 2005).

People in and around STR primarily belonged to tribal (Baiga, Gond, Kol) and Yadav communities (Jhala et al., 2011). The increasing human population exerts severe pressure on forest resources, causing habitat degradation. The grazing pressure of livestock inside the core zone (and buffer) is also intense, which inadvertently affects the habitat quality, resulting in poor habitat with the invasion of weeds, competition with wild ungulates and an increasing chance of disease transmission. However, shortly after the study period, seven villages (Suheera, Chafal, Padwar, Saidol, Dudhmania, Hatibandha and Baherwar) were successfully relocated from the core area of STR.

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CHAPTER 3: OCCUPANCY AND ABUNDANCE OF SLOTH BEAR

3.1 Introduction

Knowledge of the spatial distribution and abundance of a species is fundamental for ecological studies and a prerequisite for successful conservation planning and decision-making processes in the field of conservation biology (Kremen et al., 1994; Gaston and Rodrigues, 2003; Rondinini et al., 2005; Nichols and Williams, 2006; McCarthy and Possingham, 2007; Puri et al., 2015). Correctly identifying biotic and abiotic factors could be crucial to understanding the occurrence and estimating the population abundance reliably (Meents et al., 1983). In many relatively widespread mammal species, baseline information on their distribution pattern is scanty, which may have hindered devising conservation planning, especially in human-dominated landscapes (Mace et al., 2008; Hoffmann et al., 2010). Moreover, inaccurate abundance estimates or estimates based on empirical studies can lead to erroneous management decisions and undermine the effort to conserve the concerned species in its natural environment. However, estimating the abundance of cryptic species through the traditional capture-mark-recapture (CMR) method is labour-intensive and not always logistically feasible. Exploration and subsequent implementation of cost-effective methodologies to monitor population abundance are thus required, given the limited resource availability in the field of wildlife conservation (Parker et al., 2011). Hence, before conducting any relatively logistically expensive population estimation survey, baseline information regarding the distribution of the concerned species in a cost-efficient way could be useful (Jones, 2011).

Sign and interview-based surveys are the two most commonly used methods to assess the distribution and abundance of individually unidentifiable (or unmarked) solitary carnivores in conservation planning (Garshelis et al., 2022). However, with the advent of science and technology, passive detectors, especially automated camera traps, have become indispensable in wildlife and conservation biology (O'Connell et al., 2011; Burton et al., 2015). Despite the diverse applications of camera traps, their use has been confined to population estimation of individually identifiable or marked animals (Burton et al., 2015). Camera trap surveys have been successfully used to estimate population density through spatial capture-recapture (SCR) models for marked animals (Karanth and Nichols, 1998; O'Connell et al., 2011). However, population estimation for unmarked or individually unidentifiable animals via camera trapping has posed challenges to ecologists and conservation biologists (Gilbert et al., 2021). Also, for unmarked species like Asian bears, population monitoring in a small area (<10,000km^2) requires spatially explicit methods (e.g., camera traps or genetics), but these studies remain restricted due to resource and logistical constraints (Proctor et al., 2022). Hence, relatively cost-efficient sign surveys and expert-based knowledge are frequently relied upon to produce distribution maps of the Asian bear species but are subjected to erroneous results if the objective and sampling design of the study are not clearly defined (Garshelis et al., 2022). Without the ground verification following a scientific study design, extrapolation of the distribution and abundance based on the anecdotal perceptions of local people can be misleading (Moqanaki et al., 2018; Pokharel et al., 2022; Proctor et al., 2022).

Occupancy-based distribution is one of the widely used frameworks to understand the spatial patterns of species occurrence at a landscape scale (Jones, 2011; Karanth et al., 2011) and is often used prior to the abundance estimation of the focal species (Jhala et al., 2015). For surveys conducted at individual Protected Areas (PAs), understanding focal species' occupancy at the administrative level could be crucial for managers to gain baseline knowledge, as commonly used grid-based studies might not always be interpretable for management interventions (Jhala et al., 2008; Pillay et al., 2011, 2014; Majgaonkar et al., 2019; Srivathsa et al., 2020). Similarly, camera trap-based photographic capture rates (number of individuals photo-captured/effort in terms of trap-nights) or relative abundance indices are often used as surrogates of population abundance (Carbone et al., 2001; O'Brien et al., 2003; Bengsen et al., 2011). However, inferences drawn from these index-based surveys have been criticised due to the implicit assumption of equal detection probability (Jennelle et al., 2002; Harmsen et al., 2010; Sollmann et al., 2013; Burton et al., 2015). Rowcliffe et al. (2008) proposed the random encounter model (REM) to estimate density by incorporating information on animal movement, photographic capture rate and the detection zone of camera traps. Despite its unique and robust approach, the application of REM is limited due to the true random deployment of camera traps and the inability to account for spatial variation of abundance. Also, the need to extrapolate density from the camera trap detection zone to the entire surveyed area, and the lack of knowledge regarding the movement of elusive species, further restricts its applicability (Gilbert et al., 2021). The application of SCR frameworks for unmarked individuals, otherwise known as unmarked spatial capture-recapture (USCR; Chandler & Royle, 2013), is another promising approach to estimating population density.

Nonetheless, its limited applicability is due to the underlying computationally intensive (Bayesian) framework (Royle et al., 2014; Gilbert et al., 2021), tendency to produce imprecise estimates (Augustine et al., 2019) and judicious selection of priors on the spatial scale of animal detection at camera traps (Sun et al., 2014). On the other hand, site-structured models such as the Royle-Nichols abundance-induced heterogeneity model (RN model; Royle and Nichols, 2003) and the N-mixture model (Royle, 2004) have been widely used to draw inferences on abundance from detection-nondetection data and count data, respectively, based on spatially or temporally replicated surveys. In the recent past, the latter was more widely applied to estimate the abundance of terrestrial mammals from count data (Keever et al., 2017; Xiao et al., 2018; Kafley et al., 2019; Kidwai et al., 2019; Penjor et al., 2019; Searle et al., 2020). The major advantages of the N-mixture model are its cost-effectiveness, the ability to bypass individual identification, and the ability to account for spatial variation of abundance as a function of covariates. The N-mixture model typically requires the following assumptions: population closure at the site, equal probability of detection of all individuals at a site, no false-positive errors (i.e., misidentification or double-counting of individuals), and independent detection of individuals across the sampling units (Royle, 2004). However, N-mixture models are sensitive to assumption violations, especially false-positive errors, which are not commonly addressed (Kéry and Schaub, 2011). For free-ranging unmarked species, one individual would likely be photo-captured several times during the entire sampling occasion (O'Brien et al., 2003), which can overestimate abundance (Link et al., 2018; Nakashima, 2020). But, relaxing the assumption of population closure at the site for free-ranging species, it is justifiable to interpret the true abundance of a site as "relative abundance," which translates to the number of individuals utilising a site at a given time (Kéry & Royle, 2015).

The sloth bear has a broader geographic distribution than the other three ursids found in the Indian sub-continent (Johnsingh, 2003; Seidensticker et al., 2011). Yet, limited information is available regarding their distribution or occupancy with respect to fine-scale spatial covariates, as most of the previous studies relied on extrapolation from the opinions of experts and locals without further field verification (Garshelis et al., 1999b; Yoganand et al., 2006). Furthermore, information on the population abundance of sloth bears based on a rigorous statistical framework is notably lacking (Dharaiya et al., 2016). The range-wide population abundance of the sloth bear was estimated to be between 10,000 and 20,000 by several authors (Garshelis et al., 1999a; Chauhan, 2006; Yoganand et al., 2006; Sathyakumar et al., 2012). However, none of these estimates was reliable enough to monitor the population trends because of the differential data collection methods and types of data used in the analyses (Sathyakumar et al., 2012; Dharaiya et al., 2016). The sloth bear often conflicts with humans in some parts of its distributional range (Rajpurohit and Krausman, 2000; Bargali et al., 2005; Garcia et al., 2016; Debata et al., 2017; Dhamorikar et al., 2017; Singh et al., 2018). One recent study (Gomez et al., 2021) also indicated a steadily increasing trend of sloth bear mortality due to poaching. Thus, the ever-increasing human population, along with subsequent habitat fragmentation, poaching, and human-sloth bear conflicts, are significant concerns for conserving this species, necessitating the need for reliable abundance estimation of sloth bears. So far, only one study (Garshelis et al., 1999a) has attempted to make robust density estimates of the sloth bear, which was carried out in Royal Chitwan National Park (henceforth Chitwan), Nepal, based on a spatial mark-resight method. Reliable information on population abundance and identifying factors influencing abundance are crucial for the long-term conservation of sloth bears in human-dominated landscapes.

In this context, firstly, I conducted an occupancy survey to gain preliminary knowledge of the distribution of sloth bears in STR. Secondly, I explored the application of site-structured models with special reference to the N-mixture model (Royle, 2004) to quantify the relative abundance of sloth bears in STR.

I had three objectives which are as follows:

1. To generate baseline information on the occupancy of sloth bears in STR as a function of fine-scale spatial covariates
2. To demonstrate the use of the N-mixture model in estimating the relative abundance of sloth bears and compare the mean site abundance estimates obtained from the N-mixture model to those of the RN model
3. To identify the environmental and anthropogenic covariates that govern the relative abundance of the sloth bear.

3.2 Materials and methods

Carnivore sign survey

I conducted a carnivore sign survey from December 2016 to January 2017 in the core area of STR. Each forest Beat was chosen as a sampling unit, which I believe would be helpful for managers to understand the overall distribution of sloth bears across the administrative units of Tiger Reserve (and other forest divisions) from a cost-efficient and rapid survey (Figure 3.1) The average size of each forest Beat in the core area of STR was 12.87 km^2, which corresponded to the female sloth bear's home range size in central India (Yoganand, 2005). To maximise the detection of bear signs, forest roads, trails, and dry riverbeds falling under each Beat boundary were intensively walked (15km/Beat) by other trained field personnel and me, following the protocol of All India Tiger Estimation programs (Jhala et al., 2015). Sloth bears leave conspicuous signs such as footprints, scats, dig marks and claw marks, although the disintegration rate of those signs differs significantly. Only fresh signs (< 7 days old) were recorded while walking the sign survey to ensure seasonal temporal independence and avoid over-fitting the data. An attempt was made to cover an entire Beat within a single day by dividing three spatial replicates (5km each) and three independent field personnel teams to ensure spatial independence. However, being a free-ranging carnivore, an individual sloth bear detected at a Beat could also be detected at the adjacent Beat. Hence, estimates obtained from the occupancy analysis would be interpreted as "probability of site use" instead of true occupancy (MacKenzie et al., 2017). Each spatial replicate was further divided into 1km segments, following Hines et al. (2010). I also collected fine-scale spatial covariates relevant to sloth bears' ecology and easily collectable from a rapid survey. At each 500m interval of a segment, the number of fruiting trees which contributed $\geq$1% of the diet of sloth bears (Bargali et al., 2004; Yoganand, 2005; Ramesh et al., 2012; Rather et al., 2020) and termite mounds were counted around 10m-radius plot. Apart from these covariates, information regarding the number of fresh tree cutting, lopping and livestock presence (number of dung/pellets) were also recorded at those 10m-radius plots. Due to the unequal effort (in km) across Beats of different sizes, the area of the Beat, total segment length/Beat and segment length/area of the Beat were considered as detection covariates. I averaged the value of each covariate at every Beat to analyse further. The details of the covariates are presented in Table 3.1.

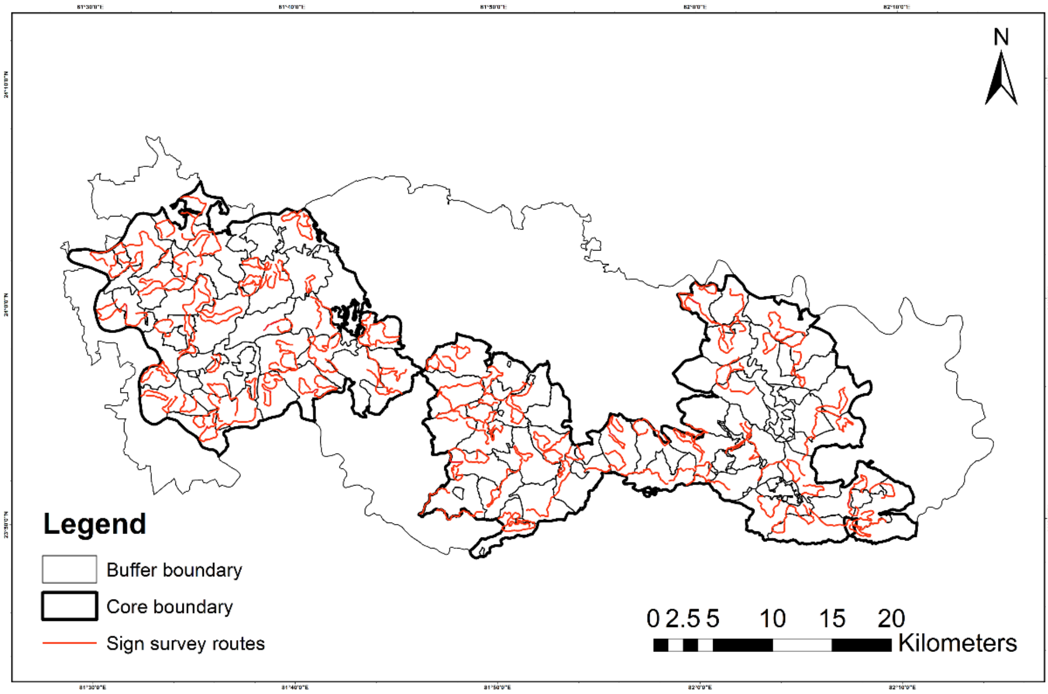

Figure 3.1. Routes of carnivore sign survey which was conducted at each forest Beat to estimate the occupancy (probability of site use) of sloth bears in the core zone of Sanjay Tiger Reserve, Madhya Pradesh, India, from December 2016 to January 2017.

Table 3.1. List of covariates to model the probability of site use by sloth bears using the single-season single-species occupancy model in Sanjay Tiger Reserve, Madhya Pradesh, India, during 2016-2017.

Covariates	**Description**	**Range of values**
Fruiting trees (#/total effort in km)	Beat-wise encounter rate of fruiting trees, i.e. total number of fruiting trees recorded from 10m-radius plots at every 500m interval of the segment(s), divided by the total effort (total segment length in km)	4.90-25.47
Termite mounds (#/total effort in km)	Beat-wise encounter rate of termite mounds, i.e. total number of termite mounds recorded from 10m-radius plots at every 500m interval of the segment(s), divided by the total effort (total segment length in km)	0.40-4.87
Lopping (#/total effort in km)	Beat-wise encounter rate of fresh lopping signs, i.e. total number of lopping signs recorded from 10m-radius plots at every 500m interval of the segment(s), divided by the total effort (total segment length in km)	0-2.40
Cutting (#/total effort in km)	Beat-wise encounter rate of fresh tree cutting signs, i.e. total number of cutting signs recorded from 10m-radius plots at every 500m interval of the segment(s), divided by the total effort (total segment length in km)	0-4.47
Livestock (#/total effort in km)	Beat-wise encounter rate of livestock dung/pellets, i.e. total number of livestock dung/pellets recorded from 10m-radius plots at every 500m interval of the segment(s), divided by the total effort (total segment length in km)	0-1.4
Area of the Beat (in km^2)	Area of each forest Beat, where carnivore sign survey was conducted	6.22-24.28
Segment length /Beat (in km)	Total effort in km which was covered at each forest Beat during carnivore sign survey	5-15
Segment length/area of the Beat	Total effort in km which was covered at each forest Beat during carnivore sign survey, divided by the area of the forest Beat	0.34-2.41

Occupancy analysis

Following MacKenzie et al. (2002) and Hines et al. (2010), I constructed detection history by assigning a score of either '1' if the sloth bear sign was recorded or '0' if there was no sign from a segment (1km length), which translated to a detection history of 15 spatial replicates for each Beat. The correlated occupancy model (Hines et al., 2010) was fitted, which is likely to be suitable for long-ranging species like sloth bears and detecting its signs from predetermined survey routes. The correlated occupancy model (Hines et al., 2010) estimates two additional parameters apart from occupancy/probability of site use (Ψ) and detection probability (p) when the sampling unit is occupied by the concerned species, which are as follows:

θ: the probability of the species being present at a segment given the previous segment was not occupied by the species

θ': the probability of the species being present at a segment given the previous segment was occupied by the species

However, given the sampling method and sloth bears' distribution in STR, I expected issues regarding model convergence. To overcome this issue, I constructed auto-correlated models without any covariates to identify the appropriate segment length, at which the detection of bears at a segment would be considered independent from the preceding segment, following Thorn et al. (2011). At that segment length, it is expected that $\theta \approx \theta'$, which would indicate the independent detections of bears across the segments and subsequently allow me to construct standard single-season single-species occupancy models (MacKenzie et al., 2002) by collapsing the detection history to that segment length (Thorn et al., 2011).

Next, candidate models of single-season single-species occupancy were constructed with both detection and occupancy covariates. Firstly, the Ψ was held constant, and p was modelled with both detection and fine-scale spatial covariates. After determining the best detection covariate(s), Ψ was modelled with the fine-scale spatial covariates. Model selection was carried out by the Akaike Information Criterion (AIC) value, and I considered models as most parsimonious if they were ≤2 ΔAIC units (Burnham and Anderson, 2002). The relative importance of models and effects of covariates were assessed based on the AIC model weights and significant (p-value<0.05) covariate relationship with p or Ψ. The model averaging was not considered in the case of a single top-ranked model or the presence of uninformative parameters in the top-ranked models (Arnold, 2010). Parametric bootstrapping was performed to evaluate the model fit (best-selected models) (MacKenzie and Bailey, 2004). All covariates were Z-transformed before analyses, and I performed Pearson’s correlation tests to avoid multi-collinearity. The pairwise Pearson’s correlation coefficient threshold was set to ±0.7 (Dormann et al., 2013), and those covariates (with correlation coefficient ≥ ±0.7) were not modelled together. The program Presence (Version 2.13.11) (Hines, 2006) was used for occupancy analyses.

Camera trap survey

The camera trapping survey was conducted from December 2016 to April 2017 (hereafter, 2017) and January 2018 to April 2018 (hereafter 2018) in the core area of STR. The study area was divided into 2km × 2km grids (henceforth, sites) following the All India Tiger Estimation protocol (Jhala et al., 2015; Figure 3.2). A total of 143 and 144 locations (one at each site) were selected for the deployment of camera traps during 2017 and 2018, respectively (Figure 3.2). The entire core area was divided into two blocks (SNP and DWLS), and the camera trapping survey was carried out one block at a time due to the limited availability of camera traps. A pair of motion-triggered camera traps (Cuddeback C1 and Cuddeback Ambush) were deployed at each site opposite to one another to increase detection (Pease et al., 2016; O'Connor et al., 2017) at an approximate height of 30-40cm above the ground. I placed the camera traps on forest roads, animal trails and dry riverbeds to maximise the photo-capture of sloth bears and other large carnivores (Karanth et al., 2011). The average distance between two adjacent camera trap locations was 2km. The time-lapse between two consecutive photographs was fixed at 15 seconds. Cameras were active 24 hours a day for an average trapping period of 42 days (range: 13-45 days) and checked once every 20-25 days to change batteries and memory cards. The location of each camera trap was collected by using Garmin e-Trex handheld GPS units (Garmin Inc.).

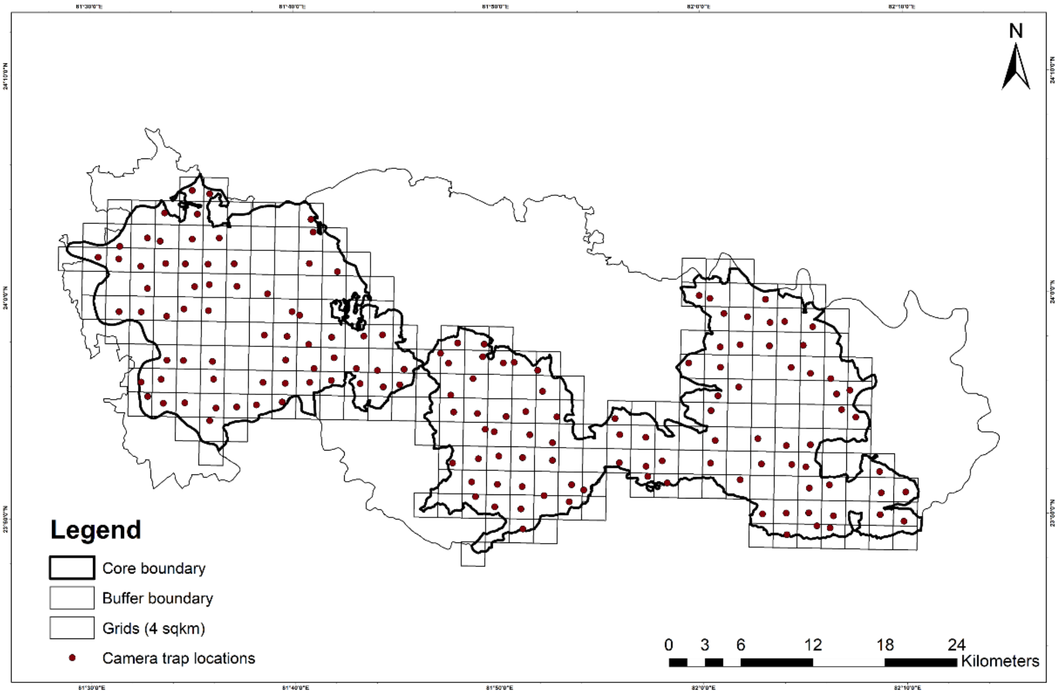

Figure 3.2. Location of camera traps which were deployed at 2×2 km^2 grids during the camera trap survey in the core zone of Sanjay Tiger Reserve, Madhya Pradesh, India, from 2016 to 2018.

To avoid pseudoreplication, I considered two consecutive photographs of sloth bears and humans captured ≥30 minutes apart as independent events (O'Brien et al., 2003; Xiao et al., 2018; Kafley et al., 2019). An interval of 30 minutes was chosen for humans as I expected a substantial number of repeated photo captures of the same individuals (single and groups) within a short time due to feeling intrigued after they detected the camera traps. The temporal sampling unit (sampling occasion) was set to three days, and the count of independent photographs of sloth bears from each site was pooled for each unit of three consecutive days, which yielded an encounter history of 15 sampling occasions. Reducing the sampling period by pooling data from a specific time (days) helped reduce overdispersion (Penjor et al., 2019).

Covariates

Local-scale spatial covariates

Habitat features were measured at each camera trap location within a 10m-radius plot. The number of fruiting trees (≥2m in height and ≥1m girth at breast height) was recorded, which comprises ≥1% of the diet of sloth bears (Ramesh et al., 2012). I also recorded the number of termite mounds and woody shrub species in every 10m and 5m-radius plot, respectively. The photo-capture rate of humans (number of photographs of humans at each location/number of days each camera trap was active) was calculated as one of the surrogates of anthropogenic pressure at each site. Details of local-scale covariates are provided in Table 3.2.

GIS-based covariates

The Land Use Land Cover (LULC) layer for STR was prepared by processing 30m spatial resolution LandSat-8 images (Image ID: LC81430432016297LGN00) dated 23 October 2016 in ERDAS IMAGINE 9.2 (Leica Geosystems) following a standard hybrid image classification protocol (Ruppert et al., 1997). The images were classified into six classes, i.e. dense forest, open forest, scrubland, agricultural land, barren land, and water bodies. The average overall accuracy of the resultant classified images was 88.89%, and the overall Kappa coefficient was 0.85. To classify the forest types in STR, I processed 30m spatial resolution LandSat-8 images (Image ID: LC81430432018110LGN00) dated 2 May 2018, following the methodology mentioned above and supported by 412 ground points. The classes included sal mixed, mixed, sal forest, bamboo mixed, grassland and non-forest. The categorisation of different forest types was based on the broad classification by Champion and Seth

(1968). An accuracy assessment of the resultant classified images was performed based on 392 ground points collected throughout the study area. These classified images' average accuracy and overall Kappa coefficient were 64.30% and 0.45, respectively.

Other covariates used in the analyses were the terrain ruggedness index (RI; Riley, 1999) and Euclidean distance-based covariates (water source, metal road and village; see Table 3.2 for details). All analyses were carried out in ERDAS IMAGINE 9.2 (Leica Geosystems) and ArcGIS 10.2 (Environmental Systems Research Institute).

Table 3.2. List of covariates used to model the relative abundance of sloth bear using the N-mixture model in Sanjay Tiger Reserve, Madhya Pradesh, India, from 2016 to 2018.

A. Environmental covariates (with abbreviations)	**Description**	**Range of values**	**Area**
Land use land cover (LULC) classes	LULC was classified into six classes: dense forest, open forest, scrubland, barren land, agricultural land and water body. The area (in km^2) of each class was calculated and extracted for each 12.40 km^2 grid in Arc GIS 10.2.	1 - 6 (Number of classes)	Dense forest- 706.66 km^2 Open forest- 447.20 km^2 Scrubland- 41.81 km^2 Barren land- 66.77 km^2 Agricultural land- 303.25 km^2 Water body- 37.95 km^2
Forest type classes	Forest type map was classified into six classes: sal mixed, mixed, sal forest, bamboo mixed, grassland and non-forest. The area (in km^2) of each class was calculated and extracted for each 12.40 km^2 grid in Arc GIS 10.2.	1 - 6 (Number of classes)	Sal mixed- 538.06 km^2 Mixed- 457.54 km^2 Sal forest- 150.03 km^2 Bamboo mixed- 55.65 km^2 Grassland- 7.28 km^2

			Non-forest- 390.02 km^2
Distance to the water source (km)	Euclidean distance to the nearest water source (natural or artificial) from the centroid of each 12.40 km^2 grid was calculated using the Spatial analyst tool in Arc GIS 10.2.	0 - 15	-
Ruggedness index	Ruggedness Index was generated for each 12.40 km^2 grid from DEM (Digital Elevation Model) by the roughness index tool in the 3D Analyst extension in Arc GIS 10.2.	9.21 - 27.61	-
Availability (or density) of fruiting trees (no./m^2)	The number of fruiting trees was counted per 10m-radius plot around each camera trap location and subsequently divided by the area (in m^2) of each plot.	0 - 0.09	-
Density of shrubs (no./m^2)	The number of woody shrubs was counted per 5m-radius plot around each camera trap location and subsequently divided by the area (in m^2) of each plot.	0 - 1.39	
No. of termite mounds	The number of termite mounds was counted per 10m-radius plot around each camera trap location.	0 - 4	-
B. Anthropogenic covariates (with abbreviations)			
Human photo-capture rate (no./total effort)	The number of total photographs of humans obtained from each camera trap was counted and divided by the total effort (trap-days or trap-nights) of each camera trap	0-11.87	-
Distance to the village (km)	Euclidean distance to the nearest village from the centroid of each 12.40 km^2 grid was calculated using the	0-12	-

	Spatial analyst tool in Arc GIS 10.2.		
Distance to the metal road (km)	Euclidean distance to the nearest metal road from the centroid of each 12.40 km^2 grid was calculated using the Spatial analyst tool in Arc GIS 10.2.	0-14	-

Relative abundance model of sloth bear

The N-mixture model (Royle, 2004; Kéry and Royle, 2015) was developed to estimate the relative abundance of sloth bears in STR within 3.52km × 3.52km grids, as this represents the minimum home range (i.e., 12.40 km^2) of sloth bear in central India (Yoganand, 2005). However, little is known about the sloth bear's home range size in human-dominated landscapes, though studies elsewhere (Joshi et al., 1995; Ratnayeke et al., 2007) have indicated much smaller average home range estimates compared to those of the central Indian landscape (Yoganand, 2005). Since camera traps were deployed within smaller grids (2km × 2km), I used the data from all 1-4 camera traps falling inside each 3.52km × 3.52km grid and considered these as spatial replicates, following Xiao et al. (2018). I used the N-mixture model to estimate the relative abundance of sloth bears from independent photographic captures as a function of environmental and anthropogenic covariates (Table 3.2).

N-mixture models essentially assume population closure (i.e., the number of individuals residing within a site is constant as no emigration or immigration occurs between sites). Due to the individually unidentifiable aspect and long-ranging behaviour of the sloth bear, I believe that the closure assumption was likely to be violated. Hence, the closure assumption was relaxed by changing the interpretation from absolute abundance at a site to the number of individuals ever associated with a particular site during a given period (Kéry and Royle, 2015), otherwise known as relative abundance. The number of independent detections of sloth bear was counted for “R” sites where, i=1, 2, 3,..., R during a sampling occasion of “T”, where t=1, 2, 3,…, T. The observed count (y_{it}) at i sites during t sampling occasions followed a binomial distribution, and the site abundance N_i followed a Poisson distribution (Kéry and Royle, 2015).

Due to the greater flexibility in modelling abundance with covariates and the ability to incorporate other effects (e.g. zero inflation, latent state), Poisson distribution is an integral part of the state process, i.e., abundance (Royle and Dorazio, 2008). On the other hand, binomial distribution accounts for imperfect detection or false-negative errors. The variation of observed counts (y_{it}) is an effect of imperfect detection of the actual (unknown) abundance N_i and its variability among sites. In simple algebraic form, the two processes could be expressed as:

State process: $N_i \sim \text{Poisson}(\lambda_i)$; $\log(\lambda_i) = \beta_0$ + effects of covariates

Observation process: $y_{it} \mid N_i \sim \text{Binomial}(N_i, p_{it})$; $\text{logit}(p_{it}) = \alpha_0$ + effects of covariates

Where N_i is the latent abundance of site i (i=1,…., R), λ_i is the mean abundance of site i, y_{it} is the count of the species at site i and occasion t (t=1,…., T), and p_{it} is detection probability at site i and occasion t.

Studies have indicated that N-mixture models with Poisson and zero-inflated Poisson (ZIP) distributions can produce reliable relative abundance estimates if detection probability is modelled effectively with appropriate covariates (Barker et al., 2018;

Kéry, 2018). I modelled the detection probability with the trapping period as it is suspected that a more extended trapping period would positively affect the sloth bear's detection probability (O'Connor et al., 2017). I also used local-scale spatial covariates (and no GIS-based covariates) likely to affect the detection probability, following Hofmeester et al. (2019). Detection probability would be held constant if none of the covariates was found to be influencing it (Kafley et al., 2019).

After determining the effect of covariates on the detection process, the relative abundance of sloth bears was modelled as a function of environmental (forest types, LULC types, RI, distance to nearest water source, availability of fruiting trees, shrub density and number of termite mounds) and anthropogenic (human photographic capture rate, distance to nearest village and metal road) covariates with three probability distributions, i.e. Poisson, ZIP and negative binomial (NB). Mean values of GIS-based environmental and anthropogenic covariates were calculated and extracted from each of the 12.40 km^2 grids. For local-scale spatial covariates where the number of camera trap locations was more than one, I calculated the average values of each covariate for the 12.40 km^2 grids. Based on the lowest Akaike Information Criterion (AIC) values, the most parsimonious models were selected from the candidate model set (Burnham and Anderson, 2002). Uninformative parameters were reported; however, inferences were drawn only on the models within ≤ 2 ΔAIC units (Burnham and Anderson, 2002) with statistically significant ($P<0.05$) parameters (Arnold, 2010). Model averaging was not considered in the case of a single top-ranked model or the presence of uninformative parameters in the most parsimonious models (Arnold, 2010). However, due to the "good fit bad prediction dilemma" as mentioned by Kéry and Royle, (2015), I further performed Goodness of Fit (GoF) tests with 1000 iterations of bootstrapping for each of the most parsimonious models of corresponding distributions and the overdispersion parameter (ĉ) was calculated (Kéry and Royle, 2015). I proceeded with further residual diagnostics and mapping of residuals if none of them passed the GoF test (when $P<0.05$) for the final checking of model adequacy (Kéry and Royle, 2015). If no specific pattern of lack of fit was found, it would be interpreted as a result of unstructured noise (Kéry and Royle, 2015), which can be corrected by inflating the range (confidence interval) of predicted abundance (Johnson et al., 2010; Kéry and Royle, 2015). All covariates were standardised using Z-transformation before analyses, and subsequently, Pearson's correlation tests were performed to avoid multi-collinearity. A pairwise Pearson's correlation coefficient value of $r \geq \pm 0.7$ was considered strongly correlated (Dormann et al., 2013) and was not modelled together, as I were interested in seeing the effect of each covariate on the relative abundance of sloth bears. All analyses were carried out using the packages "Unmarked" (Fiske and Chandler, 2011) and "AICmodavg" (Mazerolle, 2015) implemented in R (R Core Team, 2019).

In order to understand the performance of the N-mixture model, the mean site abundance estimate was compared with estimates obtained from another frequently used site-structured model, the RN model (Royle and Nichols, 2003). Since modelling the relative abundance of the sloth bear with both local-scale and GIS-based covariates by applying the RN model and subsequent comparison of the model performance (of both N-mixture and RN models) was beyond the scope of this chapter, I restricted the comparison between the null models of the two frameworks mentioned above. I chose the same probability distribution for the RN model that was identified as the most suitable for the N-mixture model. The program Presence (Version 2.13.11) (Hines, 2006) was used to estimate the mean site abundance of sloth bears by applying the RN model.

3.3 Results

Carnivore sign survey

In 59 Beats, a total of 775km (~13km/Beat) of forest roads/trails were walked during the carnivore sign survey. Sloth bear signs (n=157) were recorded from 41 Beats out of 59, which yielded a naïve occupancy estimate of 0.69 and a sign encounter rate of 20/100km.

Probability of site use by sloth bears

I found that correlated occupancy models indicated a strong correlation between segments (Θ=0.17±0.04, Θ’=0.64±0.19; Table 3.3) while using 1km segment length (i.e., 15 segments or spatial replicates). However, collapsing the detection history into five segments (of 3km each) produced the least amount of correlation (Θ ≈ Θ’) of detections between segments (Θ=0.20±0.03, Θ’=0.32±0.04). Hence, I constructed standard single-season single-species occupancy models with five segments or spatial replicates and all standardised covariates.

Table 3.3. Correlated occupancy model (null) with varied segment lengths (15km, 10km, 5km) for sloth bears in Sanjay Tiger Reserve, Madhya Pradesh, India, during 2016-2017; Model parameter estimates: ϴ - the probability of the species being present at a segment given the previous segment was not occupied by the species, ϴ' - the probability of the species being present at a segment given the previous segment was occupied by the species, Ψ - occupancy estimate, p - detection probability; SE-standard error; AIC-Akaike Information Criterion.

No. of segment	**AIC**	**ϴ ± SE**	**ϴ' ± SE**	**Ψ ± SE**	**p ± SE**
15	526.04	0.17±0.04	0.64±0.19	1.00±0.00	0.34±0.13
10	445.22	0.20±0.06	0.53±0.21	1.00±0.00	0.46±0.22
5	306.29	0.20±0.03	0.32±0.04	1.00±0.00	1.00±0.00

Single-season single-species occupancy model revealed that none of the site-specific covariates influenced the probability of site use by sloth bears, as the top-ranked models (models with $\Delta AIC \leq 2$) included the null model (Table 3.4 and Table 3.5). I could not find any significant relationship between detection probability and covariates (both detection and site-specific). However, top-ranked models retained some of the covariates, despite being statistically insignificant (95% confidence interval overlapped zero, i.e. $p>0.05$; Table 3.4 and Table 3.5). The probability of site use by sloth bears was high for all the top-ranked models (mean $\Psi=0.94\pm0.09$). The mean detection probability of sloth bear signs was 0.28±0.04. I did not adjust the standard error of the parameter estimates as there was no evidence of overdispersion in these models (mean $\hat{c}=0.88$).

Table 3.4. Details of the model selection results of the single-season single-species occupancy model for estimation of sloth bear's probability of site use (Ψ) and its detection probability (p) in Sanjay Tiger Reserve, Madhya Pradesh, India, during 2016-2017; Covariates considered: fruitingtree- number of fruiting trees, livestock- number of livestock dung/pellet, termite- number of termite mounds, cutting- number of fresh tree cutting signs, and area- area (in km^2) of forest Beat; Model selection was based on number of parameters (nPars), Akaike Information Criterion (AIC), the difference in AIC between best fit models (ΔAIC≤2), and AIC weight (AICwt); All models (including null models) are represented on the basis of the lowest to highest value of ΔAIC.

Model	**AIC**	**ΔAIC**	**AICwt**	**nPars**
Ψ(.)p(fruitingtree)	304.14	0.00	0.10	3
Ψ(.)p(.)	304.82	0.68	0.07	2
Ψ(.)p(livestock)	304.93	0.79	0.07	3
Ψ(.)p(fruitingtree+livestock)	304.94	0.80	0.07	4
Ψ(.)p(termite+livestock)	304.96	0.82	0.07	4
Ψ(.)p(fruitingtree+cutting)	305.18	1.04	0.06	4
Ψ(.)p(area)	305.39	1.25	0.05	3
Ψ(.)p(termite)	305.48	1.34	0.05	3
Ψ(.)p(cutting)	305.50	1.36	0.05	3
Ψ(.)p(fruitingtree+area)	305.65	1.51	0.05	4
Ψ(.)p(fruitingtree+termite)	305.71	1.57	0.05	4
Ψ(.)p(livestock+area)	305.72	1.58	0.05	4
Ψ(livestock)p(fruitingtree)	306.14	2.00	0.04	4

Table 3.5. Untransformed beta (β) estimates, associated standard error (SE) and 95% confidence intervals of β estimates, obtained from the single-season single-species occupancy models to determine sloth bear's probability of site use in Sanjay Tiger Reserve, Madhya Pradesh, India, during 2016-2017; Covariates considered: fruitingtree- number of fruiting trees, livestock- number of livestock dung/pellet, termite- number of termite mounds, cutting- number of fresh tree cutting signs, and area- area (in km^2) of forest Beat.

Covariates (used for modelling detection probability)	**β estimates**	**SE (Standard Error)**	**95% Confidence interval**	
			Lower confidence limit	**Upper confidence limit**
fruitingtree	-0.25	0.15	-0.54	0.04
livestock	-0.21	0.16	-0.52	0.10
termite	0.22	0.15	-0.07	0.51
cutting	-0.18	0.16	-0.49	0.13
area	-0.19	0.16	-0.50	0.12

Camera trap

A total of 191 and 5292 independent photographs of sloth bears and humans, respectively, were obtained from 5950 trap-nights (3870 and 2080 trap-nights in SNP and DWLS, respectively). Similarly, in 2018, 292 and 4314 photographs of sloth bears were obtained out of a total effort of 6480 trap-nights. Sloth bears were detected at 76 out of 143 (53.10%) camera trap locations in 2017 and 86 out of 144 (59.7%) camera trap locations in 2018. Independent detections of sloth bears varied from 1 to 17 per site (camera trap locations at 4 km^2 grids). However, in the larger grids (12.40 km^2), the presence of sloth bears was recorded at 51 out of 73 (70.0%) grids. Similarly, in 2018, 54 grids (76%) out of 71 contained detections of sloth bears.

Detection process

The detection probability of sloth bears was significantly influenced by the density of fruiting trees for Poisson, NB and ZIP distributions (Table 3.6). The estimated detection probability (p) of the best detection model (modelled with the density of fruiting trees) was found to be higher in Poisson (0.08 ± 0.01) than in ZIP (0.06 ± 0.02) and NB (0.01 ± 0.004) distributions (Table 3.6). However, in 2018, no covariates influenced the detection probability; hence, detection probability was kept constant while modelling the abundance. The estimate of detection probability was the highest for Poisson (0.11 ± 0.01), followed by ZIP (0.08 ± 0.02) and NB (0.01 ± 0.004).

Table 3.6. Model parameter estimates [while abundance (λ) was held constant] for the detection probability (p) of sloth bear, obtained from the N-mixture model in Sanjay Tiger Reserve, Madhya Pradesh, India, during 2016-2017; Three distributions [Poisson, negative binomial (NB) and zero-inflated Poisson (ZIP)] were used to model the detection probability of sloth bear; Covariates considered: fruit_density- density of fruiting trees per 10m-radius plot around each camera trap location, shrub_density- density of shrubs per 5m-radius plot around each camera trap location, termitemounds- number of termite mounds per 10m-radius plot around each camera trap location, trapdays_total- total number of days or occasions for which camera trap (s) was active; AIC- Akaike Information Criterion; Detection probability estimates (p) were provided for the best-selected models of each distribution.

Model description	**β estimates**	**SE (Standard Error)**	**P value**	**AIC**	**p**	**SE (Standard Error)**
Poisson						
[p(fruit_density)λ(.)]	0.27	0.11	0.01 (<0.05)	1020.22	0.08	0.01
[p(shrub_density)λ(.)]	0.16	0.14	0.25 (>0.05)	1025.15		
[p(termitemounds)λ(.)]	0.09	0.11	0.42 (>0.05)	1025.85		
[p(trapdays_total)λ(.)]	0.03	0.14	0.80 (>0.05)	1026.42		
NB						
[p(fruit_density)λ(.)]	0.28	0.14	0.04 (<0.05)	991.86	0.01	0.004
[p(shrub_density)λ(.)]	0.09	0.17	0.57 (>0.05)	995.44		
[p(termitemounds)λ(.)]	0.08	0.15	0.59 (>0.05)	995.47		
[p(trapdays_total)λ(.)]	0.07	0.17	0.68 (>0.05)	995.59		
ZIP						
[p(fruit_density)λ(.)]	0.29	0.11	0.007 (<0.05)	1017.04	0.06	0.02
[p(shrub_density)λ(.)]	0.16	0.13	0.22 (>0.05)	1022.56		
[p(termitemounds)λ(.)]	0.10	0.11	0.377 (>0.05)	1023.30		
[p(trapdays_total)λ(.)]	-0.04	0.16	0.81 (>0.05)	1024.04		

Relative abundance of sloth bears and effects of covariates

In top-ranked models, the relative abundance of sloth bears was significantly influenced by forest type (mixed and sal forest), LULC types (dense forest and barren land), distance to the nearest village (anthropogenic) and human photo-capture rate (anthropogenic) (Table 3.7, Table 3.8, Table 3.9 and Table 3.10). In addition to these covariates, the most parsimonious models also included the other LULC types (agricultural land, scrubland and water body), but none of these significantly influenced the relative abundance of sloth bears (Table 3.9). Models with NB distributions were the most parsimonious in terms of lower overall AIC values, followed by ZIP and Poisson models (Table 3.7 and Table 3.8). Due to uninformative parameters, model averaging was not done for any of the top-ranked models. Areas of mixed forest, sal forest and dense forest predicted a higher relative abundance of sloth bears (Table 3.9 and Table 3.10; Figure 3.3 and Figure 3.4). Similarly, the distance to the nearest village and human photo-capture rate also had significant positive effects (Table 3.9 and Table 3.10; Figure 3.3 and Figure 3.4) on sloth bear abundance.

Table 3.7. Details of the model selection results of the N-mixture model for estimation of the relative abundance of sloth bears in Sanjay Tiger Reserve, Madhya Pradesh, India, during 2016-2017; Models include the most parsimonious models with the best-selected covariates and null models for the Poisson, negative binomial (NB) and zero-inflated Poisson (ZIP) distributions; Covariates considered: fruit_density- density of fruiting trees, mixed- area of mixed forest, sal.forest- area of sal forest, human_CR- photographic capture rate of humans, vildist- distance to nearest village, Agricultural.land- area of agricultural land, Scrubland- area of scrubland and, Water.body- area of water body; Model selection was based on number of parameters (nPars), Akaike Information Criterion (AIC), difference in AIC between best fit models (ΔAIC≤2), AIC weight (AICwt) and cumulative AIC weight of models (Cumltv-wt); All models (including null models) are represented on the basis of the lowest to highest value of ΔAIC for each distribution.

Model description	**nPars**	**AIC**	**ΔAIC**	**AICwt**	**Cumltv-wt**
a. Poisson					
[p(fruit_density)λ(sal.forest+vildist+human_CR+mixed)]	7	998.08	0.00	0.36	0.36
[p(fruit_density)λ(sal.forest+vildist+human_CR+mixed+Agricultural.land)]	8	999.52	1.44	0.17	0.53
Null model [p(.)λ(.)]	2	1024.49	26.41	0.00	1.00
b. NB					
[p(fruit_density)λ(sal.forest+vildist+human_CR+mixed)]	8	983.00	0.00	0.11	0.11
[p(fruit_density)λ(mixed +human_CR)]	6	983.15	0.15	0.11	0.22
[p(fruit_density)λ(mixed+human_CR+Agricultural.land)]	7	983.43	0.43	0.09	0.31
[p(fruit_density)λ(mixed+sal.forest+human_CR)]	7	983.75	0.75	0.08	0.39
[p(fruit_density)λ(mixed+vildist+human_CR)]	7	983.81	0.81	0.08	0.47
[p(fruit_density)λ(sal.forest+vildist+human_CR+mixed+Agricultural.land)]	9	984.26	1.25	0.06	0.53
[p(fruit_density)λ(mixed+Scrubland+human_CR)]	7	984.60	1.59	0.05	0.58
[p(fruit_density)λ(mixed+Agricultural.land)]	6	984.71	1.70	0.05	0.63
[p(fruit_density)λ(mixed+Water.body+human_CR)]	7	984.96	1.96	0.04	0.67
[p(fruit_density)λ(mixed+sal.forest)]	6	984.99	1.99	0.04	0.71
Null model [p(.)λ(.)]	3	993.76	10.76	0.001	1.00
c. ZIP					
[p(fruit_density)λ(sal.forest+vildist+human_CR+mixed)]	8	995.09	0.00	0.36	0.36
[p(fruit_density)λ(sal.forest+vildist+human_CR+mixed+Agricultural.land)]	9	996.11	1.01	0.22	0.58
Null model [p(.)λ(.)]	3	1022.10	27.00	0.00	1.00

Table 3.8. Details of the model selection results of the N-mixture model for estimation of the relative abundance of sloth bears in Sanjay Tiger Reserve, Madhya Pradesh, India, in 2018; Models include the most parsimonious models with the best-selected covariates and null models for the Poisson, negative binomial (NB) and zero-inflated Poisson (ZIP) distributions; Covariates considered: Dense.forest- area of dense forest, vildist- distance to nearest village, human_CR- photographic capture rate of humans, mixed- area of mixed forest, Barren.land- area of barren land, and sal.forest- area of sal forest. Model selection was based on number of parameters (nPars), Akaike Information Criterion (AIC), the difference in AIC between best-fit models ($\Delta AIC \leq 2$), AIC weight (AICwt) and cumulative AIC weight of models (Cumltv-wt); All models (including null models) are represented on the basis of the lowest to the highest value of ΔAIC for each distribution.

Model description	**nPars**	**AIC**	**ΔAIC**	**AICwt**	**Cumltv-wt**
a. Poisson					
[p(.)λ(Dense.forest+vildist+human_CR)]	5	1298.41	0.00	0.25	0.25
[p(.)λ(vildist+human_CR+mixed+Barren.land)]	6	1298.42	0.01	0.24	0.49
[p(.)λ(sal.forest+vildist+human_CR+Dense.forest)]	6	1299.29	0.88	0.16	0.65
Null model [p(.)λ(.)]	2	1356.72	58.31	0.00	1.00
b. NB					
[p(.)λ(Dense.forest+Barren.land+human_CR)]	6	1273.86	0.00	0.32	0.32
Null model [p(.)λ(.)]	3	1303.14	29.28	1.4e-07	1.00
c. ZIP					
[p(.)λ(Dense.forest+vildist+human_CR)]	6	1296.73	0.00	0.43	0.43
[p(.)λ(sal.forest+vildist+human_CR+Dense.forest)]	7	1297.60	0.87	0.28	0.70
Null model [p(.)λ(.)]	3	1346.67	49.93	0.00	1.00

Table 3.9. Parameter estimates of N-mixture models to determine sloth bear's relative abundance in Sanjay Tiger Reserve, Madhya Pradesh, India, during 2016-2017; Estimates were derived from the Poisson, negative binomial (NB), and zero-inflated Poisson (ZIP) distributions for the mean site abundance (Lambda, λ) of sloth bears; Covariates considered: mixed- area of mixed forest, sal.forest- area of sal forest, human_CR- photographic capture rate of humans, vildist- distance to the nearest village, Agricultural.land- area of agricultural land, Scrubland- area of scrubland and, Water.body- area of water body.

Model description	**β estimates**	**SE (Standard error)**	**P value**
Poisson			
Intercept λ(.)	0.95	0.25	0.0001 (<0.005)
λ(mixed)	0.33	0.10	0.001 (<0.005)
λ(human_CR)	0.27	0.08	0.0005 (<0.005)
λ(vildist)	0.25	0.11	0.02 (<0.05)
λ(sal.forest)	0.21	0.10	0.04 (<0.05)

λ(Agricultural.land)	-0.10	0.13	0.47 (>0.05)
NB			
Intercept λ(.)	2.38	0.35	1.54e-11 (<0.005)
λ(mixed)	0.28	0.13	0.03 (<0.05)
λ(sal.forest)	0.25	0.15	0.09 (>0.05)
λ(human_CR)	0.24	0.12	0.04 (<0.05)
λ(vildist)	0.23	0.14	0.09 (>0.05)
λ(Agricultural.land)	-0.21	0.16	0.19 (>0.05)
λ(Scrubland)	0.14	0.18	0.45 (>0.05)
λ(Water.body)	-0.07	0.17	0.66 (>0.05)
ZIP			
Intercept λ(.)	1.40	0.38	0.0002 (<0.005)
λ(mixed)	0.31	0.11	0.004 (<0.005)
λ(sal.forest)	0.31	0.14	0.02 (<0.05)
λ(human_CR)	0.27	0.07	0.0002 (<0.005)
λ(vildist)	0.26	0.10	0.01 (<0.05)
λ(Agricultural.land)	-0.14	0.14	0.33 (>0.05)

Table 3.10. Parameter estimates of N-mixture models to determine sloth bear's relative abundance in Sanjay Tiger Reserve, Madhya Pradesh, India, in 2018; Estimates were derived from the Poisson, negative binomial (NB), and zero-inflated Poisson (ZIP) distributions for the mean site abundance (Lambda, λ) of sloth bears; Covariates considered: Dense.forest- area of dense forest, vildist- distance to nearest village, human_CR- photographic capture rate of humans, mixed- area of mixed forest, Barren.land- area of barren land, and sal.forest- area of sal forest.

Model description	**β estimates**	**SE (Standard error)**	**P value**
Poisson			
Intercept λ(.)	1.04	0.19	1.42e-07 (<0.005)
λ(Dense.forest)	0.53	0.09	2.85e-08 (<0.005)
λ(vildist)	0.28	0.09	3.26e-03 (<0.005)
λ(human_CR)	0.36	0.07	2.17e-07 (<0.005)
λ(mixed)	0.37	0.09	6.21e-05 (<0.005)
λ(Barren.land)	-0.46	0.26	0.07 (>0.05)
λ(sal.forest)	0.11	0.09	0.28 (>0.05)
NB			
Intercept λ(.)	2.54	0.29	1.00e-18 (<0.005)
λ(Dense.forest)	0.54	0.16	6.12e-04 (<0.005)
λ(Barren.land)	0.54	0.31	0.07 (>0.05)
λ(human_CR)	0.28	0.10	0.007 (<0.05)
ZIP			
Intercept λ(.)	1.24	0.25	8.72e-07 (<0.005)
λ(Dense.forest)	0.54	0.10	6.39e-08 (<0.005)

λ(vildist)	0.30	0.09	0.001 (<0.005)
λ(human_CR)	0.33	0.07	2.46e-06 (<0.005)
λ(sal.forest)	0.11	0.10	0.28 (>0.05)

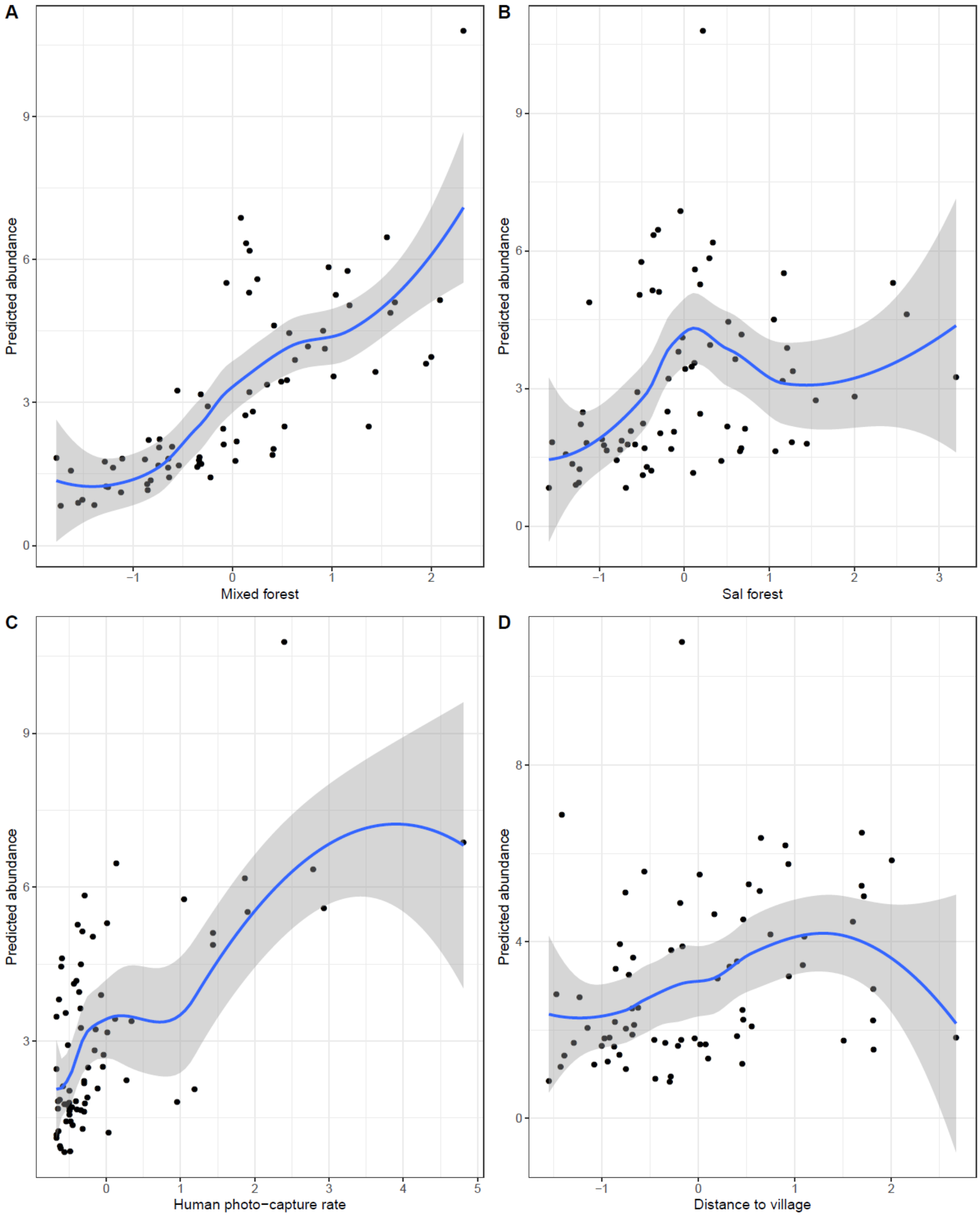

Figure 3.3. Effect of mixed forest (**A**), sal forest (**B**), human photo-capture rate (**C**) and distance to the village (**D**) on sloth bear's predicted relative abundance in Sanjay Tiger Reserve, Madhya Pradesh, India, during 2016-2017, based on the most parsimonious models of Poisson distribution; All four covariates were shown in the standardised scale.

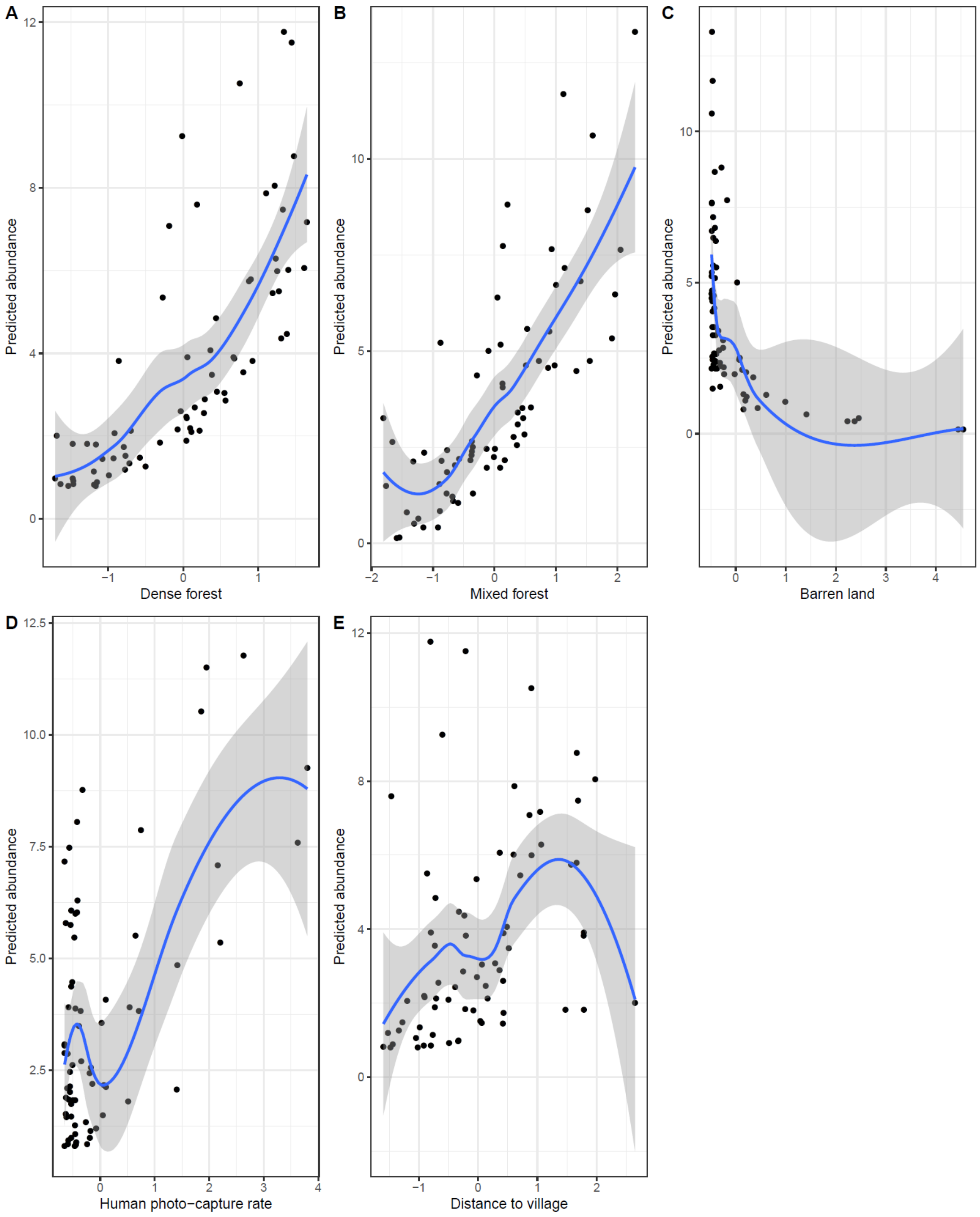

Figure 3.4. Effect of dense forest (**A**), mixed forest (**B**), barren land (**C**), human photo-capture rate (**D**) and distance to the village (**E**) on sloth bear's predicted relative abundance in Sanjay Tiger Reserve, Madhya Pradesh, in 2018, based on the most parsimonious models of Poisson distribution; All five covariates were shown in the standardised scale.

However, the models within ≤2 ΔAIC units for Poisson, ZIP and NB distributions did not pass the GoF test. Overdispersion parameter estimates (ĉ) revealed a marginal lack of fit, which did not appear to differ among Poisson (ĉ =1.31 in 2017; ĉ =1.37 in 2018), NB (ĉ = 1.19 in 2017; ĉ =1.23 in 2018) and ZIP (ĉ = 1.28 in 2017; ĉ =1.34 in 2018) distributions. Residual diagnostics and mapping of residuals did not show any specific pattern of lack of fit. Hence, this finding could be attributed to the unstructured noise or overdispersion rather than a true lack of fit. The NB model (top-ranked) produced unusually high mean site abundance (λ = 10.80 ± 3.83 in 2017; λ = 12.60 ± 3.63 in 2018) in comparison to mean site abundance estimates of ZIP (λ = 4.07 ± 1.56 in 2017; λ = 3.50 ± 0.88 in 2018) and Poisson models (λ = 2.60 ± 0.64 in 2017; λ = 2.83 ± 0.56 in 2018) (Table 3.11). The predicted site abundance of sloth bears was mapped and shown in Figure 3.5.

Table 3.11. Mean site abundance (λ) of sloth bears and associated standard error (SE) in Sanjay Tiger Reserve, Madhya Pradesh, during 2017-2018; Best-selected models were from three [Poisson, negative binomial (NB) and zero-inflated Poisson (ZIP)] distributions; Covariates considered: fruit_density- density of fruiting trees, mixed-area of mixed forest, sal.forest- area of sal forest, human_CR- photographic capture rate of humans and, vildist- distance to the nearest village.

Year	**Model description**	**Mean site abundance (λ) ± Standard error (SE)**
	Poisson	
2017	[p(fruit_density)λ(sal.forest+vildist+human_CR+mixed)]	2.60 ± 0.64
2018	[p(.)λ(Dense.forest+vildist+human_CR)]	2.83 ± 0.56
	NB	
2017	[p(fruit_density)λ(sal.forest+vildist+human_CR+mixed)]	10.80 ± 3.83
2018	[p(.)λ(Dense.forest+Barren.land+human_CR)]	12.60 ± 3.63
	ZIP	
2017	[p(fruit_density)λ(sal.forest+vildist+human_CR+mixed)]	4.07 ± 1.56

2018	[p(.)λ(Dense.forest+vildist+human_ CR)]	3.50 ± 0.88

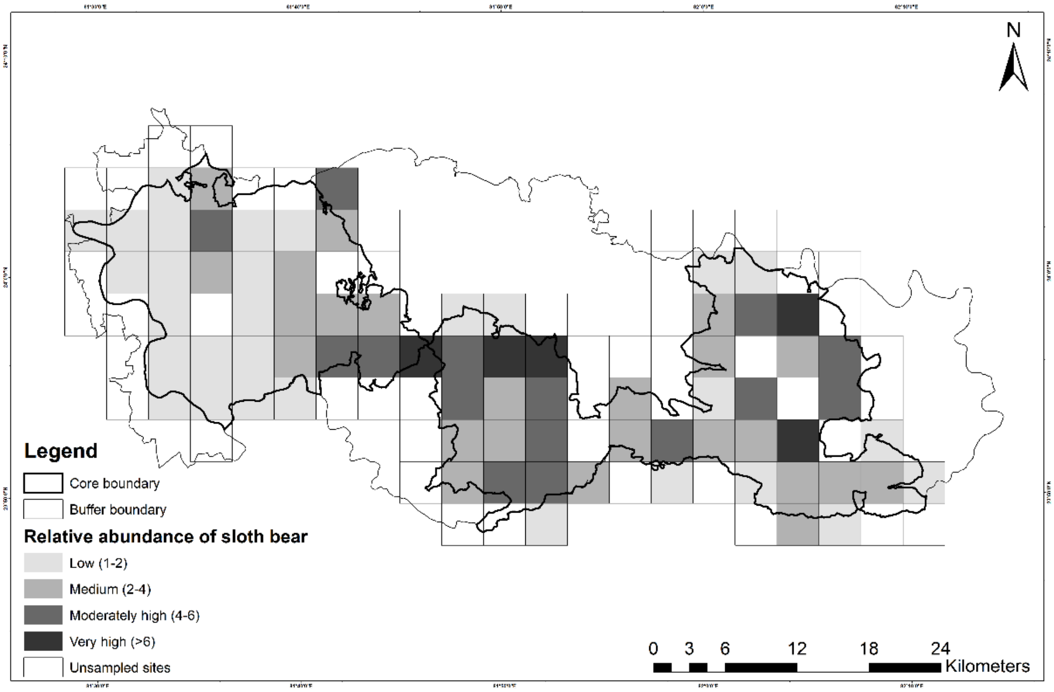

Figure 3.5. Map showing the relative abundance of sloth bears in Sanjay Tiger Reserve, Madhya Pradesh, during 2017-2018; Estimates were based on the most parsimonious model of Poisson distribution.

Comparing estimates with the RN model

The RN model (null model with Poisson distribution) produced a mean site abundance estimate of 1.62 ± 0.34 in 2017 and 1.70 ± 0.29 in 2018, slightly lower than the N-mixture model's (2.05 ± 0.35 in 2017 and 2.38 ± 0.32 in 2018) estimates. However, the mean site abundance estimate obtained from the N-mixture model in 2017 was within the 95% confidence interval range (1.08 - 2.44) of the estimate produced by the RN model. In 2018, the N-mixture model produced marginally higher mean site abundance than the upper limit of the RN model's 95% confidence interval range (1.22 - 2.37).

3.4 Discussion

Occupancy of sloth bear

I estimated sloth bears' probability of site use in the core zone of STR. The fine-scale spatial covariates unaffected the site use probability of sloth bears. Such results were due to the ubiquitous presence of sloth bears and associated less variability of their presence across the sampling units, which the covariates could not adequately explain. However, in the top-ranked models, the detection of sloth bears was influenced by fruiting trees, livestock presence, termite mounds, cutting signs, and area of the forest Beat, although none of these relationships was statistically significant. Insignificant relationships between bear occurrence and covariates could be attributed to the scale at which the covariates were collected. The abundance of termite mounds and fruiting trees is not only restricted to the forest roads/trails but could be anywhere, depending on suitable habitat factors. Hence, counting fruiting trees and termite mounds alongside the forest roads/trails may not adequately represent their availability in the entire sampling unit (Beat). Also, I could not quantify the underground termite and ant colonies, which could influence the probability of site use and detection of sloth bears. Fine-scale spatial covariates (including anthropogenic disturbances) can often fail to produce a significant relationship with occupancy and detection probability, given the similar study design I followed (Paudel et al., 2022; Pokharel et al., 2022). Perhaps, to identify the governing factors related to sloth bears' occurrence, given their wide distribution with little variation, count-based abundance models could be a more appropriate choice. However, the occupancy-based framework that I followed in this study was imperative to understanding the distribution of sloth bears, which would also guide the selection of proper methodologies for further investigations of other aspects of bear ecology. Nonetheless, the high probability of site use by sloth bears and insignificant relationships between covariates and bear occurrence indicated the sloth bear's overall wide distribution in the core zone of STR.

Efficacy of the N-mixture model

The first-ever estimates of the relative abundance of the sloth bear was reported using the N-mixture model. I also identified associated factors governing the relative abundance using count data obtained from the camera trap survey in STR. The N-mixture model can produce reliable abundance (relative) estimates from spatially replicated count data if its assumptions are met while accounting for imperfect detection (Royle, 2004). Also, its ability to model spatial variation of abundance and detection probability as a function of relevant covariates makes its inference more robust and reliable than inferences drawn from conventional index/encounter rate-based surveys (Gilbert et al., 2021). A growing number of studies have assessed the population abundance (relative and absolute) of unmarked individuals by using the N-mixture model from camera trap data (Brodie and Giordano, 2013; Keever et al., 2017; Xiao et al., 2018; Kafley et al., 2019; Penjor et al., 2019) as well as employing other field methods (Belant et al., 2016; Ficetola et al., 2018; Kidwai et al., 2019; Searle et al., 2020). Estimating the abundance of unmarked animals from camera trap data has always been a challenge for ecologists. For the past two decades, very few studies have actually assessed the population abundance of Asian bears employing statistically rigorous methods; instead, many of the studies have heavily relied upon sign index or interview-based distributions due to the relatively less expensive nature and ease of application in the field (Garshelis et al., 2022; Proctor et al., 2022). Information on sloth bear abundance and its governing factors are not well studied throughout the species' distributional range (Dharaiya et al., 2016). Most of the previous studies (Eisenberg and Lockhart, 1972; Laurie and Seidensticker, 1977; Santiapillai and Santiapillai, 1990; Ratnayeke et al., 2007; Akhtar et al., 2008) regarding the abundance of sloth bears produced empirical estimates, except for Garshelis et al. (1999), where the authors first estimated abundance from sound analytical methods in Nepal's Chitwan. However, in the case of this wide-ranging bear species, site-structured models (i.e., the RN model and the N-mixture model) may be more reliable for estimating the population trend with careful considerations of model assumptions and study design (Morin et al., 2022).

It is well-advised that abundance estimates obtained from site-structured models should be interpreted as a measure of relative, not absolute, abundance if the assumption of closure is likely to have been violated (Barker et al., 2018; Link et al., 2018; Gilbert et al., 2021). In this context, mean abundance (lambda) estimated by the N-mixture model was essentially seen as the number of individuals utilising a site (12.40 km^2 grids in the present study) instead of the total number of individuals residing within a site. Site abundance estimates obtained from the N-mixture models would be consistently larger than the actual abundance of free-ranging animals like sloth bears, as each site contains multiple overlapping territories of different individual sloth bears. In reality, the true abundance of animals comes from a considerably larger geographic extent than the actual surveyed area, leading to an overestimation of the mean site abundance and total predicted abundance (Joseph et al., 2009). In the present study, the best-selected model (with NB distribution) in terms of AIC produced substantially larger and ecologically unrealistic abundance estimates of sloth bears compared to ZIP and Poisson models. Mean abundance estimates obtained from NB and ZIP models showed that a large number (~11 and ~4 for NB and ZIP models, respectively) of bears were utilising each site with considerably low detection probability, which, in general, is unlikely for a species like the sloth bear. In the Indian sub-continent, the lack of studies focusing specifically on the density or abundance of sloth bears and other bear species in a scientifically rigorous framework poses difficulties for comparing my findings to other studies. It was expected that for a species like sloth bear, an elusive and solitary ursid, count data obtained from camera traps would be zero-inflated. The NB model performs poorly with a zero-inflated count dataset or when the focal species is not frequently found or is truly absent from a large number of sites surveyed (Joseph et al., 2009). In the present study, models other than Poisson distribution, especially the NB variant of N-mixture models, produced lower detection probabilities than Poisson and ZIP models. Due to low detection probability, N-mixture models can produce positively biased abundance estimates with less precision than other methods (Dénes et al., 2015; Dennis et al., 2015). On the other hand, the N-mixture model with a Poisson distribution can handle datasets having excessive true zeros (sites where the species is truly absent), given the true maximum site abundance is close to zero or when the species does not naturally occur in high densities (Joseph et al., 2009). In my study, the Poisson model produced a comparatively low mean site abundance (2.60±0.64) by modelling a substantial number (~30% of total sites surveyed) of unoccupied sites. I suggest interpreting the abundance estimates of sloth bears carefully with an ecologically realistic view rather than with strict adherence to the statistical properties of the models.

The mean site abundance of sloth bears produced from the N-mixture model (Poisson distribution) was comparable to estimates obtained from the RN model. Earlier studies have indicated that the RN model would not be a good choice when the focal species is common; in that case, the N-mixture model would be preferable (Dénes et al., 2015; Kéry and Royle, 2015; Gilbert et al., 2021). The RN model reportedly produces imprecise estimates from camera trap-based surveys (covering comparatively small sampled areas), especially if the focal species is cryptic (Gupta et al., 2012; Kalle et al., 2014; Morin et al., 2022). However, there is evidence that the RN model can also produce precise abundance estimates, enabling it to detect population trends (O'Brien et al., 2020). Estimates from the RN model can approximate the absolute density of the focal species if the study is conducted on a landscape scale in conjunction with an ecologically relevant grid size for camera trap deployment (Linden et al., 2017). On the other hand, simulation studies have shown that the N-mixture (Poisson and Binomial mixture) model may produce biased estimates of mean abundance in the context of false-positive detections (Nakashima, 2020). However, my findings from both models indicated the overall utility of site-structured models in estimating the relative abundance of elusive species like sloth bears. Although the number of camera trap locations/grid varied (1-4) in my study, the presence of >1 spatial replicates (camera trap locations) within most of the grids (>50%) might have improved the detection and robustness of mean site abundance estimates (Kolowski et al., 2021; Morin et al., 2022).

Applicability of other methods for unmarked population estimation

Due to the scarcity of spatially explicit studies, I could not directly compare relative abundance estimates of sloth bears with previous findings. Since the primary objective of this study was to understand the applicability of the N-mixture model, I considered other potential methods relevant to the abundance estimation of unmarked animals. Abundance estimation of Asian bears through camera traps rarely produces reliable and precise estimates for conservation planning (Morin et al., 2022). However, ancillary information obtained from statistically rigorous methods and covering a large number of sampling sites might improve the reliability of abundance estimates (Morin et al., 2022). Apart from the site-structured models presented here, USCR (i.e., the spatial count (SC) model; Chandler and Royle, 2013) and its extension (i.e., spatial presence-absence (SPA) model; Ramsey et al., 2015) can also be applied to estimate the absolute density of cryptic species. The application of the SPA model has been well-demonstrated in estimating the density of small carnivores through camera traps (Chatterjee et al., 2020a, 2020b). However, these models are computationally intensive (require Bayesian analysis) and sensitive to the appropriate choice of priors which could be challenging for a species like the sloth bear due to limited knowledge of its ranging pattern. Distance sampling through camera traps could also be a promising approach to estimating the density of unmarked animals (Howe et al., 2017), especially if the focal species is less abundant (Palencia et al., 2021). However, this method requires the random placement of camera traps which I could not implement, as this study followed the protocol of all India tiger estimation (Jhala et al., 2015). Random placement of camera traps might result in substantially low detections of sloth bears, which may not be conducive for further analysis. The N-mixture model would be particularly suitable when there are logistic constraints regarding camera placement and cost-effectiveness. Moreover, if the study intends to model the spatial variation of abundance (relative or absolute), the N-mixture model would be preferable compared to the other methods mentioned above. As long as the study objectives are clearly defined and interpretations of abundance estimates are correctly made, the N-mixture model would be a suitable option for researchers and managers.

Effects of environmental and anthropogenic covariates on relative abundance

In the present study, mixed forest, dense forest and sal forest positively influenced the relative abundance of sloth bears. In STR, I calculated the density of termite mounds and fruiting trees, utilising a 10m-radius plot around each camera trap location. I recorded a high density (35/hectare) of termite mounds in mixed forest, followed by sal mixed and sal forest (27/hectare and 23/hectare, respectively). Similarly, the density of fruiting trees was also found to be higher (208/hectare) in the mixed forest in comparison to sal mixed (197/hectare) and sal forest (100/hectare). Due to fewer plots, I could not calculate meaningful density estimates of termite mounds and fruiting trees for the bamboo mixed forest and grasslands. In STR, a high number of fruiting trees and termite mounds in mixed forests may explain the positive relationship between bear abundance and mixed forest. The relative abundance of sloth bears increased with the increasing area of dense forest in STR, which could be related to the forage availability. I found that the area of dense forest was highly (positively) correlated (0.88) with the mixed forest, which also indicated that most of the dense forest areas came under the mixed forest type. A similar observation was also reported for Panna National Park in central India, where dense forest (comprised of miscellaneous tree species) had the highest densities of fruiting trees and prey insect colonies, and sloth bears preferred this habitat significantly (Yoganand, 2005). However, the limited number of surveyed plots (only at camera trap locations) to support my findings may not be enough to explain the effects of forest types and termite mounds on sloth bear abundance, for which more rigorous sampling is needed. In the North Bilaspur Forest Division, Akhtar et al. (2004) found that signs of sloth bears were more common in sal forest than in other forest types. Similar results were also found in Chitwan, but with distinct seasonal patterns (Garshelis et al., 1999a). The occurrence of bear signs (ground holes and mound holes) in sal forest was frequent in both landscapes, perhaps due to a relatively high number of termite mounds (Garshelis et al., 1999a; Akhtar et al., 2004) and possibly the underground colonies of ants and termites. In the northern part of the Indian sub-continent, it was observed that termite mounds are more abundant in sal forests (Chakraborty and Singh, 2020). However, underground colonies of termites and ants are difficult to quantify. Hence, the number of termite mounds that are above-ground and clearly visible has been frequently used as a surrogate for the availability of ants and termites (Akhtar et al., 2004; Rather et al., 2020). I also could not find any significant relationship between the relative abundance of sloth bears and local-scale covariates, although the availability of fruiting trees positively influenced the detection probability of sloth bears. In Nepal’s Chitwan and Churia (a lowland forest outside the PAs) forests, the sloth bear’s occupancy and detection probability were positively influenced by the presence of fruiting trees and termite mounds (Paudel et al., 2022; Pokharel et al., 2022). A similar observation was also recorded by Das et al. (2014) in southern India during their occupancy-based study. However, I believe that my camera trap-based study design may not be appropriate for drawing inferences on such local-scale covariates, despite their proven importance to the sloth bear's ecology. The ubiquitous nature of termites and ants in a relatively small area (Redford, 1987) may explain the insignificant relationship between bear abundance and termite mounds. Similarly, the availability of fruiting trees and termite mounds at a particular point (10m-radius plot, in my study) may not necessarily represent the entire sampling unit (12.40 km^2 grids), which could explain the apparent insignificant contribution of fruiting tree density and termite mounds to the relative abundance of sloth bear.

Sloth bears are generally known to avoid human disturbances (Babu et al., 2015; Puri et al., 2015; Paudel et al., 2022; Pokharel et al., 2022) but also tolerate a certain degree of human pressure or have habituated to living in disturbed and fragmented forests (Akhtar et al., 2004; Prajapati et al., 2021). In the present study, I found that the relative abundance of sloth bears was positively influenced by the photographic capture rate of humans and distance to the nearest village. However, based on the current study design, it is not known whether there is a causal relationship between these variables (relative abundance of sloth bear and human photographic capture rate), and to my knowledge, no other studies have shown any such ecological relationship exists. I believe this finding may be purely attributed to the placement of camera traps on the forest roads and trails. Apart from sloth bears (and other carnivores), forest roads and trails are intensively used daily by local people living inside STR. I was interested in this relationship between human occurrence and sloth bear abundance as a matter of concern regarding the possible negative interactions between humans and sloth bears. However, to cope with extensive human pressure in STR, sloth bears have shown a fine-scale seasonal spatio-temporal segregation (Chaudhuri et al., 2022). Also, they avoided areas close to villages in STR, which is in agreement with previous studies (Babu et al., 2015; Puri et al., 2015; Paudel et al., 2022; Pokharel et al., 2022). Such behavior could be attributed to the better availability of forage and shelter (day-resting den sites) in relatively undisturbed habitats.

Nevertheless, avoidance of human settlements does not necessarily curb the possibilities of conflict between humans and sloth bears, given the prevailing extensive anthropogenic pressure in STR. Sloth bears are also known to opportunistically raid crops and forage on fruits of *Ziziphus mauritiana, Mangifera indica* and *Syzygium cumini* that are found in the villages but less abundant inside the forest (Bargali et al., 2004; Palei et al., 2020). Habituation to human settlements often leads to severe conflicts between humans and sloth bears (Bargali et al., 2005; Dhamorikar et al., 2017; Prajapati et al., 2021), so there are significant conservation implications for this species in human-dominated landscapes.

Caveats and limitations

This study has several limitations and caveats. I only collected the covariates alongside the forest roads/trails in the occupancy survey, which may not entirely represent their availability at the level of forest Beat (sampling unit). The deployment of camera traps was done to maximise the probability of capturing tigers and co-predators but was not specifically targeted to a single species, i.e. sloth bear. I could not deploy camera traps intensively due to villages and agricultural fields, which resulted in a varied number of camera traps (1-4) in each of the larger grids (12.40 km^2). Such design may have affected the detection of sloth bears as they are presumed to raid crops and forage on fruits in the villages situated inside the core area of STR. Currently, the camera trap survey was restricted to the core area of STR; however, expanding the sampling area to include the buffer zone could have improved the overall estimates of the sloth bear's relative abundance. Also, the seasonal variation of the mean site abundance of sloth bears was not assessed, as studies conducted elsewhere reported a distinct seasonal variation in sloth bear density (Garshelis et al., 1999a).

3.5 Conclusion

I assessed the sloth bear's occupancy or the site use probability in the core zone of STR as a function of fine-scale spatial covariates and found that sloth bears were widespread in the study area. However, data collection regarding the fine-scale spatial covariates throughout the sampling unit would be ideal for identifying their contribution to the probability of site use by sloth bears. I also demonstrated the application of the N-mixture model to estimate the relative abundance of sloth bears from camera trap data. Choosing an appropriate probability distribution is crucial to providing ecologically realistic estimates. The N-mixture and RN models with Poisson distribution could be suitable options for estimating the relative abundance of solitary, elusive carnivores, despite the better statistical properties of other models with ZIP and NB distributions. However, limited knowledge of sloth bears' range patterns in human-dominated landscapes and substantial variation in home range estimates throughout the species' distributional range could affect the choice of grid size (12.40

km^2 grids in the current study) for the N-mixture model. If the goal is to detect the population changes over time, site-structured models are rarely effective, especially if knowledge of certain aspects of the species' ecology (e.g., home range, population density) is limited (Morin et al., 2022). Future studies to monitor population changes of sloth bears should focus on the placement (i.e., spacing) of camera traps based on a more thorough understanding of their ranging patterns (Fuller et al., 2022). Also, a spatially extensive sampling area with several spatial replicates at each sampling unit (depending on the feasibility and size of the sampling unit) is required to assess the abundance with precision in the case of a low-density population of sloth bears. I highly recommend carrying out density estimation by applying a spatially explicit capture-recapture framework through genetics and comparing results with site-structured models and other suitable camera trap-based methods (USCR, SPA, and distance sampling). However, I believe that, in the absence of absolute abundance estimates for an unmarked, habitat generalist, and conflict-prone species like the sloth bear, knowledge of an ecologically realistic relative abundance and associated factors governing the same are imperative for its long-term conservation in a human-dominated landscape.

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CHAPTER 4: SPACE USE PATTERN BY SLOTH BEAR

PART A: Habitat Use by Sloth Bear

4.1 Introduction

Understanding the space-use by organisms has drawn the ecologists' attention and remained a focal object of interest for decades. Knowledge of animal habitat use is essential to understanding the species' interaction with the environment where it lives, which would govern the spatial distribution and abundance of the concerned species (Boyce and McDonald, 1999). More importantly, in the context of Anthropocene, understanding the space use patterns is becoming increasingly relevant to direct conservation efforts for threatened species (Morris, 2003; Klar et al., 2008; Morrison et al., 2012; Das et al., 2014; Piédallu et al., 2017; Hernando et al., 2021). It was also observed that due to the persistent and expanding anthropogenic pressure, carnivores, in particular, are forced to live in altered or sub-optimal habitats (Martínez-Abraín and Jiménez, 2016). Things become more complicated in the human-dominated landscapes, where the concerned species is habituated to human presence, uses human-modified landscapes and involves in conflict (Akhtar et al., 2004; Bargali et al., 2005; Rather et al., 2021a). Thus, knowledge of space use patterns and identification of (still) available original habitats to be used by the organism in the context of a human-dominated landscape are imperative for the conservation prioritization of the focal species (Das et al., 2014; Hernando et al., 2021).

Sloth bear may occupy a variety of habitats in lowland areas (<1000m elevation) but could be found as high as ~2000m in the Western Ghats of southern Indian states

(Seidensticker et al., 2011). Apart from deciduous forests, sloth bears are found in diverse habitats such as semi-arid forest habitats (Johnsingh, 2003), montane evergreen forests of Western Ghats (Johnsingh, 2003; Babu et al., 2015), alluvial grasslands of Himalayan foothills (Joshi et al., 1995) and dry lowland areas (with monsoon forests) of Sri Lanka (Ratnayeke et al., 2007b). Despite being a widely distributed ursid, rigorous knowledge of habitat use by sloth bears, especially in areas facing extensive anthropogenic pressure, is limited. Habitat selection and distribution of a species primarily depend on the quality and quantity of available habitat as well as the nature and extent of human pressure (Ceballos and Ehrlich, 2002; Schipper et al., 2008). The sloth bear is an omnivore, primarily sustained by colonial insects (ants and termites) and the seasonal availability of fruits (Joshi et al., 1997, 1999; Bargali et al., 2004). So far, studies regarding habitat use by sloth bears have focused on fine-scale habitat features such as the availability of termite mounds and fruiting trees (Ramesh et al., 2012; Das et al., 2014; Pokharel et al., 2022) as well as a combination of habitat features described above and broad-scale landscape features, i.e. forest and vegetation cover, terrain complexity and extent of anthropogenic pressure at both protected area (PA) level (Joshi et al., 1995; Akhtar et al., 2004; Yoganand, 2005; Ratnayeke et al., 2007a; Babu et al., 2015; Rather et al., 2021a; Paudel et al., 2022; Pokharel et al., 2022) and country-level (Puri et al., 2015). To my knowledge, only two studies (Akhtar et al., 2004; Rather et al., 2021a) have been carried out to understand the habitat use patterns by sloth bears and habitat suitability in human-dominated landscapes of central India, especially areas outside of the PA network. However, none of these two studies has teased apart the seasonal variation of habitat use as it is believed that seasonal availability of resources would shape the habitat preference by an organism accordingly (Ager et al., 2003; Godvik et al., 2009; Owen-Smith et al., 2010; Milakovic et al., 2011), which is relevant to the sloth bear as well (Joshi et al., 1995; Garshelis et al., 1999).

In this background, based on camera trap surveys, I studied the seasonal pattern of habitat use by sloth bears in Sanjay Tiger Reserve during 2017 and 2018. I modelled the photographic captures of sloth bears as a function of environmental and anthropogenic covariates at different spatial scales to identify the effect of scales on habitat use. Furthermore, I assessed the microsite habitat features in different habitat types that may influence using the specific broad habitat type(s). In this present study, I had two objectives which are as follows:

1. To assess the seasonal (summer and winter) pattern of habitat use by sloth bears at different spatial scales, and
2. To determine the season-specific role of anthropogenic drivers on the habitat use by sloth bears.

4.2 Materials and methods

Camera trapping survey

The methodological details of the camera trapping survey were mentioned in Chapter 3, under the “Materials and methods” section (also see Figure 3.2).

Habitat use

I used the occurrence of sloth bears in terms of counts of photographic records as a dependent variable. The effect of a set of environmental and anthropogenic covariates (as predictor variables) on the dependent variable was then assessed, based on a generalized linear mixed model (GLMM) framework, following Pal et al. (2020). Details of covariates used in the analysis are mentioned in Table 4.1. Since the locations of camera traps were fixed for the entire sampling period (summer and winter of each year), the location of the camera trap site was used as a random effect (Pal et al., 2020, 2022). I used areas of land use land cover (LULC) classes and forest type classes (both prepared from 30m spatial resolution Landsat-8 images (see details in Chapter 3 under "Materials and methods"), as environmental covariates, along with water availability (distance to the nearest water source) and terrain ruggedness, i.e. ruggedness index (RI) (Riley, 1999). Distance to the nearest village and metal road, presence of humans and livestock in terms of photo-capture rates (of humans and livestock) from camera traps were considered anthropogenic covariates. I tested for the effect of seasons (summer and winter) on sloth bear occurrence and as an interaction term with other covariates to assess the seasonal effect of different covariates on sloth bear occurrence. Since this study experienced unequal camera trap effort in terms of trap-nights, its impact on the photographic counts of sloth bears was also tested. A scale-specific approach was adopted to identify the perceived scale by sloth bears at which they select habitat features, as habitat selection is likely to be influenced by different spatial scales (Wiens, 1989). Failure to incorporate appropriate and biologically relevant scales in habitat selection studies may lead to incorrect inferences regarding the species-habitat relationship (Wiens, 1989; Levin, 1992; Thompson and McGarigal, 2002; Grand and Mello, 2004), yet this issue was poorly addressed in most of the habitat selection studies (McGarigal et al., 2016). In this view, mean values of environmental and anthropogenic covariates (except for Euclidean distance-based covariates and photo-capture rates of humans and livestock) were calculated and extracted for circular buffers of 0.5km, 1km, 1.5km and 2km around each camera trap location. These scales corresponded to the areas from 0.78 km^2 to 12.6 km^2, representing the home range of sloth bear in central India (Yoganand, 2005) and elsewhere in the Indian sub-continent (Joshi et al., 1995; Ratnayeke et al., 2007a). Appropriate scales at which sloth bears perceived habitat features were identified based on the significant relationship ($p<0.05$) and the associated value of the Akaike Information Criterion (AIC). Subsequently, the best scale for the covariates was selected based on the lower AIC value. Upon identifying the best-fitting scale-specific covariates, I ran models with all possible combinations along with distance-based covariates and photo-capture rates of humans and livestock.

Table 4.1. List of covariates used in generalized linear mixed models to understand the habitat use by sloth bears in Sanjay Tiger Reserve, Madhya Pradesh, India, during 2016-2018.

A. Environmental covariates (with abbreviations)	**Description**	**Range of values**	**Area**
Land use land cover (LULC) classes	LULC was classified into six classes, i.e. dense forest, open forest, scrubland, barren land, agricultural land and water body. The area (in km^2) of each class was calculated and extracted for four spatial scales (500m, 1km, 1.5km and 2km) at each camera trap location in Arc GIS 10.2	1 - 6 (Number of classes)	Dense forest- 706.66 km^2 Open forest- 447.20 km^2 Scrubland- 41.81 km^2 Barren land- 66.77 km^2 Agricultural land- 303.25 km^2 Water body- 37.95 km^2

Forest type classes (Supplementary Figure1)	Forest type map was classified into six classes, i.e. sal mixed, mixed, sal forest, bamboo mixed, grassland and non-forest. The area (in km^2) of each class was calculated and extracted for four spatial scales (500m, 1km, 1.5km and 2km) at each camera trap location in Arc GIS 10.2	1 - 6 (Number of classes**)**	Sal mixed- 538.06 km^2 Mixed- 457.54 km^2 Sal forest- 150.03 km^2 Bamboo mixed- 55.65 km^2 Grassland- 7.28 km^2 Non-forest- 390.02 km^2
Distance to the water source (in km)	Euclidean distance to the nearest water source (natural or artificial) from each camera trap location was calculated using the Spatial analyst tool in Arc GIS 10.2	0 - 15	**-**
Ruggedness index	Ruggedness Index was generated from DEM (Digital Elevation Model) by the roughness index tool of 3D Analyst extension and extracted for four spatial scales (500m, 1km, 1.5km and 2km) at each camera trap location in Arc GIS 10.2	9.21 - 27.61	**-**
B. Anthropogenic covariates (with abbreviations)			
Human photo-capture rate (#/total effort)	The number of total photographs of humans obtained from each camera trap was counted and divided by the total effort (trap-days or trap-nights) of each camera trap	0-11.87	**-**
Livestock photo-capture rate (#/total effort)	The number of total photographs of livestock obtained from each camera trap was counted and divided by the total effort (trap-days or trap-nights) of each camera trap	0-3.53	

Distance to the village (in km)	Euclidean distance to the nearest village from each camera trap location was calculated using the Spatial analyst tool in Arc GIS 10.2	0-12	-
Distance to the metal road (in km)	Euclidean distance to the nearest metal road from each camera trap location was calculated using the Spatial analyst tool in Arc GIS 10.2	0-14	-

Zero inflation in ecological datasets, such as counts of abundance, species presence/absence and so on, is one of the common issues which needs to be addressed while drawing inferences on species-habitat relationships (Clarke and Green, 1988). Inflation of zeros could be attributed to either true absence of the species due to various ecological effects or false absence, where the species remained undetected due to sampling/observational errors during the study period. For cryptic species, if zero-inflation of count dataset remained unaddressed, inference on ecological relationships would be biased, leading to incorrect/failed management interventions for conservation planning (Mapstone, 1995; Field et al., 2004; Martin et al., 2005). Hence, I tested for the possible issues of zero inflation in the count dataset (counts of photographic records of sloth bears) by performing the Score test (Broek, 1995). In the case of zero inflation, appropriate error distribution, i.e. zero-inflated Poisson (ZIP), would be chosen to account for the excess ‘true zero counts’ and ‘false zero counts’ (Martin et al., 2005). Standard distribution (i.e. Poisson) will be fitted if there is no evidence for zero inflation. The best-selected models were chosen from the candidate model set if $\Delta AIC \leq 2$, following Burnham and Anderson (2002). Uninformative parameters (if any) in best-selected models were reported, though I remained restricted to the statistically significant parameters to draw inferences (Arnold, 2010). I also checked for the overdispersion for all top-ranked models by calculating c-hat (ĉ), i.e. overdispersion parameter. All covariates were Z-transformed and were run through the pair-wise Pearson's correlation tests before analysis. Any value of the correlation coefficient, $r \geq \pm 0.7$, was considered strongly correlated (Dormann et al., 2013), and those correlated covariates were not modelled together. All statistical analyses were carried out using the package "glmmTMB" (Magnusson et al., 2021) and "AICmodavg" (Mazerolle, 2015) implemented in programme R (R Core Team, 2019).

Assessment of microsite habitat features

In support of the GLMM-based findings, which do not account for the fine-scale habitat features, I attempted to quantify two major microsite characteristics, i.e. abundance of fruiting trees and insects (termites and ants) which are relevant to the feeding ecology of sloth bears (Bargali et al., 2004; Rather et al., 2020; Paudel et al., 2022; Pokharel et al., 2022), and compared them between different forest types. In this point-based (camera trap points) survey, I assumed that the intensity of use of such point locations would not always be governed by the microsite characteristics found around the camera trap detection zone; rather, it could be an outcome of the bears' daily movement pathways (Puri et al., 2015). Moreover, vegetation characteristics and microsite habitat feature only around camera trap points may not be adequate to represent or draw inferences on the availability of such fine-scale habitat variables throughout the study area. Hence, I quantified the fruiting trees and above-ground termite mounds (as a surrogate of the abundance of termites and ants) within 10m-radius plots (n=328) laid at each 400m interval of predefined line transects (~2km length) in the core zone of STR, that are periodically (once in a year) walked to estimate the abundance of herbivores as a part of the All India Tiger Estimation program (Jhala et al., 2017). Vegetation plots (10m-radius, n=151) laid around each camera trap location were also surveyed, including locations where camera traps were removed or stolen during the survey period. The number of fruiting trees with ≥2m in height and ≥30cm in girth at breast height (GBH) was recorded as per my field observation of fruit-bearing trees during the survey period. Like the fruiting trees, other trees were counted and grouped. I calculated the proportion (No. of fruiting trees/Total number of trees) availability of fruiting trees in different forest types.

To assess the microsite habitat features in different forest types, first, I checked the normality of the two variables, i.e. proportion of fruiting trees and abundance of termite mounds in 5 forest types, by performing the Shapiro-Wilk test (Shapiro and Wilk, 1965). If the variables showed a non-normal (skewed) distribution, the Kruskal-Wallis test (Kruskal and Wallis, 1952) would be conducted to examine the differences in proportions of fruiting tree availability and termite mound abundance in different forest types. If a statistically significant difference ($p<0.05$) is found, I would further perform a pair-wise Wilcoxon rank-sum test to assess the significant difference(s) between pairs of forest types. However, if the variables are found to be normally distributed, one-way ANOVA (Analysis of Variance) test would be carried out, followed by Tukey HSD (honest significant difference) test as a post-hoc measure to identify the pair-wise differences.

4.3 Results

Camera trapping survey

In 2017, a total of 198 independent photographs (SNP= 122; DWLS= 76) of sloth bear was obtained from STR, with a total effort of 5950 trap-nights (3870 and 2080 trap-nights in SNP and DWLS, respectively). A total of 292 independent photographs of sloth bears were obtained in 2018, out of which 235 and 57 detections were from SNP and DWLS, respectively. The total effort was 6480 trap-nights in 2018. Human detection was high throughout the STR, with 5258 (SNP=3396; DWLS=1862) and 4314 (DWLS=1749; SNP=2565) independent photographs in 2017 and 2018, respectively. In summer, the detections of sloth bears were higher (n=311) than in winter (n=179). However, photographic counts of humans were more during winter (n=5258) than summer (n=4314).

Space use patterns

Our count dataset, i.e., photographic counts of sloth bears, was found to be zero-inflated (chi-square=200.88, df = 1, $p<0.005$), due to which I chose ZIP distribution for further analyses.

Different environmental covariates significantly influenced the occurrence of sloth bears, i.e., sal mixed, sal forest, mixed, bamboo mixed, dense forest, open forest, barren land, scrubland and RI at different spatial scales (Table 4.2).

Table 4.2. Best-selected spatial scales (as per Akaike Information Criterion (AIC) value and level of significance ($P<0.05$)) of different environmental covariates to model the occurrence of sloth bear in Sanjay Tiger Reserve, Madhya Pradesh, India during 2016-2018; Covariates considered: salmixed_500m- Area of sal mixed forest at 500m spatial scale, salforest_1km- Area of sal forest at 1km spatial scale, openforest_1km- Area of open forest at 1km spatial scale, barrenland_1km- Area of barren land at 1km spatial scale, bamboo_1.5km- bamboo mixed forest at 1.5km spatial scale, RI_1.5km- Ruggedness index values at 1.5km spatial scale, mixed _2km- Area of mixed forest at 2km spatial scale, denseforest_2km- Area of dense forest at 2km spatial scale, scrubland_2km- Area of scrubland at 2km spatial scale.

Model description	**AIC**	**Estimates**	**Standard error (SE)**	**P-value**
500m scale				
salmixed_500m	1045.4	-0.333	0.099	0.005
1km scale				
salforest_1km	1053.6	0.231	0.103	0.021
openforest_1km	1051.4	-0.225	0.099	0.024
barrenland_1km	1031.5	-0.889	0.211	2.63e-05
1.5km scale				
bamboo_1.5km	1051.5	0.215	0.097	0.027
RI_1.5km	1052.1	0.212	0.104	0.041
2km scale				
mixed_2km	1035.5	0.455	0.101	6.07e-06
denseforest_2km	1041.9	0.385	0.101	0.0001
Scrubland_2km	1043.6	-0.3915	0.1102	0.0004

The season had a prominent effect on the occurrence of sloth bears. However, no significant effect (Estimate = -0.023, Standard error = 0.083, $p = 0.786$ (>0.05)) of camera trap effort (trap-nights) on the photographic counts of sloth bear was observed. In top-ranked scale-optimized models (based on ΔAIC), the occurrence of sloth bears was significantly influenced by the area of mixed forest (2km scale), sal forest (1km scale) and barren land (1km scale) (Table 4.3, Figure 4.1 and Figure 4.2). Sloth bears used mixed forests significantly in both seasons (Figure 4.3). Distinct seasonal effects of two anthropogenic covariates, i.e., human occurrence and distance to the nearest village, were also observed (Table 4.4, Figure 4.1 and Figure 4.3). Regardless of seasons, distance to the nearest village had a marginally significant (p=0.05) positive effect on the sloth bear occurrence (Table 4.4, Figure 4.1 and Figure 4.2). However, sloth bears were frequently detected in areas close to the villages and camera trap locations with high human photo-capture rates during winter (Table 4.4 and Figure 4.3). The scale-optimized best-selected models showed no evidence of overdispersion (c-hat value ranged from 0.74 to 0.76).

Table 4.3. Details of models with best-selected scale-specific covariates (ΔAIC≤2) to determine the relationship between environmental and anthropogenic covariates with sloth bear occurrence in Sanjay Tiger Reserve, Madhya Pradesh, India, during 2016-2018; Covariates considered: mixed_2km- area of mixed forest at 2km spatial scale, season- two seasons i.e. winter and summer, salforest_1km- area of sal forest at 1km spatial scale, human_CR- Photographic capture rate of humans from camera traps, vildist_point- Euclidean distance to the nearest village, and barrenland_1km- area of barren land at 1km spatial scale. AIC- Akaike Information Criterion, ΔAIC-Difference between AIC values, and df- degrees of freedom.

Model Name	**Model description**	**AIC**	**ΔAIC**	**df**	**c-hat (overdispersion parameter)**
Model 1	~ mixed_2km * season + salforest_1km + human_CR * season + vildist_point * season	993.119	0.00	11	0.74
Model 2	~barrenland_1km+salforest_1km+human_CR*season+vildist_point*season	993.709	0.60	10	0.76

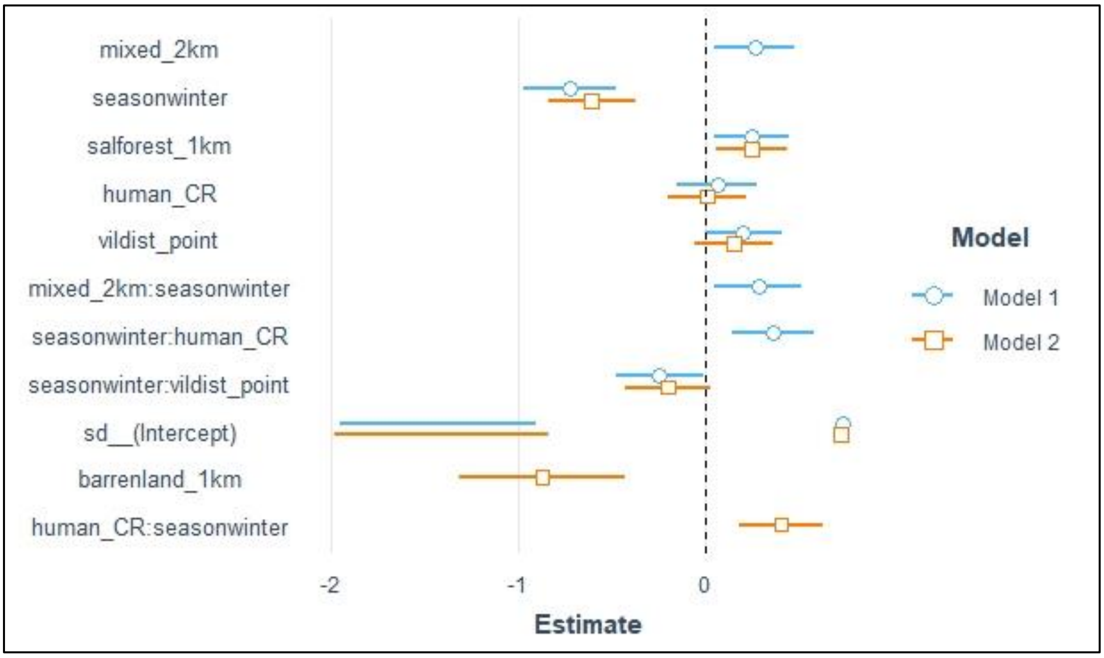

Figure 4.1. Visual representation of estimates of fixed effect covariates and associated 95% confidence interval, obtained from top-ranked scale-optimized models to examine the sloth bear space use in Sanjay Tiger Reserve, Madhya Pradesh, India, during 2016-2018. Covariates considered: mixed_2km- area of mixed forest at 2km spatial scale, salforest_1km- area of sal forest at 1km spatial scale, human_CR- Photographic capture rate of humans from camera traps, vildist_point- Euclidean distance to the nearest village, barrenland_1km- area of barren land at 1km spatial scale, mixed_2km:seasonwinter- mixed forest during season winter, seasonwinter:human_CR- Photographic capture rate of humans from camera traps during winter, and seasonwinter:vildist_point- Euclidean distance to the nearest village during winter. sd_(Intercept)- Estimates of intercepts of the models.

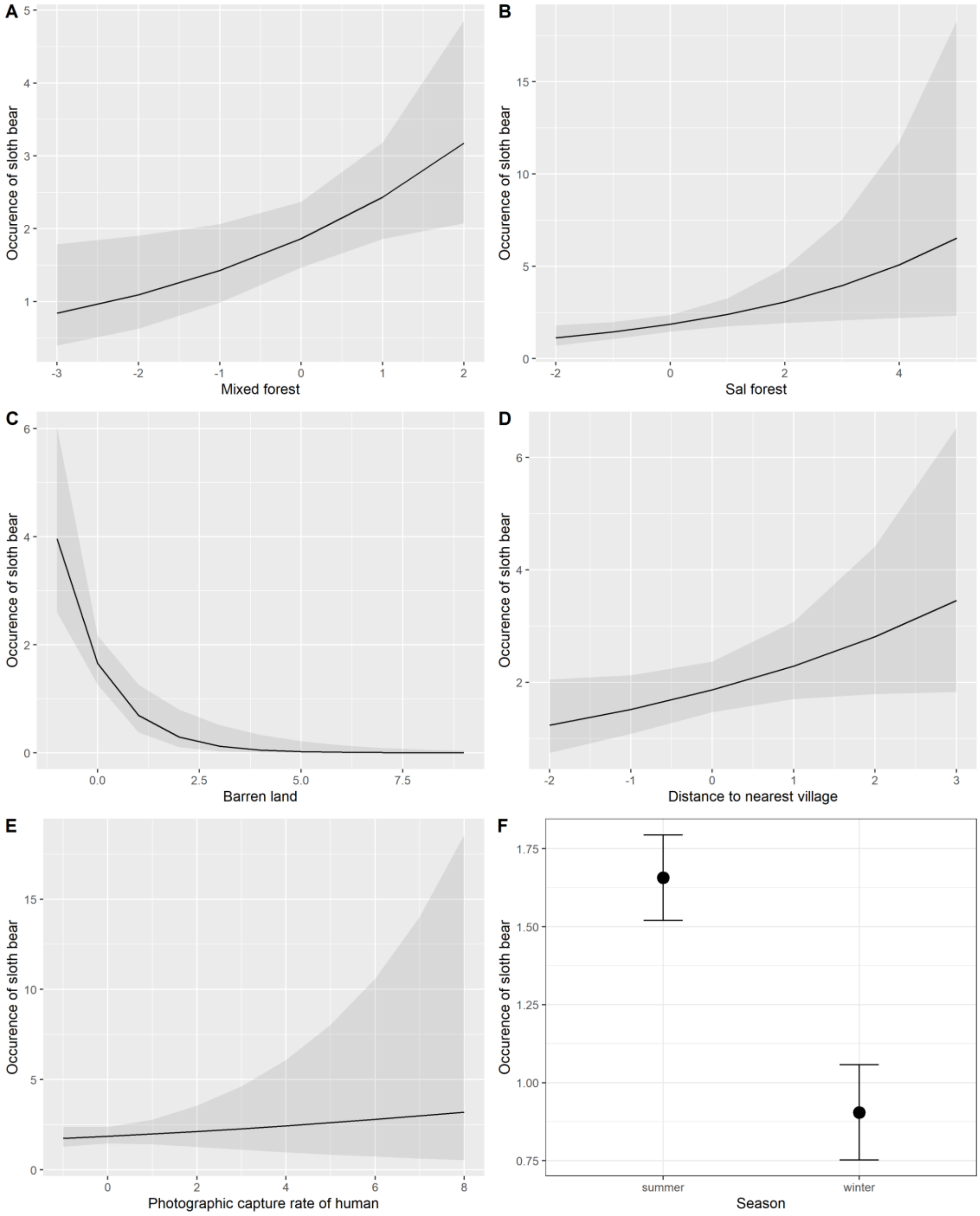

Figure 4.2. Effect of mixed forest (**A**), sal forest (**B**), barren land (**C)**, distance to nearest village (**D**), photographic capture rate of human (**E**) and season i.e. summer and winter (**F**) on the predicted occurrence of sloth bears in Sanjay Tiger Reserve, Madhya Pradesh, India, during 2016-2018, based on the top-ranked models; Area of mixed forest, sal forest and barren land were at 2km, 1km and 1km scale, respectively; All covariates were shown in standardized scale (Z-transformed).

Table 4.4. Estimates of fixed effect covariates, associated standard error (SE) and P-values for covariates from top-ranked models (models within ΔAIC≤2) to determine the space use by sloth bear in Sanjay Tiger Reserve, Madhya Pradesh, India, during 2016-2018. Covariates considered: mixed_2km- area of mixed forest at 2km spatial scale, salforest_1km- area of sal forest at 1km spatial scale, human_CR- Photographic capture rate of humans from camera traps, vildist_point- Euclidean distance to the nearest village, barrenland_1km- area of barren land at 1km spatial scale, mixed_2km:seasonwinter- mixed forest during season winter, seasonwinter:human_CR- Photographic capture rate of humans from camera traps during winter, and seasonwinter:vildist_point- Euclidean distance to the nearest village during winter.

Covariates	**Estimates**	**Standard error (SE)**	**P value**
mixed_2km	0.266	0.110	0.015
salforest_1km	0.251	0.102	0.014
human_CR	0.067	0.109	0.536
vildist_point	0.205	0.106	0.052
barrenland_1km	-0.871	0.224	0.0001
seasonwinter	-0.725	0.126	9.70e-09
mixed_2km:seasonwinter	0.284	0.117	0.015
seasonwinter:human_CR	0.366	0.113	0.001
seasonwinter:vildist_point	-0.243	0.118	0.039

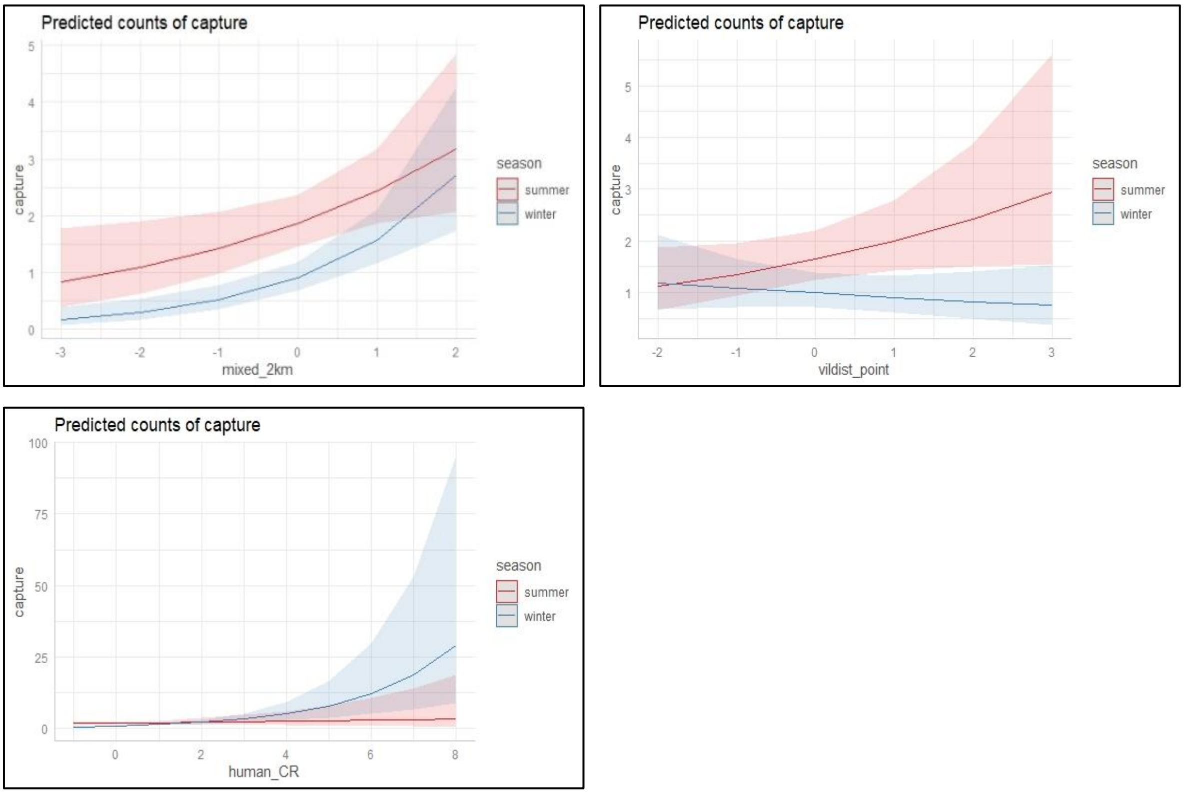

Figure 4.3. Seasonal effect of mixed forest, distance to the nearest village and photographic capture rate of humans on the occurrence of sloth bears obtained from the top-ranked models, in Sanjay Tiger Reserve, Madhya Pradesh, India, during 2016-2018. Abbreviations of covariates: capture- Photographic counts of sloth bear, mixed_2km- area of mixed forest at 2km spatial scale, vildist_point- Euclidean distance to the nearest village, human_CR- Photographic capture rate of humans from camera traps.

Assessment of microsite habitat features

Thirteen species of fruiting trees (Table 4.5) were recorded from 479 vegetation plots. The mixed forest harbours the highest density (139.22/hectare) of fruiting trees, followed by sal mixed (62.36/hectare), sal forest (27.59/hectare), bamboo mixed (2.66/hectare) and grassland (0.8/hectare). Similarly, the density of termite mounds was highest (11.83/hectare) in mixed forest, followed by sal mixed (5.52/hectare), sal forest (4.45/hectare), bamboo mixed (0.332/hectare) and grassland (0.332/hectare).

However, due to the low sample size, the density of fruiting trees and termite mounds in bamboo mixed forest and grassland were not used to draw the inference. Tendu (*D. melanoxylon*) occurred at the highest densities (135.83/hectare), followed by char (*B. cochinchinensis*) (40.55/hectare), mahua (*M. longifolia*) (32.78/hectare), jamun (*Syzygium cumini*) (7.31/hectare), amaltas (*Cassia fistula*) (7.25/hectare) and bhilwa (*Semecarpus anacardium*) (4.05/hectare).

Table 4.5. Details of fruiting trees and termite mounds, recorded in different forest types from vegetation plots (n=479) in Sanjay Tiger Reserve, Madhya Pradesh, India, during 2017-2018.

Fruiting tree species	**Mixed (n=202)**	**Sal mixed (n=150)**	**Sal forest (n=104)**	**Bamboo mixed (n=15)**	**Grassland (n=8)**
Tendu (*Diospyros melanoxylon*)	1356	470	185	25	7
Char (*Buchanania cochinchinensis*)	291	210	99	6	4
Mahua (*Madhuca longifolia*)	238	163	85	6	1
Jamun (*Syzygium cumini*)	88	13	9	0	0
Amaltas (*Cassia fistula*)	69	23	14	3	0
Bhilawa (*Semecarpus anacardium*)	12	32	17	0	0
Aola (*Emblica officinalis*)	19	15	3	0	0
Kosom (*Schleichera oleosa*)	4	8	3	0	0
Ghont (*Ziziphus xylopyrus*)	12	0	0	0	0
Bel (*Aegle marmelos*)	2	2	0	0	0

Bargad (*Ficus benghalensis*)	2	1	0	0	0
Dumar (*F.racemosa*)	0	1	0	0	0
Mango (*Mangifera indica*)	1	0	0	0	0

The proportion of fruiting trees and termite mound abundance in different forest types showed a non-normal distribution (p<0.05; hence the observed distribution significantly differed from the normal distribution). The Kruskal-Wallis test indicated significant differences of proportion of fruiting trees [Kruskal-Wallis chi-squared = 154.07; df = 4; p-value <0.005 (2.2e-16)] and termite mound abundance [Kruskal-Wallis chi-squared = 16.03; df = 4; p-value <0.005 (0.003)], between different forest types (see also Figure 4.4 and 4.5). Subsequently, the multiple pair-wise Wilcoxon rank sum test revealed significant differences in the proportion of fruiting trees between three forest types (i.e. mixed, sal mixed and sal forest) (Table 4.6). Similarly, the abundance of termite mounds significantly differed between mixed and sal mixed forests.

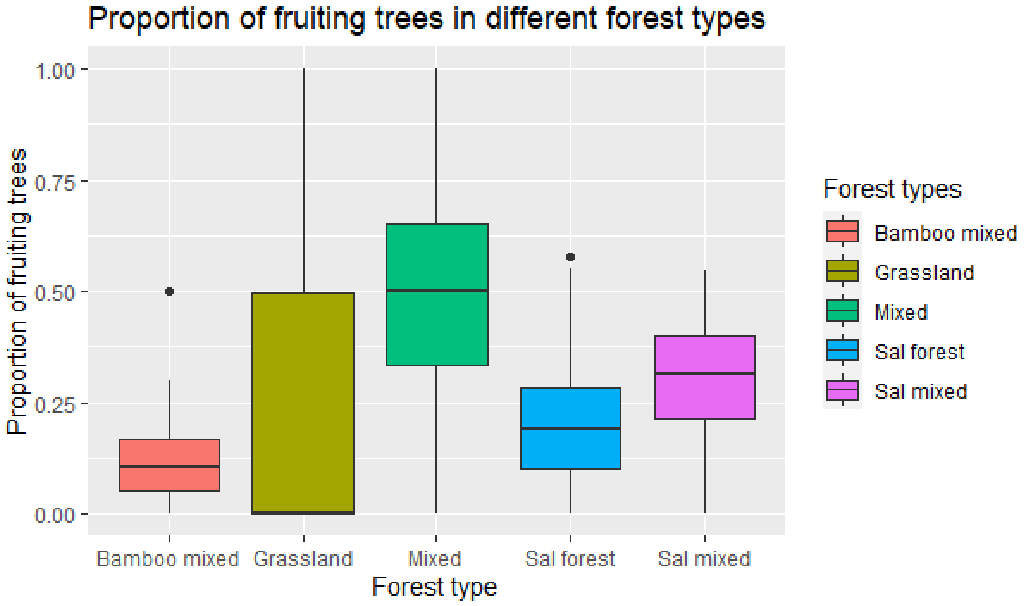

Figure 4.4. Proportion of fruiting trees (Total no. of fruiting trees/total number of trees) in different forest types (bamboo mixed, grassland, mixed, sal forest and sal mixed), recorded from vegetation plots (n=479) in Sanjay Tiger Reserve, Madhya Pradesh, India, during 2016-2018.

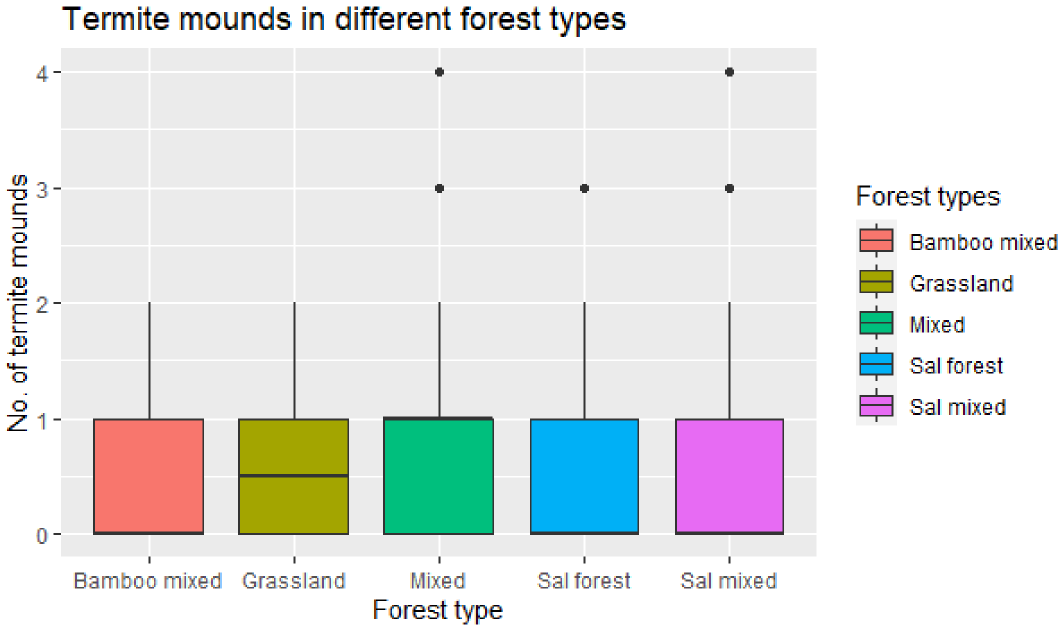

Figure 4.5. Abundance of termite mounds in different forest types (bamboo mixed, grassland, mixed, sal forest and sal mixed), recorded from vegetation plots (n=479) in Sanjay Tiger Reserve, Madhya Pradesh, India, during 2017-2018.

Table 4.6. Results of multiple pair-wise Wilcoxon rank sum test to test the significant differences ($p<0.05$) of proportions of fruiting trees and termite mound abundance between multiple pairs of different forest types in Sanjay Tiger Reserve, Madhya Pradesh, India, during 2017-2018.

Proportions of fruiting trees	**Bamboo mixed**	**Grassland**	**Mixed**	**Sal forest**	**Sal mixed**
Bamboo mixed	-	0.492	9.0e-07	0.065	8.2e-05
Grassland	0.492	-	0.065	0.287	0.176
Mixed	9.0e-07	0.065	-	2e-16	2.8e-16
Sal forest	0.05	0.287	2e-16	-	4.0e-09
Sal mixed	8.2e-05	0.176	2.8e-16	4.0e-09	-
Abundance of termite mounds	**Bamboo mixed**	**Grassland**	**Mixed**	**Sal forest**	**Sal mixed**
Bamboo mixed	-	0.648	0.170	0.648	0.648
Grassland	0.648	-	0.648	0.849	0.699
Mixed	0.170	0.648	-	0.086	0.004
Sal forest	0.648	0.849	0.086	-	0.648
Sal mixed	0.648	0.699	0.004	0.648	-

4.4 Discussion

This study sheds light on the space use patterns of sloth bears in STR, which is facing substantial anthropogenic pressure. My findings identified the different habitat types and anthropogenic factors with respect to seasons, which influenced the space-use by sloth bears, and provided baseline information on the space-use pattern of a habitat generalist, less-studied ursid in STR.

The present study’s findings broadly agree with other studies on sloth bears in the Indian sub-continent. In STR, sloth bears occurred in mixed and sal forests more often than other forest types, which could be attributed to the increased foraging opportunities in these forest types. Habitat preference as a function of foraging opportunity has been studied globally for different species, including bears (Brodeur et al., 2008; Nielsen et al., 2010; Milleret et al., 2018; Twynham et al., 2021); however, there could also be a trade-off between functional responses (foraging and cover) while selecting habitats (Mysterud and Ims, 1998). Sloth bear shows a distinct seasonal pattern in its diet selection by sustaining (primarily) on sugar-rich fruits during summer and a varied proportion of insects (mainly ants and termites) throughout the year (Joshi et al., 1997; Bargali et al., 2004; Yoganand, 2005; Baskaran et al., 2015; Palei et al., 2020; Rather et al., 2020). In STR, I found mixed forest harboured more fruiting trees and termite mounds than sal mixed and sal forest. In agreement with this finding, the proportion of fruiting trees also significantly differed between mixed forest, sal mixed forest and sal forest. Perhaps due to the relatively high number of fruiting trees and termite mounds in the mixed forest, sloth bears used more of this forest type, irrespective of seasons, as I found a strong positive relationship between sloth bear occurrence and mixed forest in both seasons. In Panna National Park of central India, sloth bears significantly preferred dense forest with miscellaneous tree species (or dense mixed forest) due to the year-round high availability of fruiting trees and insects (Yoganand, 2005). A similar observation was also reported from Sri Lanka, where sloth bears preferred areas having high tree density (high forest) with a conceivably high number of fruiting trees and termites (Ratnayeke et al., 2007a). However, preference for sal forest by sloth bear in STR could also be attributed to the termite mounds and perhaps underground colonies of termites and ants. In north Bilaspur Forest Division (NBFD), central India, Akhtar et al. (2004) found that sloth bear signs were more common in sal forest than in sal mixed and mixed forest. Sloth bears in Nepal’s Royal Chitwan National Park, sustained primarily on termites in the upland sal forest during wet season, where the number of termite mounds was higher than in the alluvial grasslands (Joshi et al., 1997; Garshelis et al., 1999). However, the high number of termite mounds and associated feeding signs (ground holes and mound holes) did not necessarily indicate the intensive use of sal forest by sloth bears, rather it could be an indication of the availability of termites and ants and could be related to the inaccessibility of alluvial grasslands due to flood during wet season (Joshi et al., 1997; Garshelis et al., 1999). Ground holes (made by sloth bears) in sal forest could also be attributed to the presence of subterranean termite mounds, which is difficult to quantify. I only recorded the number of visible above-ground termite mounds as a proxy for the abundance of termites and ants; thus, I speculated that relatively high occurrence of sloth bear in sal forest could be primarily due to the availability of termites and partly, fruiting trees. In STR, abundance of termite mounds differed between mixed and sal mixed forest, but remained insignificant between sal mixed and sal forest and marginally significant (p=0.06) between mixed and sal forest. This perhaps added further support to explain the high occurrence of sloth bears in sal forest due to the availability of termites. Nevertheless, apart from termite mounds, underground colonies of ants and termites in sal forest (and likely in other forest types as well), which was not quantified, may influence the space use by sloth bears. However, despite the presence of fruiting trees and termite mounds, sal mixed forest could not influence (evident from top-ranked models) the occurrence of sloth bears significantly. I believe this could be a statistical artefact, or perhaps due to the widespread availability of sal mixed forest (Table 4.1) in this study area and the resultant high representation with less variability of this forest type at varied circular buffers (especially at 500m scale) around each camera trap location.

I found that season had a pronounced effect on the photographic capture of sloth bears. In STR, sloth bears were significantly more detected during the summer than in winter. Since the summer is the main fruiting season, sloth bears perhaps explore larger areas in search of fruit-bearing trees and, thus, are frequently exposed to the camera traps (Ratnayeke et al., 2007a). However, during winter, sloth bears primarily subsist on termites and ants (Joshi et al., 1997; Bargali et al., 2004; Yoganand, 2005; Baskaran et al., 2015; Rather et al., 2020), which are ubiquitous and can be found in large quantities in a small area (Redford, 1987), and perhaps caused the relatively restricted movement during winter (Ratnayeke et al., 2007a). Sloth bears give birth to cubs during winter (end of November to February) and restrict their movement to the den sites during the ‘cubbing’ period (Joshi et al., 1999; Yoganand, 2005), which may also have reduced the detection of sloth bears during winter.

Sloth bears, in general, avoid human disturbances or areas close to habitations and degraded lands (Akhtar et al., 2004; Yoganand, 2005; Ratnayeke et al., 2007a; Das et al., 2014; Babu et al., 2015; Puri et al., 2015; Paudel et al., 2022; Pokharel et al., 2022), and my findings also broadly corroborated these previous studies. However, in some parts of its distribution, sloth bears are forced to live in fragmented and disturbed areas (Akhtar et al., 2004; Bargali et al., 2012) and frequently occur outside PAs (Puri et al., 2015). In STR, sloth bears strongly avoided areas where degraded barren lands were proportionally high (at a 1km scale). Such avoidance could be attributed to the less availability of forage and cover, as evident in other areas (Yoganand, 2005; Ratnayeke et al., 2007a). Barren lands in STR are generally found alongside rivers, streams and areas near villages. These areas are characterized by the least ground cover, primarily due to yearly heavy livestock grazing. Interestingly, Rather et al. (2021b) showed that degraded lands and farmlands could be suitable habitats for sloth bears in a human-dominated landscape of central India. However, the study was solely conducted in a multi-use landscape with a high level of human pressure (buffer zone of Bandhavgarh Tiger Reserve, Madhya Pradesh). It did not include the adjoining least disturbed core zone of the Tiger Reserve, hence may have produced conflicting results. In STR, by and large, the occurrence of sloth bears was positively (marginally, p=0.05) influenced by the distance to the nearest village, but during winter, sloth bears occurred more in areas close to the villages. In human-dominated landscapes of NBFD, sloth bears frequently entered the villages in search of fruits (*Ziziphus mauritiana, Mangifera indica, Syzygium cumini*) and cultivated crops (groundnuts, corns) during winter and monsoon (Akhtar et al., 2004; Bargali et al., 2004), which was also the case for STR. I did not encounter any specimen of *Z. mauritiana* in the surveyed vegetation plots, which indicated the less abundance of this species in the forest or highly patchy in distribution, yet an important food plant species for sloth bears during winter and the onset of summer in central Indian landscape (Bargali et al., 2004; Yoganand, 2005). In an agro-forest landscape of eastern India, sloth bears frequently foraged on cultivated crops and fruits of *Z. mauritiana*, during monsoon and winter (Palei et al., 2020). Perhaps this opportunistic foraging on fruits and crops explained the observed negative relationship between sloth bear occurrence and the distance to the nearest village in STR during winter.

Contrary to my expectations, my findings could not establish any deterministic relationship between the occurrence of sloth bears and human presence (through photographic capture rate) across years. However, sloth bears occurred more during winter in areas where humans were frequently detected. I believe such results were due to the preferential placement of camera traps on forest roads and trails to maximize the detection of sloth bears (and other carnivores). People living inside STR predominantly used the roads and trails while traversing the forest. Moreover, in winter, spatial overlap between sloth bears and humans was found to be higher in comparison to summer in STR (Chaudhuri et al., 2022), which may explain the significant positive relationship between human presence and sloth bear occurrence. Hence, I do not interpret this result as sloth bears preferred areas with high human presence but raised concerns about the high interface between sloth bears and humans and possible conflictual outcomes (Chaudhuri et al., 2022). Sloth bears in STR showed fine-scale spatio-temporal segregation to survive in a human-dominated landscape (Chaudhuri et al., 2022), which is also evident for other bear species across the globe (Chi and Gilbert, 1999; Tollefson et al., 2005; Gaynor et al., 2018; Ladle et al., 2018).

One advantage of my study was to determine the effect of scales in the habitat-use pattern of sloth bear, which is often overlooked while establishing an in-depth species-habitat relationship and thus, may result in incorrect inferences (Wiens, 1989; Poizat and Pont, 1996; Thompson and McGarigal, 2002; Grand et al., 2004; Shirk et al., 2014). A multiscale approach for habitat selection/suitability studies has been adopted for carnivores with special reference to bears in the recent past (Johnson et al., 2005; Ciarniello et al., 2007; Moyer et al., 2008; Martin et al., 2012; Mateo Sánchez et al., 2013; Dar et al., 2021; Rather et al., 2021a). However, the choice of scale (the upper limit) coincided with the minimum home range size of female sloth bears in central

India (Yoganand, 2005), which was much smaller than the upper limit (10 km) of the scale taken by Rather et al. (2021a). It was observed that most of the myrmecophagous mammals have smaller, highly overlapped home ranges (McNab, 1983), perhaps due to either their unique diet, which could be abundant in a relatively small area or lower basal metabolic rate (McNab, 1992). However, compared to other ursids, the home range of sloth bears is much smaller except for a few instances where American black bears roughly corresponded to the home range of sloth bears (Garshelis and Pelton, 1981; Joshi et al., 1995). Small home range sizes and highly overlapped territories of sloth bears in the Indian sub-continent could be attributed to the abundant food resources (termites and ants) across the years. Additionally, less territorial shift as a function of foraging in closely-spaced habitats, which is common in the plain lands of the Indian sub-continent (Joshi et al., 1995), can also cause smaller home ranges than other bears. Although the home range size of sloth bears varied extensively (from ~2 km^2 to ~80 km^2) in the Indian sub-continent, information on ranging patterns of sloth bears in a human-dominated landscape perhaps would better my understanding of its scale-specific space use.

Caveats and limitations

One of the major limitations of this study was the choice of appropriate scale(s), which may have influenced my findings. However, this was purely due to the insufficient knowledge of sloth bears' ranging patterns in human-dominated landscapes. Less spatial coverage could also be an issue, as I could not deploy camera traps close to the villages and other sensitive areas where stealing of cameras was common. I could not adequately sample the abundance of ants and termites in the study area; otherwise, it could have improved my knowledge of the habitat use by sloth bears with respect to food availability, especially in the sal forest. Lastly, this survey was only restricted to two seasons, i.e. summer and winter, due to logistic constraints; hence I could not draw inferences on sloth bear space use during monsoon.

4.5 Conclusion

My findings demonstrated the scale-specific influence of environmental and anthropogenic covariates on the space use patterns of sloth bears with respect to seasons in a human-dominated landscape of central India. Despite seasons, sloth bears preferred areas with a higher proportion of mixed and sal forest. However, sloth bears were also attracted to villages during winter, probably to forage on fruits and cultivated crops, but strongly avoided degraded barren lands regardless of seasons. I believe the present study would provide insights into the space use pattern of sloth bears in a human-dominated landscape where sloth bear-human conflict exists. The movement of humans in sloth bear habitats could only be restricted if the primary focus is targeted toward the voluntary relocation of villages from the core zone of STR. Meanwhile, during the process of village relocation, sensitization of local people to the behaviour of sloth bears (such as sloth bears occurring close to villages during winter, the importance of mixed forest in sloth bear’s survival, and its preferred food items) should be carried out to avoid conflicts and achieve broader conservation goals. Habitat restoration activities in village relocation sites and barren lands with special reference to planting fruiting trees and shrubs (e.g. species of *Ziziphus*) which are less common in the forest, could also be an option. I also specifically emphasize conducting long-term research on sloth bears with sound field (radio-telemetry, intensive camera trapping) and analytical framework to fill the knowledge gap on the ecological aspects of sloth bears in a human-dominated landscape, with an adaptive management approach towards conflict-mitigation.

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PART B: Den Selection by Sloth Bear

4.6 Introduction

Management of specialised habitats such as dens is an essential aspect of conserving lesser-known threatened species. However, this aspect is often overlooked as a major focus of management interventions remains restricted primarily to broad-scale habitat types of concerned species. These specialised habitats (dens in rocky outcrops or escarpments) possess a significant ecological role beyond their area of occurrence at an ecosystem scale (Hunter, 2017) and attract many mammalian and avian species by providing critical breeding ground and resting/nesting/roosting sites (Fitzsimons and Michael, 2017).

Bears are known to hibernate at winter dens (also known as metabolic denning) to reduce energy loss during adverse climatic conditions and associated food scarcity (Tietje and Ruff, 1980; Fowler et al., 2021) as well as for parturition (maternal metabolic denning) (López-alfaro et al., 2013; Fowler et al., 2021). However, the majority of the studies on denning ecology were restricted to the northern temperate bears (Seryodkin et al., 2003; Zhang et al., 2007; Baldwin and Bender, 2008; Elfström et al., 2008; Koike and Hazumi, 2008; Mangipane et al., 2018; Schafer et al., 2018; Wei et al., 2019). On the other hand, four bear species, i.e. sloth bear, sun bear, Andean bear and giant panda, do not use winter dens for hibernation but utilise cavities and crevices (situated at escarpments, rugged areas and rocky outcrops), hollow fallen logs and ground holes for day resting, parturition or both (Joshi et al., 1999; Te Wong et al., 2004; Yoganand, 2005; Akhtar et al., 2007; Zhang et al., 2007; Hall and Swaisgood, 2009; Wei et al., 2019; Van Horn et al., 2021).

Sloth bears use dens at rocky outcrops or escarpments during the daytime to avoid heat stress and threats from conspecifics and large carnivores during parturition (Joshi et al., 1999; Yoganand, 2005; Akhtar et al., 2007). Yoganand (2005) found that, in the central Indian landscape, sloth bears used dens situated at rocky outcrops more than its availability, despite substantial human disturbance (Akhtar et al., 2007). In a human-dominated landscape, management of such habitat features could be critical for the conservation of sloth bear, as these features might have significant effects on overall individual fitness and cub survival, which was evident for other bear species in the context of human disturbance (Swenson et al., 1997; Linnell et al., 2000; White et al., 2001; Evans et al., 2016). However, to my knowledge, little is known about denning ecology of sloth bears in terms of local scale habitat features that influence the den selection, except for a single study conducted in the central Indian landscape by Akhtar et al. (2007). I, therefore, took this opportunity to fill the knowledge gap for sloth bear conservation in a human-dominated landscape. I utilised ground knowledge and indirect signs to locate sloth bear dens which were actively used and potentially available dens, which were less frequently used or unused.

The present study has the following objectives:

1. To identify used and available dens, and

2. to compare microsite characteristics between used and available dens and identify the determinant factors for den selection by sloth bear.

4.7 Materials and methods

Identification of dens

Dens used by sloth bears or “bear dens” were searched in the core zone of STR from 2016 to 2018, based on the prior knowledge of potential dens located at each forest Beat and sloth bear occurrence in those dens. The ground-level staff of the forest department, i.e. Forest guards, were contacted, and information on possible sloth bear dens and the frequency of bears occupying those dens at respective forest Beats were collected. The search effort was restricted to forest Beats due to the feasibility of locating dens, as it was expected that the staff of the concerned beat would be well equipped with ground knowledge. Apart from this, potential dens were also located opportunistically or during the carnivore sign survey at each forest Beat. Since sloth bears are known to actively use cavities or crevices situated at hillocks, dry streambeds and rugged areas in this landscape and in general (Akhtar et al., 2007; Yoganand et al., 2013), I primarily attempted to locate these types of dens. However, I could not identify the dens that were used mainly for parturition; hence, my study focused on microsite characteristics of den sites, which may also include natal dens, apart from day-resting dens. Also, other potential den sites such as hollow tree trunks, patches of lantana or dense shrub were not searched for, either due to too few locations or a high degree of ambiguity associated with den use by sloth bears. After encountering each potential den, the structure of the den and bear signs (scats, footprints, dig marks, claw marks) in and around the den were visually inspected. Potential dens were defined as cavities large enough to accommodate at least one adult sloth bear to enter and sit/lie inside the cavity (Akhtar et al., 2007). Bear dens were periodically monitored during

2016-2018 and classified as "used" by sloth bears based on frequent sightings and indirect evidence of bears in and around den sites. However, upon finding an apparently suitable den with no definite or very few signs of sloth bear, it was classified as a potential but unused/relatively less used or "available" den.

Collection of covariates

Site-specific covariates around the dens (both used and available) were collected from July to December 2019. Conducting a ground survey during this period was chosen due to the less frequent use of cavities/crevices by sloth bears, as observed in Panna National Park (PNP), central India, based on a radio-telemetry study (Yoganand, 2005). Dens were visited during daylight hours (08:00 am to 04:00 pm) and approached closely if the den was found unoccupied by a sloth bear. For each den site, twenty variables were collected (Table 4.7). At 15m and 5m-radius from the main entrance/opening of each den, the number of trees (≥ 2m height and ≥ 30cm girth at breast height [GBH]) and shrubs (0.90m≥height<2m) were counted, respectively. For trees, the basal area from GBH was calculated (in m^2). The basal area for fruiting trees contributing to the sloth bear's diet (≥1%) was also calculated separately. As a measure of cover, shrub abundance was categorised into four classes, i.e., low (1%-25%), moderate (26%-50%), high (51%-75%), and very high (76%-100%), based on counts at each plot. The steepness of the route to the main entrance of the den was determined based on the slope between the valley and the main entrance of the den and categorised into four classes (plain [0°-15°], moderate slope [16°-30°], slope [31°-45°] and steep slope [>45°]). Since I expected dens to be located in rugged terrains, I used slope and ruggedness index (RI; Riley, 1999) at a 50m-buffer around each den, calculated from ASTER GDEM (Global digital elevation map) by using ArcGIS 10.2 (ESRI, R., 2011). The visibility around the dens was measured by using a concealment board of 1m tall, with 10cm coloured graduations (Noon, 1981), in three cardinal directions from the main entrance of the den from a distance of 10m, 30m and 50m. Proportions of visibility in 3 directions were averaged for each den.

Table 4.7. Descriptions of site-specific covariates that were collected at den sites of sloth bear in Sanjay Tiger Reserve, Madhya Pradesh, India, in 2019.

Type	Variables	Description
Continuous	No. of openings	The main entrance and adjacent openings (if any) of a single den
Continuous	Elevation (in m)	Elevation at den site (in front of the main entrance), measured by GPS
Continuous	Height from the valley (in m)	The subtracted value between the elevation of the den (at the main entrance) and the elevation of the valley (the relatively plain area around the den site)
Categorical	Steepness of route	Steepness from the valley to the main entrance of the den, expressed as four categories (plain [0°-15°], moderate slope [16°-30°], slope [31°-45°] and steep slope [>45°])
Continuous	Temperature difference (°C)	The temperature difference between inside and outside of the den
Continuous	Basal area (in m^2) of trees(at 15m radius from the main entrance)	Basal area was calculated from the GBH of each tree (including fruiting trees) multiplied by the total number of trees
Continuous	Basal area (in m^2) of fruiting trees (at 15m radius from the main entrance)	Basal area was calculated from the GBH of each fruiting tree multiplied by the total number of fruiting trees
Categorical	Shrub abundance	The number of shrubs was counted at 5m-radius and ranked (1-4) based on abundance
Continuous	Canopy cover (%)	Visuals estimation of canopy cover around the den
Continuous	No. of fallen logs	Number of fallen logs (>10cm in diameter) at 15m-radius

Continuous	No. of tree cutting signs	Number of tree cutting at 15m-radius from the den
Continuous	No. of lopping signs	Number of lopping signs at 15m-radius from the den
Continuous	No. of grass/bamboo cutting signs	Number of grass/bamboo cutting signs at 15m-radius from the den
Continuous	No. of livestock dung/pellet	Number of livestock dung/pellet at 15 m radius from the den
Continuous	No. of termite mounds	Number of termite mounds at 15m radius from the den
Continuous	Distance to the nearest village (in m)	Euclidean distance to the nearest village from den calculated by ArcGIS 10.2
Continuous	Distance to the nearest water source (in m)	Euclidean distance to nearest water source from den calculated by ArcGIS 10.2
Continuous	Ruggedness Index (RI)	Terrain ruggedness index was calculated from ASTER DEM and extracted for 50m buffer around each den by ArcGIS
Continuous	Slope (°) at 50m	The slope was calculated from ASTER DEM and extracted for 50m buffer around each den by ArcGIS
Continuous	Visibility at 10m, 30m and 50m	Visibility was measured by using a concealment board from the main entrance of the den, in three cardinal directions

Statistical analyses

Before any statistical test, all continuous variables (n=18) were standardised, and I used the Shapiro-Wilk test to assess the data distribution to ensure normality (Shapiro and Wilk, 1965). For variables with normal distribution, independent samples t-tests were performed, and for variables with non-normal distribution, the Mann-Whitney U test (Mann and Whitney, 1947) was carried out to compare each of the variables for used and available dens. As most of the variables were potentially interrelated, principal component analysis (PCA) was carried out in order to reduce the dimensionality and eliminate the multicollinearity amongst variables. To best account for the den use, I selected the uncorrelated principal components (PCs) based on eigenvalues >1 (Kaiser, 1970) and the proportion of cumulative variance explained by those chosen PCs. Subsequently, I assessed the contribution of variables of selected PCs by using varimax rotation to get loading coefficients. Values of loading coefficients ≥0.5 were only taken for the interpretation of den use (Zhang et al., 2007). Furthermore, the selected PCs and two categorical variables (steepness of route and shrub abundance) were used to develop generalised linear models (GLM) to identify the minimal set of variables that best influenced the den use (Zhang et al., 2007; Van Horn et al., 2021). The choice of GLM framework allowed us to include categorical variables in modelling. Candidate models were compared based on Akaike Information Criterion, corrected for small sample size (AIC_c), and inference(s) were drawn for models within ≤2 ΔAIC_c (Burnham and Anderson, 2002) with significant p values (<0.05) of variables in those models (Arnold, 2010). Uninformative parameters were reported as well; however, model averaging was not performed in the presence of uninformative parameters included in top-ranked models (Arnold, 2010). All analyses were carried out in program R (R Core Team, 2019).

4.8 Results

From 2016 to 2018, a total of 70 dens were identified in the core zone of STR, out of which 35 were categorised as “used” and the remaining as “available” (Figure 4.6). All dens were pre-existing cavities and crevices below or between boulders situated at hillocks, rugged areas, or dry streambeds.

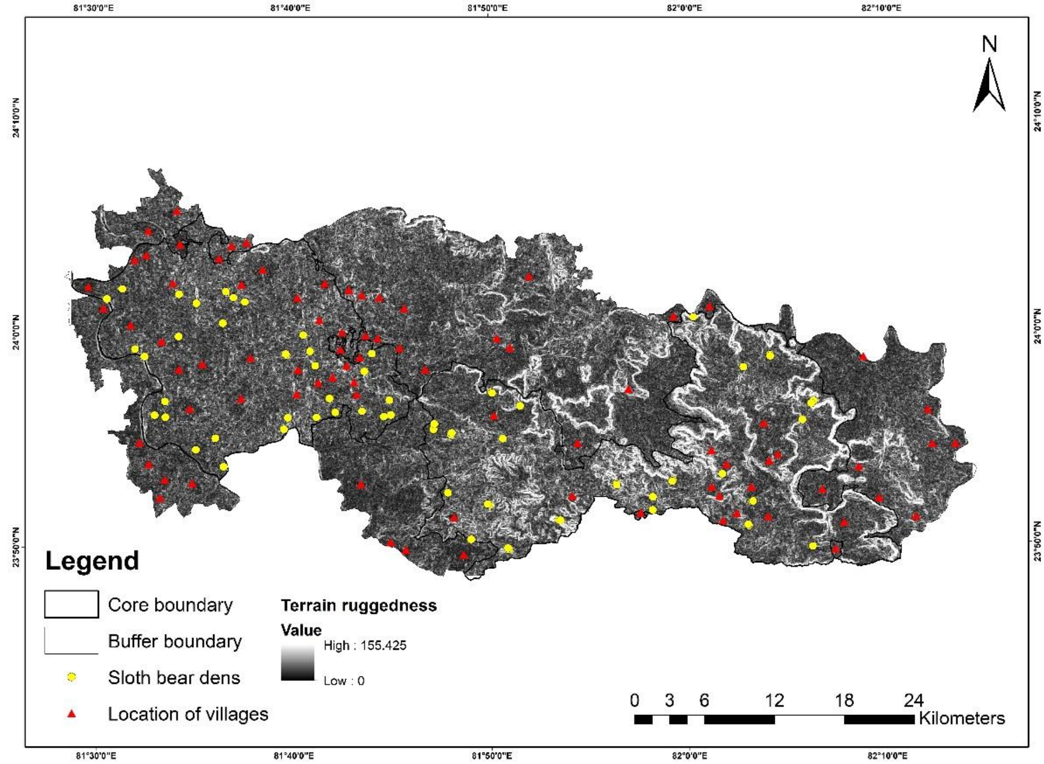

Figure 4.6. Locations of dens at the core zone of Sanjay Tiger Reserve, Madhya Pradesh, India, along with locations of villages.

Out of 20 variables, RI, slope and visibility (at 10m, 30m and 50m) were significantly different between used and available dens (Table 4.8, Figure 4.7), indicating the choice of better-concealed dens in rugged terrain. The number of cutting signs and presence of livestock dung/pellet also significantly differed between used and available dens; however, median values of both the variables were found to be 0 (range of tree cutting signs: 0-3 for used and 0-6 for available; range of livestock dung/pellet: 0-15 for used and 0-3 for available; see Table 4.8), which deterred us from drawing inference. Similarly, the canopy cover was marginally different (t= 1.79, p =0.08) between used and available dens.

Table 4.8. Results of univariate tests for site-specific covariates of used and available dens in Sanjay Tiger Reserve, Madhya Pradesh, India, in 2019. *“.”, “*” and “**” denote P values <0.1, <0.05 and <0.005, respectively.*

Variables	**Used**		**Potentially available/unused**		**t**	**U**	**P value**
	Mean (STD)	**Median (range)**	**Mean (STD)**	**Median (range)**			
No. of openings	-	1 (1-6)	-	1 (1-4)	-	546.5	0.34
Elevation (in m)	-	498 (361-677)	-	488 (368-672)		611.5	0.99
Height from the valley (in m)	-	12 (0-50)	-	7 (0-50)		505	0.21
Temperature difference (°C)	2.22 (1.91)	-	1.946 (1.552)	-	-0.66	-	0.51
Basal area (in m^2) of trees (at 15m radius from the entry)	-	69397.35 (20224.27-198011.2)	-	58760.11 (17042.21-119469.8)	-	489	0.15
Basal area (in m^2) of fruiting trees (at 15m radius from the entry)	-	6764.73 (0-53851.52)	-	5580.19 (0-59585.67)		616	0.97
Canopy cover (%)	35.68 (17.20)	-	28.51 (16.25)	-	1.79	-	0.08$^{\cdot}$
No. of fallen logs	-	0 (0-2)	-	0 (0-2)	-	628.5	0.77
No. of tree cutting signs	**-**	**0 (0-3)**	**-**	**0 (0-6)**	**-**	**766.5**	**0.02***
No. of lopping signs	-	0 (0-7)	-	0 (0-11)	-	614	0.98
No. of grass/bamboo cutting signs	-	0 (0-15)	-	0 (0-25)	-	616.5	0.94
No. of livestock dung/pellet	**-**	**0 (0-15)**	**-**	**0 (0-3)**	**-**	**757**	**0.03***
No. of termite mounds	-	0 (0-2)	-	0 (0-3)	-	682.5	0.32
Distance to the nearest village (in m)	-	2184.03 (863.77-5904.74)	-	2333.45 (941.54-5785.26)	-	590	0.79

Distance to the nearest water source (in m)	-	930 (30-3064.44)	-	1027.03 (30-3043.04)	-	661.5	0.57
Ruggedness Index (RI) at 50m		**16.99 (7.75-44.79)**	**-**	**14.15 (6.59-41.24)**	**-**	**430**	**0.03***
Slope (degree) at 50m		**9.33 (1.62-21.28)**	**-**	**5.34 (1.14-27.82)**	**-**	**436**	**0.04***
Visibility at 10m		**0.2 (0-0.33)**		**0.23 (0-0.35)**	**-**	**793**	**0.03***
Visibility at 30m		**0.05 (0-0.18)**		**0.08 (0-0.35)**		**809**	**0.02***
Visibility at 50m		**0 (0-0.10)**		**0 (0-0.26)**		**818.5**	**0.003****

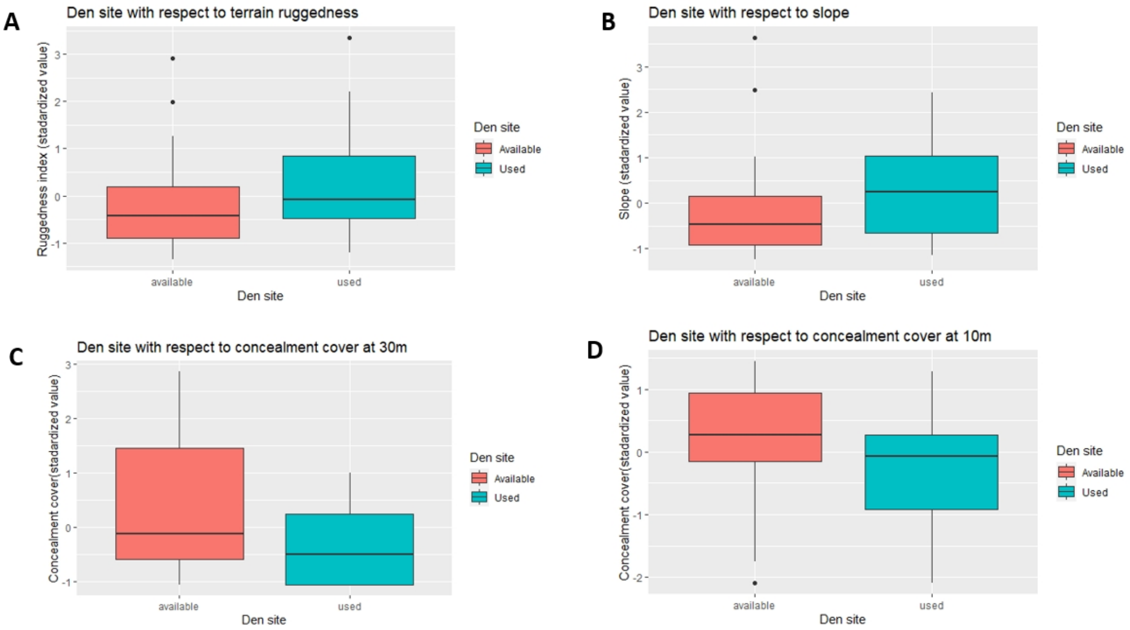

Figure 4.7. Boxplots showing den sites (used and available) with respect to **A**. ruggedness index (RI), **B**. slope, **C**. concealment cover at 30m and **D**. concealment cover at 10m, in Sanjay Tiger Reserve, Madhya Pradesh, India, during 2019; Standardised values of covariates were presented here.

Results of PCA showed that the first six PCs explained 64% of the total variation of data (Table 4.9). PC1 and PC2 together were found to explain 30% of the variance (Figure 4.8). Based on values of loading coefficients, it was found that height from the valley (0.54), RI (0.52), slope (0.55), visibility at 10m (-0.51) and 30m (-0.51) primarily influenced PC1 and PC2. However, PC3, which explained an additional 10% of the variation, was affected mainly by the number of tree cutting and lopping signs.

Table 4.9. Results of principal component analysis (PCA), aimed at understanding site-specific covariates of used and available dens in Sanjay Tiger Reserve, Madhya Pradesh, India, in 2019.

Variables	**PC1**	**PC2**	**PC3**	**PC4**	**PC5**	**PC6**
No. of openings	-0.05	0.05	0.08	-0.17	**-0.57**	-0.02
Elevation	0.11	-0.14	-0.09	0.01	-0.03	0.50
Height from valley	**0.54**	0.00	0.04	-0.02	0.09	-0.06
Temperature difference	0.05	-0.06	-0.10	0.10	-0.17	**-0.51**
Basal area of trees	0.04	-0.14	0.07	0.42	-0.37	-0.05
Basal area of fruiting trees	0.06	-0.13	0.27	0.10	0.02	-0.37
Canopy cover	0.08	-0.23	-0.30	0.39	0.01	-0.21
No. of fallen logs	-0.03	0.06	-0.10	0.00	**-0.59**	0.05
No. of tree cutting signs	0.10	-0.10	**0.50**	-0.17	-0.05	-0.17
No. of lopping signs	-0.03	0.18	**0.52**	0.06	-0.05	0.01
No. of grass/bamboo cutting signs	-0.29	0.26	0.12	0.38	0.12	0.04
No. of livestock dung/pellet	0.00	-0.12	0.45	0.09	0.10	0.10
No. of termite mounds	0.01	0.02	0.10	**0.53**	0.05	-0.01
Distance to nearest village	0.05	0.13	-0.13	0.29	0.14	0.06
Distance to nearest water source	0.05	-0.11	0.15	0.21	-0.21	0.50
Ruggedness Index (RI) at 50m	**0.52**	0.09	0.02	0.08	0.01	0.07
Slope at 50m	**0.55**	0.05	0.00	0.00	-0.02	0.03
Visibility at 10m	-0.01	**-0.51**	-0.05	0.01	-0.14	-0.01
Visibility at 30m	-0.05	**-0.51**	0.03	-0.15	0.11	0.02
Visibility at 50m	-0.10	-0.43	0.10	0.09	0.16	0.08
Eigenvalue	3.58	2.47	1.95	1.87	1.49	1.33
Proportion of variance	0.18	0.12	0.10	0.09	0.08	0.07
Cumulative proportion of variance	0.18	0.30	0.40	0.49	0.57	0.64

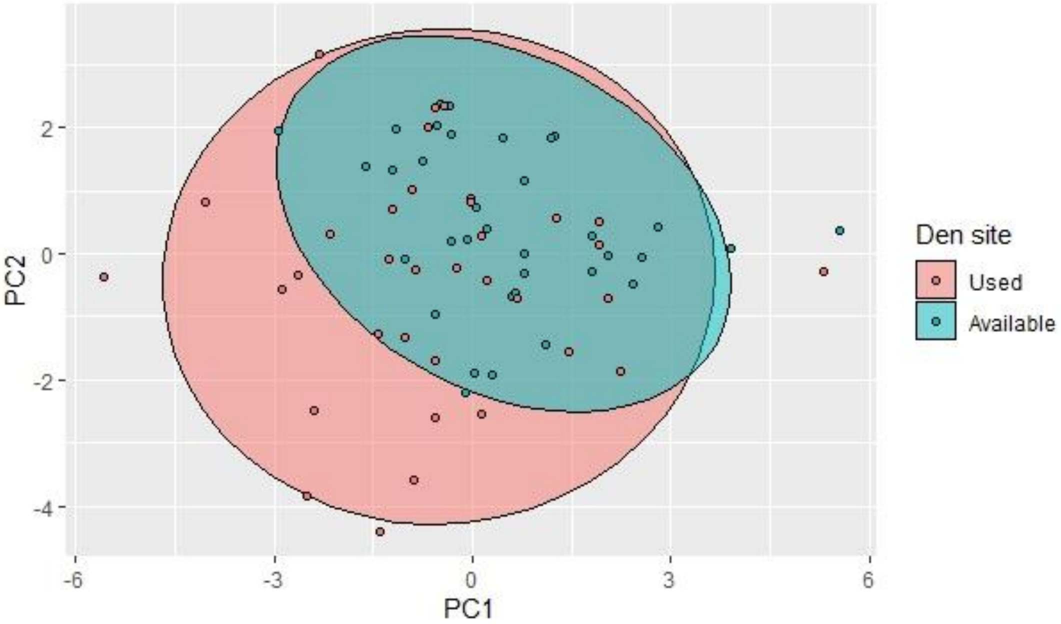

Figure 4.8. Biplot produced from principal component analysis (PCA), showing interrelations between continuous variables collected from den sites and two principal scores (PC1 and PC2, along with the proportion of explained variance) in Sanjay Tiger Reserve, Madhya Pradesh, India.

Although PC1 and PC2 explained a greater amount of variation, I added all 6 PCs in the GLM framework to assess the effects of variables (based on values of loading coefficients ≥0.5) on den site selection by sloth bear. It was found that the best-selected models (ΔAICc ≤ 2) include combinations of PC1, PC2 and PC3 (Table 4.10).

Table 4.10. Models (with ΔAICc ≤ 4) obtained from generalised linear modelling (GLM) framework, using principal components (PC1-PC6) and two categorical covariates (steepness of route and shrub abundance), for predicting determinant factors of den use by sloth bear in Sanjay Tiger Reserve, Madhya Pradesh, India, in 2019.

Models	**logLik**	**AICc**	**ΔAICc**	**weight**
PC1+PC2	-42.125	**90.613**	**0**	**0.330**
PC1+PC2+PC3	-41.296	**91.208**	**0.594**	**0.245**
PC1+PC2+PC3+PC4	-41.282	93.502	2.888	0.078
Steepness of route	-42.528	93.672	3.059	0.072

PC1+PC2+Steepness of route	-40.341	94.015	3.402	0.060
PC2+ Steepness of route	-41.599	94.136	3.523	0.057
PC1+ Steepness of route	-41.676	94.289	3.675	0.053

However, no significant relationship ($P>0.05$) was found for PC3 (Table 4.11). Due to uninformative parameters in the best-selected models, model averaging was not done. Combined with the results of PCA, it was found that terrain complexity (RI and slope), height from the valley and better concealment cover (visibility) predicted the use of den sites by sloth bears (Figure 4.9).

Table 4.11. Estimates, associated standard errors (SE), and P values of variables obtained from selected models (based on ΔAICc and P value) of GLM framework to identify factors influencing den use by sloth bear in Sanjay Tiger Reserve, Madhya Pradesh, India, in 2019.

Variables	**Estimates**	**SE**	**P value**
PC1	**0.394**	**0.165**	**<0.05**
PC2	**0.434**	**0.187**	**<0.05**
PC3	-0.258	0.207	>0.05
PC4	0.033	0.201	>0.05
Steep slope (Steepness of route)	**1.386**	**0.559**	**<0.05**

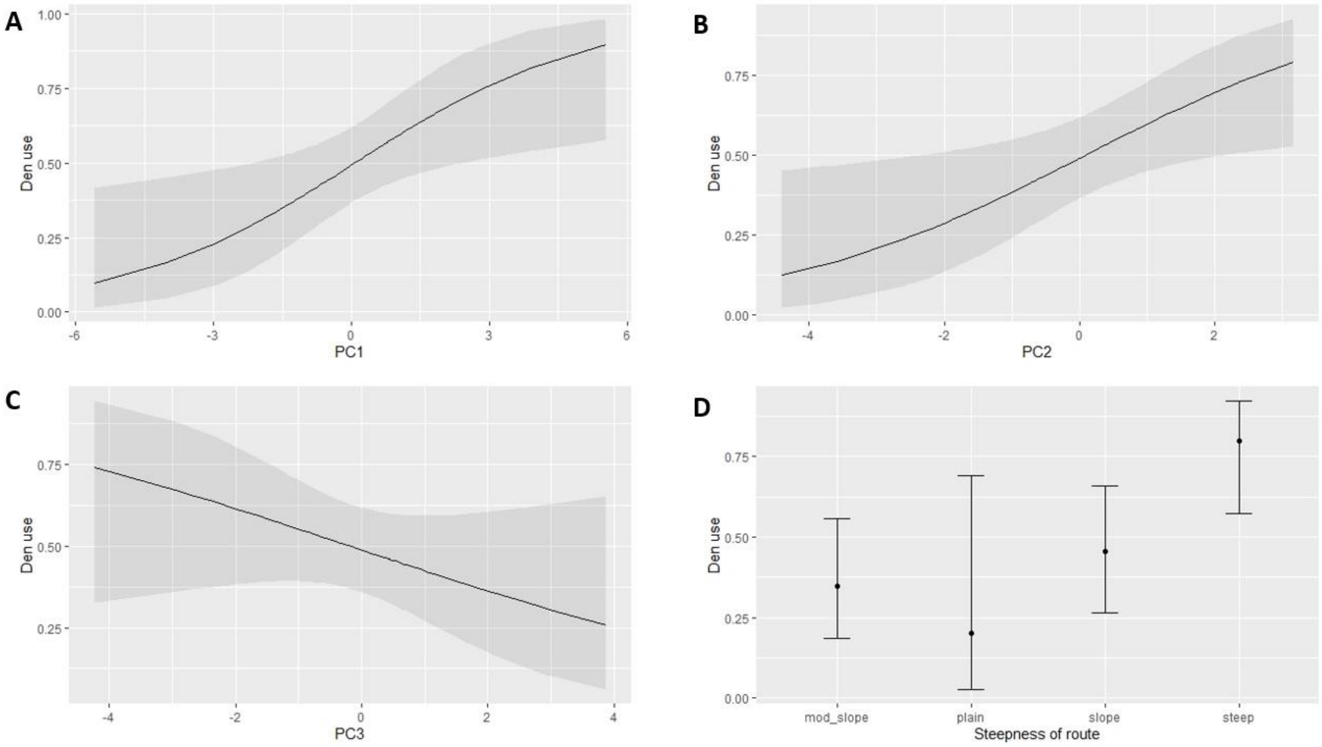

Figure 4.9. Relationship between predicted den use by sloth bears, principal components, i.e. PC1 (**A**), PC2 (**B**) and PC3 (**C**), and steepness of route (**D**) obtained from selected models of the generalised linear framework in Sanjay Tiger Reserve, Madhya Pradesh, India.

4.9 Discussion

The results indicated that den site characteristics were more inclined towards the terrain complexity and associated concealment cover to be used by sloth bears. Day resting dens, as well as natal dens, used by sloth bears, have been documented previously by several authors (Eisenberg and Lockhart, 1972; Laurie and Seidensticker, 1977; Joshi et al., 1999; Yoganand, 2005; Akhtar et al., 2007; Baskaran et al., 2015). Slope and RI were constantly found to contribute to den usage in univariate tests and subsequent GLM, which depicted the importance of more rugged areas around the den. This finding is in agreement with other studies on sloth bears conducted in the central Indian landscape (Yoganand, 2005; Akhtar et al., 2007), as well as other bear species worldwide (Seryodkin et al., 2003; Reynolds-Hogland et al., 2007; Zhang et al., 2007; Baldwin and Bender, 2008; Elfström et al., 2008; Koike and Hazumi, 2008; Wei et al., 2019; Iosif et al., 2020; Van Horn et al., 2021). Preference for higher slopes and rugged terrain could be attributed to the better protection from anthropogenic threats, as these areas were presumably inaccessible to humans and livestock. However, given the wide spatial distribution of human activities in STR, it is logical that bears would prefer secluded areas such as rocky outcrops or escarpments for day resting. A similar observation was also recorded in North Bilaspur Forest Division (NBFD), a human-dominated landscape in central India, where sloth bears denned mostly in hillocks or rugged areas (Akhtar et al., 2007). Sloth bears were also observed using escarpment habitat more than its availability for day resting and parturition in PNP (Yoganand, 2005). Terrain ruggedness and the den’s height from the valley would significantly reduce the visibility around the den, providing sufficient concealment cover for sloth bears. Terrain ruggedness and high concealment cover around the den might also reduce the risk of being exposed to large carnivores (tiger and leopard, in this case), especially during cubbing period, which was reported in the Indian subcontinent (Joshi et al., 1999; Yoganand et al., 2013) and elsewhere for other bear species (Seryodkin et al., 2003). However, due to insufficient information on the predation of sloth bears by tigers and leopards, I believed that den selection in rugged terrain with high concealment cover was more likely to be associated with avoidance of human disturbance.

We found no significant relationship between the proximity of human settlements and den selection by sloth bears in STR. Both used and available dens were located an average of ~2km away from the nearest village. However, in NBFD, bear dens were found to be situated chiefly in close proximity (<1km) to villages, which probably facilitated crop raiding and foraging on fruits (*Ziziphus* sp., *Syzygium* sp., *Mangifera indica* etc.) in and around villages (Akhtar et al., 2007). In general, other bear species are sensitive to human disturbance and prefer to den away from human settlements or high human-used areas (Goodrich and Berger, 1994; Linnell et al., 2000; Gaines, 2003; Waller et al., 2014), although exceptions exist (Baldwin and Bender, 2008). It appears that den selection by sloth bears close to human settlements is primarily driven by the availability of dens and the extent of anthropogenic pressure (in terms of settlements and human movement) on the habitat where bears survive. Human settlements interspersed with small habitat patches perhaps forced bears to rely on these available sites for denning (Akhtar et al., 2004, 2007). However, in the core zone of STR, due to its better protection status compared to NBFD (unprotected forest area), sloth bears can still find denning areas further away from villages and human activities despite widespread human pressure. From univariate tests, I found that used den sites were mostly devoid of tree cutting signs (5 out of 35 used dens, ~14%) and livestock presence (6 out of 35 used dens, ~17%), which was an indication towards the avoidance of disturbed areas for denning. Less number of tree cutting signs (19 out of 70 dens, ~27%) and detection of livestock dung/pellet (20 out of 70 dens, ~29%) around den sites also indicated general inaccessibility to the locations of dens (both used and available).

The presence of food plants and termites at den sites did not seem to affect den use by sloth bears. Akhtar et al. (2007) observed the presence of certain food plants (*Ficus* sp.) at most of the den sites in NBFD. However, in STR, the presence of fruiting trees (in terms of basal area) did not differ significantly between used and available dens, which translated to the bear’s tendency to seek more important factors while denning, such as cover and seclusion, as mentioned above, rather than food availability. The presence of trees (including fruiting trees) can also provide necessary cover, and most of the dens (used and available) were located in well-wooded areas, which confirmed that sufficient cover mostly came from terrain complexities.

Caveats and limitations

However, the present study had several limitations. The temperature difference between outside and inside of the dens possibly plays a vital role in den selection by sloth bears (Yoganand, 2005; Akhtar et al., 2007) and other bear species as well (Wei et al., 2019). I could not thoroughly investigate the role of temperature due to logistic constraints. There was a possible distinct seasonal pattern of den use by sloth bears, as Yoganand (2005) mentioned in central India, which I could not discern in this present study. The lack of radio-collared individuals in STR also hindered us from precisely separating used dens from less used/unused ones. Thus I had to rely exclusively on ground-based knowledge. Despite these limitations, this study demonstrated the importance of ground-based knowledge in monitoring the den sites of sloth bears.

4.10 Conclusion

This present study has conservation implications. Since little is known about the denning ecology of sloth bears, my preliminary findings would help wildlife managers identify the potential dens for day resting and parturition. Day resting in cooler habitats is necessary for sloth bears, which is associated with a reduction in the rate of metabolism and body temperature in mammals, especially during seasons of high thermal stress (Siegel, 2008). Though poorly understood, the selection of den sites amidst extensive anthropogenic pressure is likely to cause stress on the sloth bear's physiology and can affect population growth due to cub abandonment, as observed in other bears (Linnell et al., 2000; White et al., 2001). Hence, it is imperative to identify preferred sloth bear dens (day resting and natal den) and subsequently provide necessary protection from human disturbance by imposing restrictions on human movement at potential denning habitats and raising awareness amongst local people explaining the importance of conserving such habitats. Preferred dens located near villages should be targeted first to undertake protection measures, and human activities (stone querying, livestock grazing) should be restricted. I believe that the management of denning habitats is crucial for the long-term survival of sloth bears, which would also benefit other wild animals in this landscape.

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CHAPTER 5: FOOD HABITS OF SLOTH BEAR

5.1 Introduction

The knowledge of the food habits of an organism provides insights into its ecology and management (Putman, 1984), especially in ever-expanding human-dominated landscapes from the global perspective (Ellis et al., 2010) and if the species is of conservation importance (Miquelle et al., 1996; Rodríguez et al., 2007; Lyngdoh et al., 2014; Hertel et al., 2016; Kirby et al., 2016; Tosoni et al., 2018). Amongst carnivores, bears are highly adaptable, individualistic and opportunistic omnivores that can rely on natural and human-induced food items and frequently come into conflict with humans (Robbins et al., 2004; Elfström et al., 2014; Herrero, 2018). Bears are reported to show a high level of behavioural plasticity in human-dominated areas (Martin et al., 2010; Ordiz et al., 2016; Zeller et al., 2019) and varied diet selection with respect to seasons and availability (Bojarska and Selva, 2012; Merkle et al., 2013; Hertel et al., 2016; Stenset et al., 2016; Palei et al., 2020; Rather et al., 2020). Like other organisms, the availability of feeding resources is a well-proven predictor for different ecological attributes of bears, i.e. space use, population density, reproductive success, and denning chronology (Hilderbrand et al., 1999; Costello et al., 2003; Nielsen et al., 2010; Krofel et al., 2017; McClelland et al., 2020). However, conflicting situations arise when bears are drawn toward the human-settlements due to the availability of human-induced foods (Charoo et al., 2011; Merkle et al., 2013; Kirby et al., 2016; Morales-González et al., 2020; Prajapati et al., 2021) or when the natural resources are shared between bears and humans (anthropogenic expansion into bear habitat) in fragmented and degraded landscapes (Treves and Karanth, 2003; Can et al., 2014). In this context, knowledge of the food habits of bears in areas having substantial anthropogenic pressure, thus, may guide the managers to allocate conservation efforts.

Sloth bears rely primarily on termites and ants as omnivores but heavily utilise seasonal fruits (Baskaran et al., 1997; Joshi et al., 1997; Bargali et al., 2004; Yoganand, 2005; Palei et al., 2020; Rather et al., 2020). In human-dominated landscapes, sloth bears often raid crops and opportunistically forage on certain fruits and human food wastes (Bargali et al., 2004; Palei et al., 2020; Prajapati et al., 2021), which increased the sloth bear-human interaction and perhaps acted as a precursor of conflict situations (Akhtar et al., 2004; Bargali et al., 2005; Prajapati et al., 2021). Throughout its distributional range, sloth bears were reported to avoid human disturbances (Puri et al., 2015); however, in some parts, they are forced to live in sub-optimal habitats and share resources with humans (Akhtar et al., 2004). Perhaps the flexible foraging behaviour and its ability to switch over different food items (either natural or cultivated) make sloth bears survive in human-dominated landscapes (Palei et al., 2020), but with the cost of human-related threats, as well.

With this view, I investigated the food habits of sloth bears in the Sanjay Tiger Reserve. My primary objective was to study the seasonal variation of food habits of sloth bears, with respect to the availability of food items, in the context of a human-dominated landscape.

5.2 Materials and methods

Scat collection

During 2017 and 2018, I systematically surveyed forest roads, trails, and dry streambeds of each forest Beat [5 km segments (n=3); total effort=15km/Beat], to search for scats of sloth bears while conducting carnivore sign surveys for tigers and co-predators (Jhala et al., 2017) in the core zone of STR. Scats were collected in winter (December 2016 to February 2017 and November 2017 to February 2018) and in summer (March 2017 to June 2017 and March 2018 to June 2018); however, during monsoon (July-October), I could not collect enough scat samples to draw a conclusive inference, due to inaccessibility issues. Attempts were made to collect relatively fresh scat samples (<7 days) by visually inspecting samples to avoid any season-specific ambiguities. Scats were identified by their size, shape, visible food remnants (termites, ants, seeds), and ancillary signs (dig mark, footprint, and claw mark) of sloth bears associated with scats. Locations of scats were also recorded by a handheld unit of GPS (Global Positioning System; Garmin e-Trex), along with information regarding forest type, elevation and freshness of the sample. All samples were collected in labelled zip-lock bags, sun-dried and sent to the laboratory for analysis.

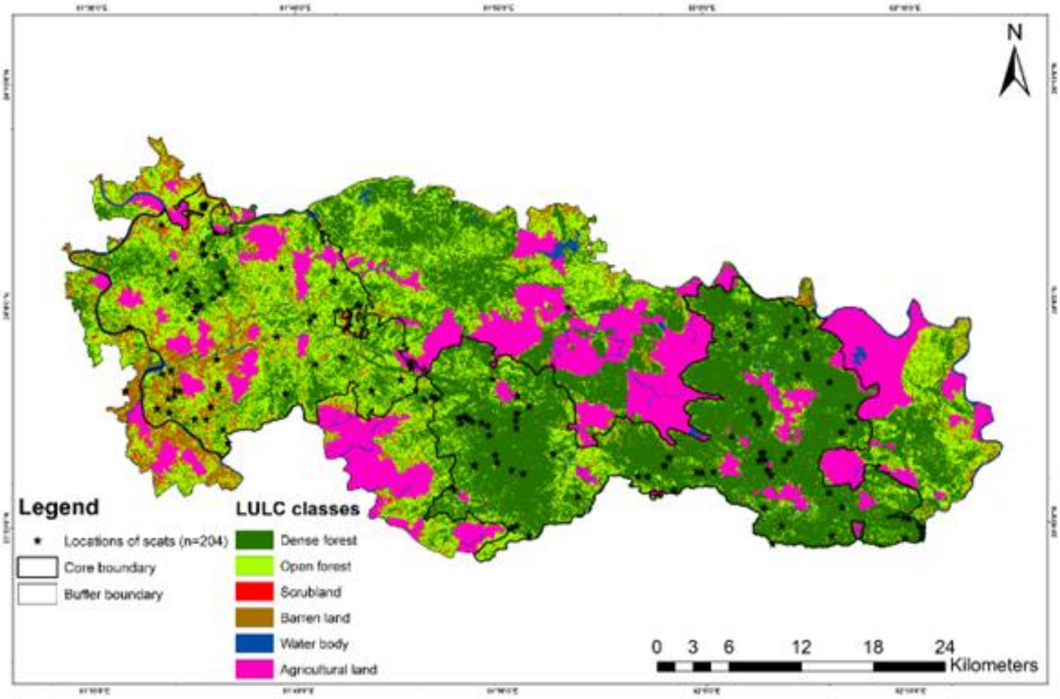

Figure 5.1. Locations of scats (n=204) that were collected in winter and summer during 2017-2018 in the core zone of Sanjay Tiger Reserve, Madhya Pradesh, India.

Firstly, scat samples were soaked overnight and rinsed in tap water using two sieves of 0.4mm and 0.7mm, to remove dirt and other fine particles other than food items (Bargali et al., 2004; Rather et al., 2020). After that, I oven-dried (40°C) the washed samples and subsequently manually separated each food item with the help of forceps, needles and a handheld magnifying glass. Food species were identified by visually observing conspicuous undigested parts such as seeds and pericarp of fruits for plant matter and chitinous exoskeleton of insects for the animal matter. The attempt was made to identify each food species at least up to the genus level (for plants), and insects were identified and grouped in broad categories (such as ants, termites, beetles etc.). All unidentifiable matters were grouped separately. In the case of hairs found in the scat [as sloth bears are known to scavenge occasionally (Bargali et al., 2004; Ramesh et al., 2009)], I identified the species following standard procedure (Mukherjee et al., 1994). I used all the scats for the diet analysis.

Data analyses

I calculated the frequency of occurrence per scat (FO% per scat; the total number of occurrences of each food species in the total number of scats analysed/total number of scats analysed x100) and per feed occurrence (FO% per feed occurrence; the total number of occurrences of each food species in the total number of scats analysed/ total number of occurrences of all food species from all analysed scats x 100) for each food item to assess the relative importance of each food species (Klare et al., 2011; Liccioli et al., 2015). I also determined the minimum number of scats required to capture the entire dietary spectrum of sloth bears in winter and summer by constructing observation-area curves (Odum and Kuenzler, 1955; Mukherjee et al., 1994).

I compared the frequency of occurrence of each broad feed item category (Plant matter and animal matter) between winter and summer by conducting a $\chi 2$ test, and the significance level was set at 0.05. The Levins niche breadth index (B) Krebs, 1999; Levins, 1968) was calculated to measure the niche breadth of the diet profile of sloth bears in two seasons. The formula of Levins index was as follows: $B=1/(\Sigma P_j)^2$, where P_j is the proportion of occurrence of food species j. Further, Levins index, i.e. B, was standardised [($B_{standard}$ = (B-1) / (N-1); N=total number of feed items] to interpret the results on a scale of 0-1, where 0 indicates highly specialist (selective in diet preference) and 1 indicates highly generalist (non-selective in diet preference) following Colwell & Futuyma (2013). I also calculated the Horn-Morisita index (C_{hn}; corrected for proportions or transformed count data) of dissimilarity as a measure of dietary overlap of sloth bear between winter and summer (Morisita, 1959; Horn, 1966). The value of C_{hn} was interpreted based on Langton (1982), where $0 < C_{hm} \leq 29\%$ was low, $30\% \leq C_{hm} \leq 59\%$ was medium and $C_{hm} \geq 60\%$ was considered high overlap.

5.3 Results

Scat analysis

A total of 204 scats of sloth bear was collected during 2017-2018, out of which 87 and 117 scats were collected during winter and summer, respectively. A total of 18 food species (insects=6, plants=11 and hairs of livestock) were found, with a higher number of species (n=16) observed during summer in comparison to winter (n=11). I found that 58 and 75 scats were adequate to capture the dietary spectrum of sloth bears during winter and summer, respectively, as indicated by observation-area curves (Figure 5.2).

An average of 2.31±0.09 and 2.15±0.10 occurrences of species were recorded during winter and summer, respectively. A higher number (n=9) of plant species was observed during summer, in comparison to winter (n=5), whereas the number of insect feed items was broadly the same in the two seasons (winter=5, summer=6). During winter, amongst insects, both FO per scat and feed occurrence were highest for termites, followed by black ants, red ants, beetles and honey bees, whereas *Ziziphus mauritiana* dominated the diet of sloth bears amongst plants, followed by guava (*Psidium guajava*), *Z. oenoplia*, and species of fig (Table 5.1). Contrastingly, in summer, plant matters (i.e. *Diospyros melanoxylon*, *Cassia fistula*, *Syzygium cumini* and *Madhuca longifolia*) were most frequently found in food species, whereas insects (termites, black ants, honey bees, larvae, red ants and beetles) were also consumed often, but less than those of winter (Table 5.1). Five Species of insects (termites, black ants, red ants, beetles and honey bees), three species of plants (*D. melanoxylon*, *Z. mauritiana* and *P. guajava*) and hairs of livestock were common for both seasons. Between seasons, consumption of animal matter ($\chi2=8.45$, df=1, $p<0.05$) and plant matter ($\chi2=68.44$, df=1, $p<0.05$) were significantly different from each other (Table 5.2).

Table 5.1. Frequency of occurrence percentage per scat (FO/scat) and per feed occurrence (FO/feed occurrence) observed from scats of sloth bears for winter and summer during 2017-2018 in Sanjay Tiger Reserve, Madhya Pradesh, India. 'n' denotes the number of scats examined for diet analysis during winter and summer

Sl. No.	**Feed item**		**Winter (n=86)**		**Summer (n=117)**		
		Availability of food species●	**FO/scat (%)**	**FO/feed occurrence (%)**	**FO/scat (%)**	**FO/feed item (%)**	**Food species of sloth bears collected/used by local people**
	Insects						
1	Termite	Year-round	90.697	39.196	48.718	22.709	-
2	Black ant	Year-round	79.069	34.171	28.205	13.147	-
3	Red ant	Year-round	16.279	7.035	5.128	2.390	-
4	Beetle	Year-round	11.628	5.025	4.273	1.992	-
5	Honey bee	Year-round	1.163	0.502	6.838	3.187	Collection of honey
6	Larvae	Year-round	-	-	6.838	3.187	-
Plants (fruits/flowers)							
1	Tendu (*Diospyros melanoxylon*)	April-May	2.325	1.005	70.940	33.067	Fruits are collected for consumption purpose
2	Ber (*Ziziphus mauritiana*)	January-March	13.953	6.030	5.128	2.390	*Ripened fruits are collected for consumption purpose
3	Makoi (*Z. oenoplia*)	November-January	5.814	2.512	-	-	-
4	Guava (*Psidium guajava*)	Year-round	5.814	2.512	0.855	0.398	*Fruits are collected for consumption purpose

5	Jamun (*Syzygium cumini*)	June-July	-	-	5.983	2.789	ʿFruits are collected for consumption purpose
6	Amaltas (*Cassia fistula*)	March-June	-	-	13.675	6.374	-
7	Bel (*Aegle marmelos*)	May-June	-	-	1.709	0.797	*Fruits are collected for consumption purpose
8	Bhilwa (*Semecarpus anacardium*)	March-April	-	-	2.564	1.195	Fruits are collected for medicinal properties and consumption purpose
9	Mahua (*Madhuca longifolia*)	March-April (for flower) and June-July (for fruits)	-	-	5.983	2.788	Flowers and fruits are collected for various purposes, viz. making of country liquor, consumption and extraction of oils
10	Chiraunji (*Buchanania cochinchinensis*)	April-May	-	-	3.419	1.594	Fruits are collected mainly for consumption purpose
11	Species of Fig (*Ficus* sp.)	Year-round	2.325	1.005	-	-	-
Others							
1	Animal hair (livestock)	-	2.325	1.005	5.128	2.390	-
2	Unidentified	-	6.977	3.015	3.419	1.594	-

*plant species were rarely found inside the forest [evident from the surveyed vegetation plots (n=479)] and mainly concentrated in and around villages; ʿAvailable in both forests and around villages; ● Availability of ripened fruits was mentioned following Krishen, 2013

Table 5.2. Results of χ^2 tests to assess the differences between two major feed categories (Animal matter and plant matter) in two different seasons, i.e. winter and summer, identified from the scats of sloth bears collected during 2017-2018 in Sanjay Tiger Reserve, Madhya Pradesh, India.

Feed category	**Occurrence records**		**χ^2**	**df**	**p-value**
	Winter	**Summer**			
Animal matter	173	123	8.446	1	0.004 (<0.05)
Plant matter	26	129	29.369	1	2.2e-16 (<0.005)

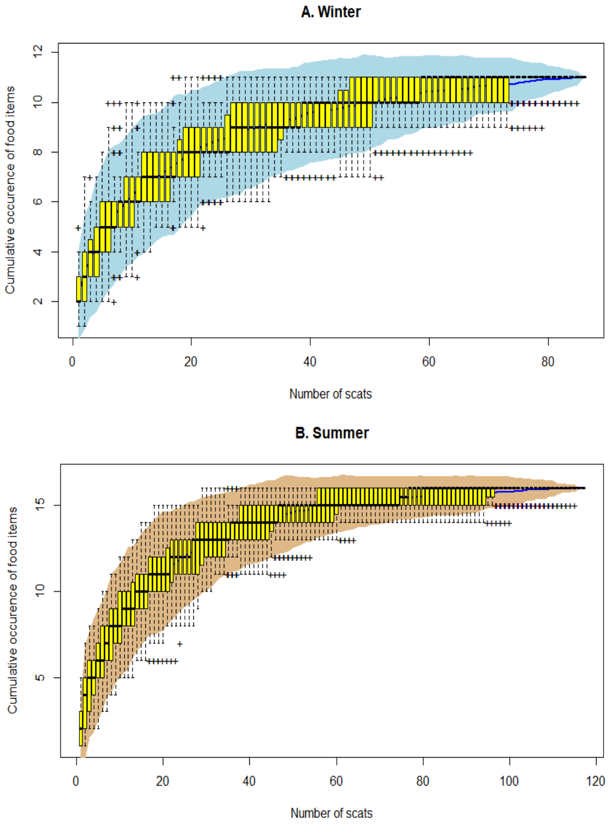

Figure 5.2. Observation-area curve was generated to quantify the number of scats required to accurately represent the dietary spectrum of sloth bears in winter (A) and summer (B) in Sanjay Tiger Reserve, Madhya Pradesh, India, from 2017 to 2018.

Levins niche breadth index indicated an overall narrow or specialised diet preference of sloth bears, irrespective of seasons; however, dietary niche breadth was slightly higher in summer ($B_{standard}$=0.29) than in winter ($B_{standard}$=0.25). Horn-Morisita index indicated a medium dietary overlap (i.e. 40%) of sloth bears between seasons.

5.4 Discussion

I reported the seasonal diet preference of sloth bears in STR. I found distinct seasonal food choice patterns shown by sloth bears, with a year-round high consumption of termites and ants. My findings were broadly in agreement with previous studies on the diet preference of sloth bears (Joshi et al., 1997; Bargali et al., 2004; Yoganand, 2005; Ramesh et al., 2009; Palei et al., 2020; Rather et al., 2020). However, the proportion of insects and fruits in sloth bears' diet varies with their geographic distributions. In Nepal's Royal Chitwan National Park (RCNP), termites and ants dominated the diet of sloth bears irrespective of habitat and season, although sloth bears did prefer plant matters significantly during summer (Joshi et al., 1997). Despite a steady proportion of insect diet, sloth bears showed more preference for fruits in the Mudumalai Tiger Reserve of southern India (Baskaran et al., 1997; but see Ramesh et al., 2009). Also, the utilisation of fruits by sloth bears may differ with their availability. In STR, I found that sloth bears heavily feed on fruits during summer; however, only two species (*D. melanoxylon* and *C. fistula*) significantly contributed (in terms of FO) to the diet of sloth bears. I believe it was related to the high abundance of fruiting trees in STR (233/hectare) compared to other study sites (Baskaran et al., 1997; Joshi et al., 1997; Akhtar et al., 2004; Rather et al., 2020). Only a few species, especially *D.*

melanoxylon, formed bulk (~58%) of the total density of fruiting trees in STR and frequently occurred (~71%) in scats of sloth bears during summer, but also indicated a relatively low abundance and diversity of other potential fruiting trees. A similar observation was made in another human-dominated landscape of central India (Rather et al., 2020). The other two preferred species, i.e. *C. fistula* and *S. cumini*, were less abundant in STR; however, the latter was more restricted to the riparian habitats, which I could not survey adequately. Similar to *S. cumini*, another fruiting tree, i.e. *Z. mauritiana*, was either more common in villages or less abundant and highly patchy in distribution, but an important food plant species for sloth bears during late winter and the onset of summer, as indicated by my findings and others' (Bargali et al., 2004; Yoganand, 2005; Palei et al., 2020). In STR, sloth bears are known to feed on fruits (*Z. mauritiana, S. cumini, P. guajava*) from villages that are less abundant in the forest, as also observed by Akhtar et al. (2004) in North Bilaspur Forest Division (NBFD), central India. In Panna National Park of central India, the abundance of fruiting trees was also reflected in the annual food intake by sloth bears (Yoganand, 2005). Besides *D. melanoxylon*, two other potential food plant species (*B. cochinchinensis* and *M. longifolia*) also occurred in high densities but were less preferred by sloth bears in STR. It seems that these species' fruits (and flowers of *M. longifolia*) were either less realised to bears or highly utilised within a relatively short period, which was not adequately captured through my sampling. The digestibility of flowers of *M. longifolia* could also be a reason for not being frequented in the scats of sloth bears during summer. Furthermore, both of these food species (fruits and flowers) are harvested by local people extensively, especially the flowers of *M. longifolia* (Siddiqui et al., 2014; Islam and Quli, 2017; Kumar et al., 2018; Rajput et al., 2018; Mishra and Poonia, 2019; Mishra et al., 2021), which may have limited its availability for sloth bears. Fruits of *D. melanoxylon*, too, are collected by the local people (Mishra et al., 2021), but perhaps its high abundance facilitated the shared use of fruits between humans and sloth bears in STR. In NBFD, a significant influence of the year-round availability of fruits of *Ficus* sp. (species of figs) in the diet of sloth bears was documented (Bargali et al., 2004). Fruits of *Ficus* sp. are not as demanding to local people as other fruits mentioned above, which may reduce the overlapped resource use between sloth bears and humans. However, I could not find enough evidence regarding the consumption of fruits of *Ficus* sp. by sloth bears in STR, perhaps due to the rarity of the species. Nevertheless, the seasonal contribution of fruits (despite their less variation) to the diet of sloth bears was distinctive in STR.

As a myrmecophagous ursid, sloth bears preferred termites and ants throughout the year, with a relatively higher reliance in winter, as indicated by the findings of my study. Feeding on termites and ants can be energetically efficient, possibly due to their higher nutritional values than other insects (Redford and Dorea, 1984). The ubiquity of termites and ants, and the flexible nature of diet preference of myrmecophagous mammals, may result in their broad and diverse distribution (Redford, 1987), which may also be the case for sloth bears in the Indian subcontinent (Johnsingh, 2003; Dharaiya et al., 2016; Steinmetz et al., 2020). Other bears are reported to feed on colonial insects frequently as well when they are available (Noyce et al., 1997; Swenson et al., 1999; Fredriksson et al., 2006; Stenset et al., 2016; Tosoni et al., 2018). Also, bears often shift to a myrmecophagous diet when plant matters no longer meet the necessary nutritional requirements (Noyce et al., 1997). However, sloth bears are uniquely adapted to live in environments where the abundance of fruiting trees is relatively low but rich in termites and ants, compared to the other two bear species living in southeast Asia (Steinmetz et al., 2020). It appeared that the abundance of termite mounds was plentiful in STR, as compared to RCNP and Bandhavgarh Tiger Reserve (central India) (Garshelis et al., 1999; Rather et al., 2020), but comparable with NBFD (Akhtar et al., 2004). My findings corroborated the relatively high dependence of sloth bears on termites and ants, as indicated by other studies in human-dominated landscapes of central India (Bargali et al., 2004; Rather et al., 2020) and elsewhere (Joshi et al., 1997).

The dietary niche breadth of sloth bears was narrow for both seasons, indicating a specialised diet pattern. The niche breadth was slightly higher in summer than in winter due to the relatively high diversity of food species, especially plant matter (fruits). A similar observation was also made by Palei et al. (2020) in a human-dominated agroforestry landscape. Medium dietary overlap (40%) between seasons could be attributed to the shared consumption of insects (termites and ants) and a few food plants, especially fruits of *Z. mauritiana*, which were consumed both during winter and the onset of summer (Month of March). Depending on the number of shared food species, high seasonal (winter and summer) dietary overlap of sloth bears was evident elsewhere (Bargali et al., 2004; Palei et al., 2020). However, fewer available food species (especially food plants) may have resulted in lower seasonal dietary overlap and niche breadth in STR. I believe that higher consumption of abundant food plant species (e.g. *D. melanoxylon*) and a greater reliance on termites and ants may have facilitated sloth bears' survival in a human-dominated landscape of STR.

Caveats and limitations

I acknowledge the limitations of the present study. Despite being an important food species for sloth bears, I could not assess the abundance of ants and termites (other than mound-living) at vegetation plots, which may have limited the understanding of the myrmecophagy of sloth bears. Similarly, the quantification of fruits was not carried out, which could have shed light on the relationship between the availability versus relative consumption of seasonal fruits by sloth bears. I also could not assess the diet of sloth bears during monsoon due to accessibility issues and logistic constraints.

5.5 Conclusion

However, despite the limitations mentioned above, my study demonstrated the survival of sloth bears in a human-dominated landscape regarding diet preference. It was also observed that some of the important food plant species of sloth bears (e.g. *Z. mauritiana*) are only abundant in villages, which brings sloth bears to proximity to the human settlements and increases the chance of conflict. Plantation and management of such food plants in the degraded lands and other selected areas could reduce the human-sloth bear interface. Also, as mentioned before, species of *Ficus* can be crucial for the survival of sloth bears in human-dominated landscapes; hence, management of such trees should be prioritised. However, voluntary relocation of villages from the core zone of STR and subsequent habitat management actions at village relocation sites would improve the habitat quality substantially. Meanwhile, monitoring the trends in diet preference of sloth bears should also be carried out through sound methods, which is expected to guide the managers to devise proactive plans to conserve sloth bears in human-dominated landscapes.

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CHAPTER 6: HUMAN-SLOTH BEAR INTERACTION AND CONFLICT PREDICTION

6.1 Introduction

Ever-expanding human population and resultant degradation and fragmentation of habitat inadvertently affected wildlife and increased the proximity-driven interaction between humans and wildlife (Messmer, 2000, 2009; Inskip and Zimmermann, 2009; Elfström et al., 2014; Joshi et al., 2016). Often these human-wildlife interactions posed challenges to conservation initiatives due to the significant damages in the form of livestock depredation, human-mauling, crop loss and retaliatory killing of the concerned species (Rajpurohit and Krausman, 2000; Madhusudan, 2003; Treves and Karanth, 2003; Bargali et al., 2005; Karanth et al., 2012). However, identifying the underlying reason behind such conflict scenarios is necessary to address human-wildlife conflict (Sutherland et al., 2009). Amongst other species of wildlife, conflict with large carnivores (including bears) has a significant impact on human health and livelihood worldwide (Treves and Karanth, 2003; Chapron et al., 2014). Human-carnivore interaction has been studied widely to understand the conflict's nature and extent and draw inferences on possible conflict mitigation (Redpath et al., 2013; Nyhus, 2016).

Expansion of anthropogenic activities has affected the distribution of carnivores and their global temporal and spatial use of the habitat (Dirzo et al., 2014; Gaynor et al., 2018; Tucker et al., 2018). Also, humans directly or indirectly impact carnivores' mortality and physiological fitness (Rode et al., 2007; Darimont et al., 2015).

However, it was observed that human-wildlife interaction could be complex and dependent on several factors (Tablado and Jenni, 2017). Wild animals, especially carnivores, can respond to increased or persistent anthropogenic activities in a variety of ways (Ordiz et al., 2011; Carter et al., 2012; Elfström et al., 2014; Tablado and Jenni, 2017; Suraci et al., 2019b), including strong fear responses (Suraci et al., 2019a). The response of carnivores towards human occurrence may be governed by the spatial extent and intensity of the anthropogenic pressure prevailing in the landscape. A human-dominated landscape, especially with a year-round high occurrence of humans, can affect the space use pattern of carnivores, leading to avoidance of areas where human disturbance is high (Chi and Gilbert, 1999; Fortin et al., 2016). Many wildlife species, including bears, consider humans a perceived threat or predator as they elicit similar responses in the presence of natural predators (Joshi et al., 1999; Frid and Dill, 2002; Ciuti et al., 2012). However, bears were also frequently attracted to human settlements for opportunistic foraging, especially crop-raiding, livestock depredation and feeding on garbage (food wastes) (Charoo et al., 2011; Morales-González et al., 2020; Prajapati et al., 2021). Despite these exceptions, animals would likely avoid places where the pressure of humans persists (Dröge et al., 2017). Similarly, if the temporal pattern of human occurrence is more predictable to animals, it is possible that animals would segregate their movement by increasing the time of activity when humans are least active (Suraci et al., 2019b) instead of avoiding spatially (Nickel et al., 2020). Such behaviour would, in turn, facilitate coexistence (Fagen and Fagen, 1994; Fortin et al., 2016; Srivastava et al., 2020).

It is expected that human-wildlife conflict is a direct outcome of a high degree of the human-wildlife interface (Pack et al., 2013). However, unlike Europe and North American countries, in the south and south-east Asian regions, the prevalence of human-wildlife conflict is primarily due to rich and endangered biodiversity and associated developmental activities in and around forest areas (Madhusudan and Karanth, 2002; Sodhi et al., 2010). The high dependence on natural resources driven by poverty and expanding human population impacted biodiversity (Sodhi et al., 2010; Chao, 2012) adversely and subsequently increased conflict scenarios due to highly overlapped resource use (Treves et al., 2006). In this view, there is a paucity of studies considering spatio-temporal interaction between humans and carnivores with global distribution, such as Asian bear species, to address conflict issues more effectively (Can et al., 2014; Anand and Radhakrishna, 2017).

As mentioned in previous chapters, the sloth bear is notably involved in conflict with humans in some parts of its distributional range, especially in human-dominated landscapes (Rajpurohit and Krausman, 2000; Bargali et al., 2005; Akhtar and Chauhan, 2008; Garcia et al., 2016; Debata et al., 2017; Dhamorikar et al., 2017; Singh et al., 2018; Sharp et al., 2020; Prajapati et al., 2021). However, except for a few anecdotal pieces of evidence, attacks on humans by sloth bears are primarily defensive rather than intentional or predatory, as they perceive humans as threats or predators (Sharp et al., 2020). In human-dominated forests of central India, it was found that most of the victims belonged to tribal communities with high natural resource dependency and were attacked inside the forest while collecting non-timber forest produce (NTFP) (Dhamorikar et al., 2017), as well as adjacent to the village peripheries and agricultural lands (Dhamorikar et al., 2017; Singh et al., 2018). Seasonal and temporal patterns of these attacks varied between different landscapes (Bargali et al., 2005; Dhamorikar et al., 2017; Singh et al., 2018). Sloth bears are mainly nocturnal and crepuscular in activity (Yoganand, 2005; Bargali et al., 2012; Ramesh et al., 2013) but could be active throughout the daylight hours in some parts of their distribution (Joshi et al., 1999). It appears that most conflict cases happened due to the overlapped activity between sloth bears and humans during the crepuscular period (Akhtar et al., 2004; Bargali et al., 2005). However, there is a shortfall of studies which describe the fine-scale interaction and overlap (spatial and temporal) between sloth bears and humans following a statistically robust framework in a human-dominated landscape. Also, to my knowledge, no research work has been conducted to predict probable conflict sources and hotspots based on occurrence records of sloth bears and humans.

In this view, I utilized the season-specific detection records of sloth bears and humans obtained from camera trap surveys conducted in Sanjay Tiger Reserve (STR), central India, to understand the interaction between sloth bears and humans. Firstly, based on detection non-detection data of sloth bears and humans, two-species occupancy models (Richmond et al., 2010) were developed to assess the species interaction in terms of association or segregation. Furthermore, I estimated the extent of spatial and temporal overlap using the relative abundance index (RAI) (O’Brien et al., 2003; Rovero and Marshall, 2009) of sloth bears and humans. I expected that sloth bears would avoid humans spatially regardless of seasons, as evident from other studies (Paudel et al., 2022; Pokharel et al., 2022). However, I also assumed that sloth bears would be more nocturnal in disturbed habitats to avoid human encounters during daylight hours. Finally, I used geographic profiling/geo-profiling (GP) to identify the conflict source and dispersion (i.e. hotspots), a novel approach to tackle wildlife crime and predict conflict hotspots (Faulkner et al., 2018; Struebig et al., 2018), which was earlier only restricted to criminology (Rossmo, 2000).

With this background, I had the following objectives:

1. To assess the seasonal spatial interaction and spatiotemporal overlap between sloth bears and humans in the context of a human-dominated landscape, and
2. to identify the probable conflict source sites and hotspots that can serve as a basis for dealing with sloth bear-human conflict mitigation strategies.

6.2 Materials and methods

Camera trapping

The details of the camera trapping method were mentioned in Chapter 3, under the “Materials and methods” section. The camera trap data was clubbed of two summer and two winter seasons to represent the entire core area of STR for each season (overall winter and summer).

Spatial interaction

In the first step, I used two-species conditional occupancy models (Richmond et al., 2010) to assess if sloth bear shows spatial avoidance towards human occurrence in two seasons (winter and summer). The two-species occupancy model estimates the conditional probability of occupancy and detectability of a less dominant species in the presence of a dominant species (Richmond et al., 2010). I interpreted the "probability of occupancy" of a species as the probability of "use" since I believed that size of the site would potentially violate the closure assumption and independence between sites (MacKenzie et al., 2017, p447). If species A and species B are to co-occur and A is dominant over B, the following parameters are estimated in the two-species model:

i. Ψ^{A} (Probability of occupancy of A)
ii. Ψ^{Ba} (Probability of occupancy of B when A is absent)
iii. Ψ^{BA} (Probability of occupancy of B when A is present)
iv. p^{A} (Detection probability of A when B is absent)
v. p^{B} (Detection probability of B when A is absent)
vi. r^{A} (Detection probability of A when both A and B were present)
vii. r^{BA} (Detection probability of B when A was present and detected)
viii. r^{Ba} (Detection probability of B when A is present but not detected)

In this present study, I assumed that human (species A) activity does not depend on the occurrence of sloth bears (species B). Sloth bears probably perceive humans as a threat equivalent to predators (tigers and leopards), with which they are sympatric (Laurie and Seidensticker, 1977; Yoganand, 2005). Given the extensive anthropogenic pressure in STR, it is very likely that sloth bears would adjust their space use in accordance with human occurrence. In this view, I assumed the sloth bear to be a subordinate species in the two-species occupancy model. Although considering humans as the dominant species in two-species occupancy models was not common [see Parsons et al., (2016)]; however, I intended to assess the degree of spatial avoidance of sloth bears in response to the extensive occurrence of a perceived threat (i.e. humans) in a human-dominated landscape.

The species interaction factor (SIF) is a derived parameter estimated from the two-species occupancy model. A value of SIF >1 shows the degree of species aggregation rather than avoidance, and a SIF value<1 indicates spatial avoidance or segregation, where two species are less likely to occur with each other than expected, under the hypothesis of species independence. A SIF value ~1 indicates that both species can occur independently. Following Richmond et al. (2010), SIF can be calculated as

$$\boldsymbol{SIF} = \frac{\boldsymbol{\Psi^{A}\ \Psi^{BA}}}{\boldsymbol{\Psi^{A}[\Psi^{A}\Psi^{BA} + (1 - \Psi^{A})\Psi^{Ba}]}}$$

I calculated SIF and compared it between seasons to assess the avoidance of space used by sloth bears in the presence of humans. I also carefully inspected the values of other parameter estimates (Ψ^{Ba}, Ψ^{BA}, p^{B}, r^{BA} and r^{Ba}) and compared them between seasons. Model comparison and selection were based on Akaike's Information Criterion (AIC) (Burnham and Anderson, 2002). I used the program Presence (Version 2.13.11) (Hines, 2006) to estimate two species' occupancy and related parameters.

Spatial and temporal overlap

Spatial overlap

In the second step, the extent of spatial overlap between sloth bear and human was estimated for summer and winter as I was interested in assessing the interaction on a finer scale, which might not always be discernible from occupancy models. In order to do so, the relative abundance index (RAI; O'Brien et al., 2003; Rovero & Marshall,

2009) was calculated at each camera trap location for both species using the following formula:

RAI = (No. of independent photographs/total no. of days during which each camera trap was operating) x100

Considering each camera trap location as spatially independent from other locations, I calculated the spatial overlap by applying Pianka's overlap index (Pianka, 1974). This method takes account of the proportion of RAI values of a species' pair with respect to different camera trap locations. The output value (estimate of overlap) ranges between 0, i.e. no overlap, to 1 or complete overlap. Although it was developed initially to estimate niche overlap, several studies applied Pianka's index to estimate spatial overlap between co-occurring species and human activities (Ramesh et al., 2012b; Yang et al., 2019; Mori et al., 2020; Zhao et al., 2020). I also estimated 95% confidence interval (CI) of the spatial overlap estimate by 10,000 bootstrapped replicates. All analyses related to spatial overlap estimation were carried out in R (R Core Team, 2019) by using the package "spaa" (Zhang et al., 2013).

Temporal overlap

To assess the activity pattern of sloth bears and humans for summer and winter, I used the same camera trap detections that were used to compute spatial overlap. Following the methodology described by Ridout & Linkie (2009), a non-parametric kernel density function was fitted to the circular time data obtained from the camera trap photographs of both species, which would produce the pairwise (sloth bear-human) activity patterns. Then, the coefficient of overlap (delta, Δ) was estimated from the area under the curve of two different density functions (sloth bear and human). Values of Δ ranged from 0 to 1, where 0 denotes no overlap and 1 denotes complete overlap (Linkie & Ridout, 2011; Meredith & Ridout, 2014). I calculated Δ_4 when the number of independent events of both sloth bear and human was>75, except for one season (winter 2018), when the number of sloth bear photographs was <75, and hence, Δ_1 was calculated, as mentioned by Linkie & Ridout (2011). The 95% CI was generated by 10,000 bootstrapped samples of overlap coefficient to check the precision of the estimates (Linkie and Ridout, 2011). Analyses regarding the estimation of overlap coefficient were carried out in package "overlap" (Meredith and Ridout, 2021), implemented in R (R Core Team, 2019).

One-sample Watson's U^2 test (Watson, 1961) is a Goodness of Fit test, which was performed to determine whether the distribution of each species' detection time significantly deviates from the uniform distribution of activity. Similarly, I performed a two-sample Watson's U^2 test (Watson, 1962) to assess if the activity distributions of sloth bears and humans were significantly different to each other between summer and winter. I also tested for the difference in activity of sloth bears and humans separately between seasons through a two-sample Watson's U^2 test. Both one-sample and two-sample Watson's U^2 tests were performed in the software "Oriana" (Kovach, 2011).

Geographic profiling for sloth bear-human conflict

Sloth bear-human conflict incidents

Records of conflict incidents between sloth bears and humans in the form of human injury and death were obtained from the forest department of STR for ten years (2009-2019). I retained the records only for the core area of STR, as the camera trapping exercise was conducted only in the core area. However, the GPS locations of these incidents were generally not recorded by the forest department; instead, the name of the concerned forest beat and compartment were mentioned. As compartments are the smallest (average size 2.12 km^2) administrative units of the study area (and all other Tiger Reserves), I generated centroids of compartments of forest Beats where conflict incidents occurred and used this information for further analysis.

Geographic profiling

Geographic profiling (GP) is a technique used to estimate the probable sources of the spatial encounters (human-sloth bear encounters in this case) when the sources are unknown (villages or sloth bear territories). It works on the Bayesian framework based on a distance decay process (Verity et al., 2014). I used the Dirichlet Process Mixture (DPM) (Verity et al., 2014; Faulkner et al., 2017) model to produce a geographic profile of sloth bear-human encounter using the package Rgeoprofile 2.1.0 in R (https://github.com/bobverity/Rgeoprofile). GP has been proven advantageous over conventionally used prediction models such as kernel density and ensemble modelling for mapping invasive species as well as conflict or risk assessment (Stevenson et al., 2012; Struebig et al., 2018). The process consists of two parts: the incident locations where the conflict occurred and the source locations from where the conflict might have arisen. The model divides the incident locations into ‘n’ clusters. Then the source sites are predicted using the Gibbs sampler (Geman and Geman, 1984) within a Markov Chain Monte Carlo framework until it converges on the posterior distribution of sigma (σ). Sigma (σ) is the standard deviation of the dispersal distribution, and it means how far I expect the sources from incident sites. There are two approaches to test the model efficiency: one is by using the hit score percentage (HS % ranging from

0% to 100%) and the second by using the Gini coefficient (G, ranging from 0 to 1) (Gini, 1921; Faulkner et al., 2018). The GP model provides an optimal search process for the source locations or anchor points. The hit score percentage (proportion of the area covering the crime divided by the total area) was used to test the model's efficiency with a lower hit score value depicting more accuracy in the search process. The Gini coefficient was used to assess the model performance; the higher the Gini coefficient, the more accurate the geographic profile (Faulkner et al., 2018).

I produced a geographic profile of sloth bear-human encounter using a) conflict data of sloth bear attacks on humans and b) camera trap data where both the sloth bear and human were detected together (but at different times). I used the compartment centroids for the conflict data, and camera trap data consisting of only sloth bears or humans were excluded. Duplicate points were removed from the data as the DPM model tends to fit very low sigma values with repeated locations (Struebig et al., 2018). The mean sigma (σ) used for the analysis was 2 km [assuming 87% of the dispersal events occurring within 2 km of the source; following Faulkner et al. (2017)] and ran the full data set with locations of conflict and camera trap, using ten chains with a burn-in of 10,000. Firstly, the resultant model will identify the probable conflict source sites, followed by conflict hotspots or dispersion around the source sites. Next, I compared any seasonal shift in the conflict probable areas in STR using the summer and winter camera trap locations separately where sloth bears and humans were both detected. In this study, I assumed the camera trap data where both the sloth bear and human were detected as probable conflict sites. So, model validation was conducted to test the efficacy of camera trap data (n=88) alone to predict future probable conflict areas. I generated a geographic profile based solely on camera trap data (where both sloth bear and human were detected). Sloth bear attack sites (n=57) were then overlaid on the geographic profile layer to check whether the conflict sites were falling over the predicted probable conflict areas generated by the camera trap data.

6.3 Results

A total of 179 and 311 independent detections of sloth bears were obtained from camera traps during winter and summer, respectively, during the entire survey period of 12,430 trap-nights (winter=6,435 trap-nights; summer=5,995 trap-nights). Detection of sloth bears was substantially higher in SNP (winter=122; summer=235) than DWLS (winter=57; summer=76), irrespective of seasons (Figure 6.1). Similarly, detection of humans was found to be higher in SNP (winter=3396; summer=2565) than DWLS (summer=1749; winter=1862) (Figure 6.1).

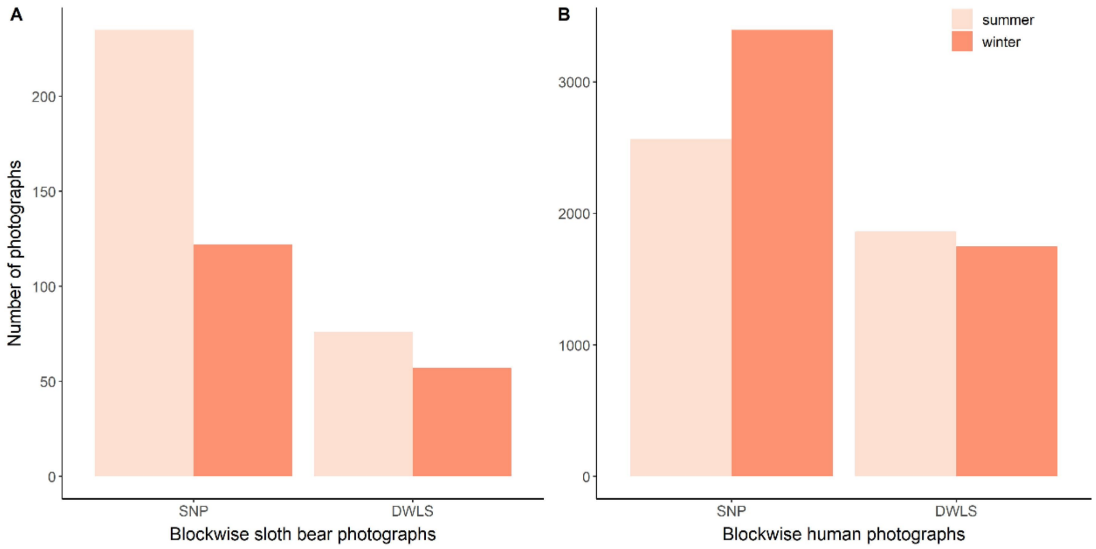

Figure 6.1. Block-wise photographic captures of sloth bears (**A**) and humans (**B**) for winter and summer were obtained from camera trap surveys from 2017 to 2018 in the core area of Sanjay Tiger Reserve, Madhya Pradesh, India.

Spatial interaction

During winter and summer, SIF values (SIF value ± Standard Error) were found to be 0.98±0.007 (95% CI: 0.97-0.99) and 1.00 ± 0.03 (95% CI: 0.93-1.07), respectively (Table 6.1). So, no difference in species interaction was observed between seasons by and large. The closeness of the SIF value towards 1 indicated no distinguishable pattern of spatial avoidance shown by sloth bear towards human occurrence. The probability of use of sites by sloth bears in the presence of humans was found to be less [Ψ^{BA} (SE) = 0.71 (0.04)] than when humans are absent [Ψ^{Ba} (SE) = 1.00 (0.00)] during winter. However, during summer, the probability of use of sites by sloth bears did not differ much [Ψ^{BA} (SE) = 0.79 (0.04); Ψ^{Ba} (SE) = 0.77 (0.17)] in the presence or absence of humans.

Table 6.1. Values of species interaction factor (SIF), probability of use ($\boldsymbol{\Psi^{BA}}$) of sites by sloth bears (denoted as "B") in the presence of humans (denoted as "A"), probability of use ($\boldsymbol{\Psi^{Ba}}$) of sites by sloth bears in the absence of humans (denoted as "a") and detection SIF (delta, **δ**), along with associated standard error (SE), calculated for two seasons (winter and summer) in Sanjay Tiger Reserve, Madhya Pradesh, India, during 2017-2018.

Season	**SIF ± SE**	$\boldsymbol{\Psi^{BA}}$ **± SE**	$\boldsymbol{\Psi^{Ba}}$ **± SE**	**δ ± SE**
Overall winter (2017 & 2018)	0.98 ± 0.007	0.71 ± 0.04	1.00 ± 0.00	1.05 ±0.06
Overall summer (2017 & 2018)	1.00 ± 0.03	0.79 ± 0.04	0.77 ± 0.17	1.06 ± 0.07

Spatial and temporal overlap

Spatial overlap

Higher spatial overlap (0.44) between sloth bears and humans was observed during winter than in summer (0.31). A high variation in sloth bear and human spatial overlap was observed between seasons in DWLS compared to SNP (Table 6.2).

Table 6.2. Spatial overlap index values of species pair (sloth bear and human), with associated 95% confidence interval (CI), during 2017-2018 in Sanjay Tiger Reserve, Madhya Pradesh, India.

Season	**Area**	**Spatial overlap between sloth bear and human with 95% CI**
Winter 2017	SNP	0.41 (0.21-0.68)
Summer 2017	DWLS	0.17 (0.10-0.33)
Winter 2018	DWLS	0.51 (0.24-0.78)
Summer 2018	SNP	0.38 (0.18-0.62)
Overall winter (2017 & 2018)	SNP+DWLS	0.44 (0.27-0.65)
Overall summer (2017 & 2018)	SNP+DWLS	0.31 (0.16-0.49)

Temporal overlap

It was found that, during summer, the temporal overlap between sloth bears and humans was higher than in winter (Table 6.3, Figure 6.2). Sloth bears were found to be more crepuscular and nocturnal (Watson’s U^2=0.09; $p<0.05$ for winter; Watson’s U^2=0.51; $p<0.005$ for summer) irrespective of the seasons, whereas humans’ activity were found to be primarily diurnal (Watson’s U^2=2.25, $p<0.005$ for winter; Watson’s U^2 =3.85; $p<0.005$ for summer; Table 6.4). During summer, human activity started before 06:00 and extended as long as 22:00 (Figure 6.2). However, during winter, human activities mostly started after 07:00 and continued till 20:00 (Figure 6.2). Watson's two-sample U^2 test revealed that sloth bear and human activity patterns significantly differed (Watson's $U^2 = 9.84$, $p<0.001$ for winter; Watson's $U^2 = 11.13$, $p<0.001$ during summer; Table 4) from each other, for both the seasons. However, between seasons, the activity of sloth bears and humans differed significantly ($U^2 = 0.22$, $p<0.05$ for sloth bears; $U^2 = 19.61$, $p<0.001$ for humans).

Table 6.3. Estimates of temporal overlap (Δ_4) between sloth bear and human, with an associated 95% confidence interval (CI), during 2017-2018 in Sanjay Tiger Reserve, Madhya Pradesh, India. *due to fewer detections (<75) of sloth bears, Δ_1 was calculated instead of Δ_4.

Season	Area	Temporal overlap (Δ_4) between sloth bear and human with 95% CI
Winter 2017	SNP	0.343 (0.282-0.404)
Summer 2017	DWLS	0.402 (0.323-0.483)
Winter 2018	DWLS	0.183 (0.110-0.262)*
Summer 2018	SNP	0.460 (0.405-0.515)
Overall winter (2017 & 2018)	SNP+DWLS	0.288 (0.238-0.337)
Overall summer (2017 & 2018)	SNP+DWLS	0.459 (0.414-0.505)

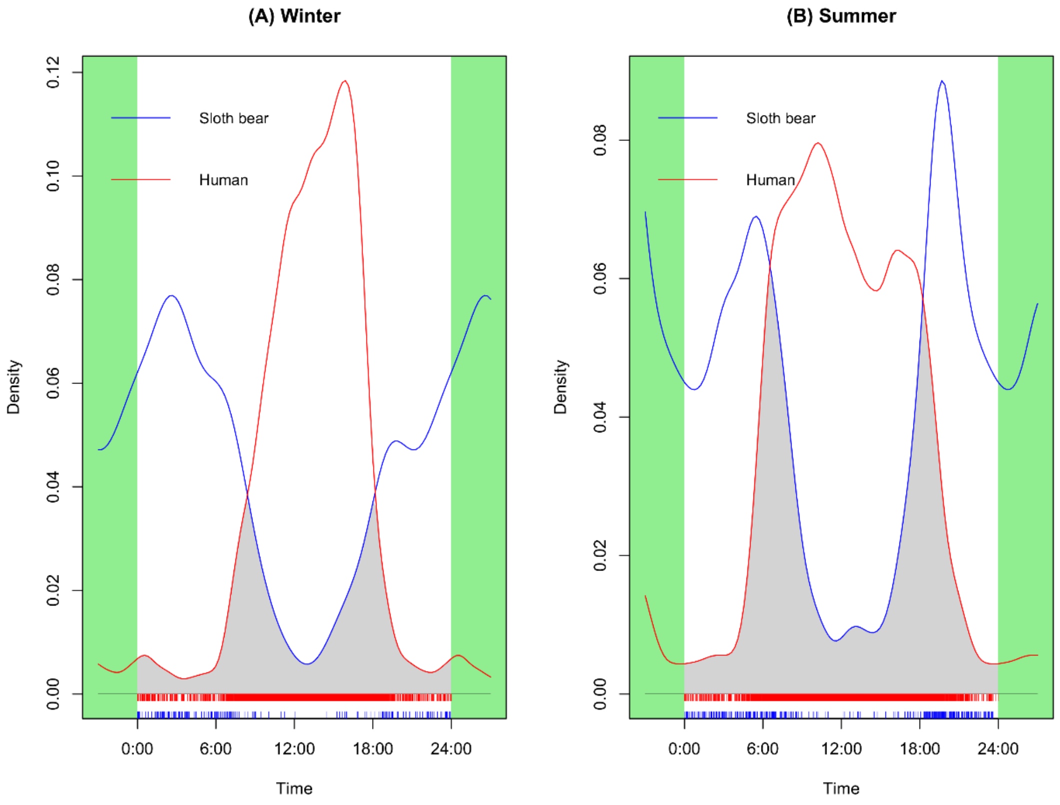

Figure 6.2. Activity overlap and overlap coefficient (Δ_4), i.e. grey shaded area, between sloth bear (blue solid line) and human (red solid line) for overall winter (**A**) and summer (**B**) in Sanjay Tiger Reserve, Madhya Pradesh, during 2017-2018.

Table 6.4. Circular statistics (including one-sample and multi-sample tests) of seasonal patterns of temporal activity of sloth bears and humans during 2017-2018 in Sanjay Tiger Reserve, Madhya Pradesh.

Season	**Variables**	**Sloth bear**	**Human**
Winter	Mean vector	01:45	13:40
	95% confidence interval	12:48 – 02:42	13:35 – 13:46
	Watson's U^2 one sample (p value)	0.09 (p<0.05)	2.25 (p<0.005)
	Watson's U^2 two sample (p value)	9.84 (p<0.001)	
Summer	Mean vector	00:18	12:09
	95% confidence interval	11:23 – 01:13	11:59 – 12:19
	Watson's U^2 one sample (p value)	0.51 (p<0.005)	3.85 (p<0.005)
	Watson's U^2 two sample (p value)	11.13 (p<0.001)	

Sloth bear-human conflict prediction through geographic profiling

Sloth bear-human conflict incidents

A total of 58 records of attacks (57 injured, one fatal; ~5 attack incidents/year) on humans by sloth bears occurred in the core area of STR from 2009 to 2019. During winter, 37.9% of the total attack incidents occurred, followed by 34.5% in monsoon and 27.6% in summer. Most of the victims were males (88%), compared to females (12%). Out of 58 attack sites, 13 were located in the villages (either in revenue lands or in agricultural fields). A total amount of 12,80,368 INR (average 22,075 INR; range 400-2,75,612 INR) was paid to the victims or victims’ family as compensation during 2009-2019 (1,28,036 INR/year).

Due to one repetitive location, centroids of compartments were generated for 57 conflict incidents for geographic profiling of sloth bear-human conflict.

Geographic profiling for conflict

The first model was run to produce the overall geographic profile of sloth bear-human encounters in STR using locations (n=57) of bear attacks on humans and 88 camera trap locations that detected both sloth bears and humans. The top 10% of the probable source sites (n=8) were mentioned in Table 6.5 and Figure 6.3. The Gini coefficient of the model was 0.97. For the second model, the seasonal shift in the probable conflict sites was assessed using 81 and 66 camera trap locations for summer and winter, respectively. The model identified nine sites and five sites as probable sources of conflict in summer and winter, respectively, as mentioned in Table 6.6, Figure 6.4 and

Figure 6.5. The Gini coefficient for summer and winter models were 0.90 and 0.99, respectively.

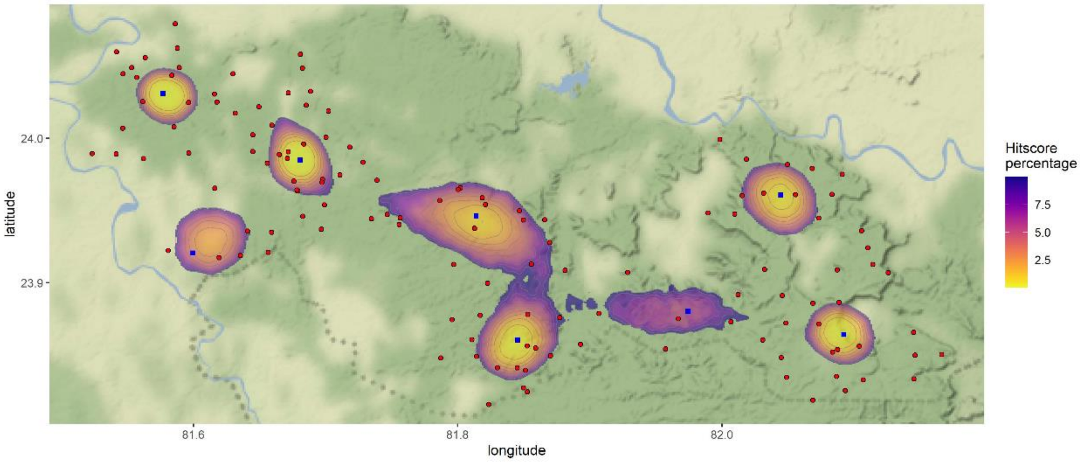

Figure 6.3. Map showing the top 10 % of the probable sloth bear-human conflict areas (probable source sites and dispersion or hotspots around source sites) in Sanjay Tiger Reserve, Madhya Pradesh, India, based on locations of sloth bear attacks on humans (from 2009 to 2019) and camera trap survey (locations where sloth bears and humans were both photo-captured) conducted during 2017-2018; blue square points (n=8) were probable source locations (or anchor points) of conflict; red points were the camera trap locations which photo-captured both sloth bears and humans; lighter colour shade indicated high conflict probabilities.

Table 6.5. Hit score percentages of the probable source locations estimated from the Dirichlet Process Mixture (DPM) modelling using conflict (sloth bear attacks on humans from 2009 to 2019) and camera trap locations (where sloth bears and humans were both photo-captured) as incident sites in Sanjay Tiger Reserve, Madhya Pradesh, India, during 2017-2018.

No. of source sites	**Latitude**	**Longitude**	**Hit score %**
1	23.98	81.68	0.00
2	24.03	81.58	0.05
3	23.87	82.09	0.17
4	23.86	81.85	0.25
5	23.96	82.04	0.54
6	23.95	81.81	1.06
7	23.92	81.60	3.93
8	23.88	81.95	6.57

Table 6.6. Hit score percentages of the conflict source locations estimated from the Dirichlet Process Mixture (DPM) modelling for summer and winter seasons in Sanjay Tiger Reserve, Madhya Pradesh, India, during 2017-2018.

Season	No. of clusters	Latitude	Longitude	Hit score %
Summer	1	23.87	82.10	0.02
	2	23.87	81.85	0.04
	3	24.01	81.56	0.49
	4	23.95	81.68	0.55
	5	23.97	82.05	1.15
	6	23.95	81.82	3.55
	7	23.94	81.75	7.77
	8	23.87	81.97	13.53
	9	24.05	81.68	27.28
Winter	1	23.86	82.08	0.01
	2	23.89	81.85	0.13
	3	24.02	81.60	0.57
	4	23.96	82.06	1.05
	5	23.96	81.72	1.28

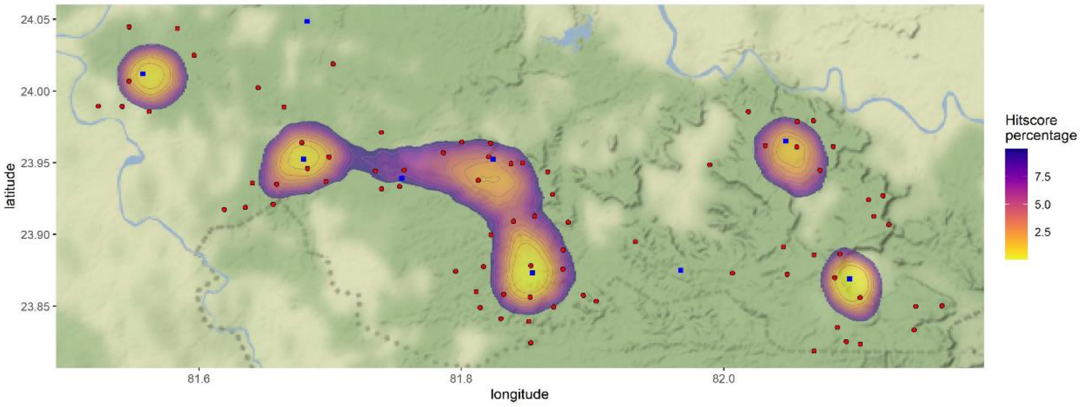

Figure 6.4. Map showing the top 10 % of the probable sloth bear-human conflict areas [probable source sites (n=9) and dispersion or hotspots around source sites] in summer in Sanjay Tiger Reserve, Madhya Pradesh, India.

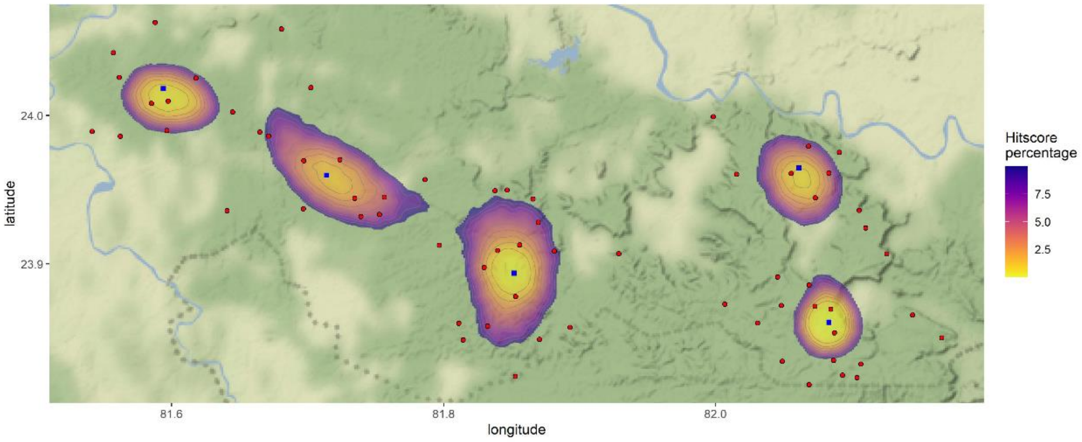

Figure 6.5. Map showing the top 10 % of the probable sloth bear-human conflict areas [probable source sites (n=5) and dispersion or hotspots around source sites] in winter in Sanjay Tiger Reserve, Madhya Pradesh, India.

A third model was run using 88 camera trap locations where sloth bears and humans were detected together to test the efficacy of camera trap data for conflict prediction (Figure 6.6). A profile based on camera trap data alone produced a similar conflict prediction. It was able to identify 21% of the sloth bear attacks on humans (12 attack sites out of 57) by searching less than 10% of the study area and 50% of these conflicts (29 attack sites out of 57) by searching just 20% of the study area (Table 1 of Annexure 1). Thus, it validated the model performance (Gini coefficient=0.91).

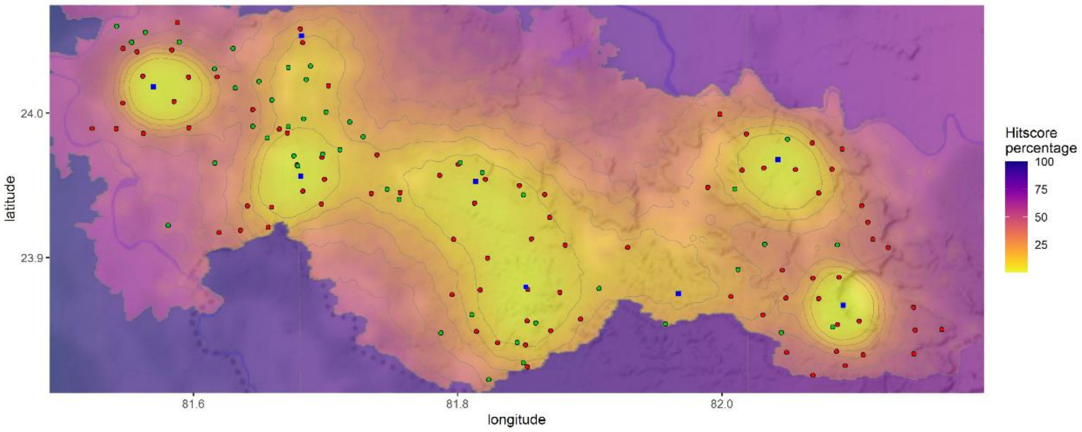

Figure 6.6. Prediction of source-dispersion of sloth bear-human conflict using camera trap survey conducted in Sanjay Tiger Reserve during 2017-2018, Madhya Pradesh, India; the lighter shaded areas indicate high conflict probable areas, and the blue squared points were the probable source sites (n=8) of conflict hotspots; light green points were the conflict sites where attacks on humans by sloth bears occurred between 2009 to 2019; red points were the camera trap locations which photo-captured both sloth bears and humans; twenty-nine conflict sites are falling within <20 % of the search area and 12 conflict sites within <10 %, thus validating the model performance.

6.4 Discussion

This chapter describes the spatial interaction and spatio-temporal overlap between sloth bears and humans with respect to seasons, i.e. winter and summer in STR. Sloth bears were found to independently co-occurred with humans in terms of space use, which was in contradiction to my expectations. However, distinct seasonal variation of spatial and temporal overlap between sloth bears and humans was observed at a finer scale, which perhaps facilitates sloth bears to co-exist with humans in a human-dominated landscape. Finally, in the context of extensive human activities in sloth bear habitat and resultant conflict incidents, I identified the probable conflict source sites and hotspots in STR based on available conflict records and detections of sloth bears and humans from camera traps.

Spatial interaction and spatio-temporal overlap between sloth bear and human

Two-species occupancy models showed that sloth bears could independently occur with humans irrespective of seasons, which, in my knowledge, is not an entirely novel situation. In Mudumalai Tiger Reserve, the occupancy of sloth bears was uninfluenced by the distance to the nearest human settlements (Ramesh et al., 2012a). Sloth bears can be habituated to persistent human presence and attracted to the human settlements in search of abundant seasonal fruits (*Ziziphus mauritiana*, *Syzygium cumini*), cultivated crops (groundnut, maize) and even garbage (food wastes) (Akhtar et al., 2004; Palei et al., 2020; Prajapati et al., 2021), which also increased the sloth bear-human conflict (Bargali et al., 2005; Prajapati et al., 2021). However, increased human activity or encounters with humans would facilitate the shifting of temporal activities of many mammalian species, including carnivores (Gaynor et al., 2018), instead of spatial avoidance (Nickel et al., 2020) or could induce habituation towards human activity (Albert and Bowyer, 1991; Barnes Jr and Wilker, 2000; Elfström et al., 2014). On the other hand, bears can also strongly avoid human activities both spatially and temporally (Chi and Gilbert, 1999; Tollefson et al., 2005; Ladle et al., 2018). I believe the spatial distribution of humans or anthropogenic activities could be the governing factor of such spatial avoidance or spatial co-occurrence. In the human-dominated landscape, it is not always possible to avoid humans spatially as it would cost to miss out on potential resource-rich areas, which would negatively affect the foraging success and overall fitness of the animal. In STR, human occurrence and livestock grazing in sloth bear habitats are extensive. I found high occupancy (Ψ^{A} = 0.96 ± 0.02 in winter; Ψ^{A} = 0.84 ± 0.03 in summer) of humans in STR, which indicated a substantial level of anthropogenic pressure throughout the study area, regardless of seasons. Despite this extensive human pressure, the probability of use of sites by sloth bears was also high (Ψ^{BA} = 0.71 ± 0.04 in winter; Ψ^{BA} = 0.79 ± 0.04 in summer; Table 6.2), which indicated that irrespective of seasons, sloth bears are utilizing sites independently with humans. It was also found that detection SIF (delta, $\boldsymbol{\delta}$) was slightly higher than 1 ($\boldsymbol{\delta}$ = 1.05 ±0.06 in winter; $\boldsymbol{\delta}$ = 1.06 ± 0.07 in summer; Table 6.2), indicating some sort of association in detection between sloth bears and humans. It could be attributed to the relatively high preference for trail use by wide-ranging carnivores (Harmsen et al., 2010; Karanth et al., 2011; Kays et al., 2017; Nickel et al., 2020) apart from humans, which would facilitate animals to cover a larger distance in an energetically efficient way (Dickie et al., 2017).

However, spatial overlap estimation based on Pianka's overlap index (Pianka, 1974) revealed seasonal variation between sloth bears and humans. The intensity of space use by sloth bears was lower (0.31) during summer than in winter (0.44), indicating that bears used sites less frequently during summer in the presence of humans than in winter. Given the widespread distribution of human activities during winter and summer, it was expected to have more overlapped areas between the two, without any seasonal variation. Less spatial overlap between sloth bears and humans during the summer season could be attributed to comparatively high temporal overlap during summer with respect to winter (see Table 6.4). Human activities were primarily diurnal regardless of seasons in STR. However, during summer, human activities started well before 06:00 in the morning and continued till the late evening hours, which substantially overlapped with the crepuscular activity of sloth bears (Yoganand, 2005; Bargali et al., 2012; Ramesh et al., 2013). Increased human activities in the early morning and crepuscular hours during summer (especially from March to the first half of April) are primarily associated with the collection of flowers of Mahua, which is used to prepare alcoholic beverages (Kumar and Rao, 2007; Mewada, 2012). The selling of dry Mahua flowers significantly helps in the income generation of indigenous people (Islam and Quli, 2017). Hence, it is widely considered a vital livelihood option for indigenous people in India (Kumar et al., 2018; Mishra and Poonia, 2019). The Mahua flowers also significantly contributed to the diet of sloth bears during the onset of summer (Bargali et al., 2004; Yoganand, 2005), which, coupled with its crepuscular activity, increased the interaction between sloth bears and humans (Akhtar et al., 2004). In STR, based on my camera trap survey, it was found that the frequency of occurrence of sloth bears and humans at per hour interval was high during the early morning (05:00-07:00) and evening (19:00-20:00) in summer (Figure 6.7). Sloth bears used sites less intensively where chances of human encounters were high during the crepuscular period. Contrastingly, during winter, high spatial overlap could be associated with low temporal overlap between sloth bears and humans. Like summer, human activities were not found to start in the early morning hours during winter but remained more active during broad daylight hours for a comparatively short time. In line with this, morning (07:00-08:00) and early evening hours (15:00-16:00) showed the maximum frequency of occurrence of sloth bears and humans at per hour interval; however, the magnitude of the frequency of occurrence was less compared to summer (Figure 6.7).

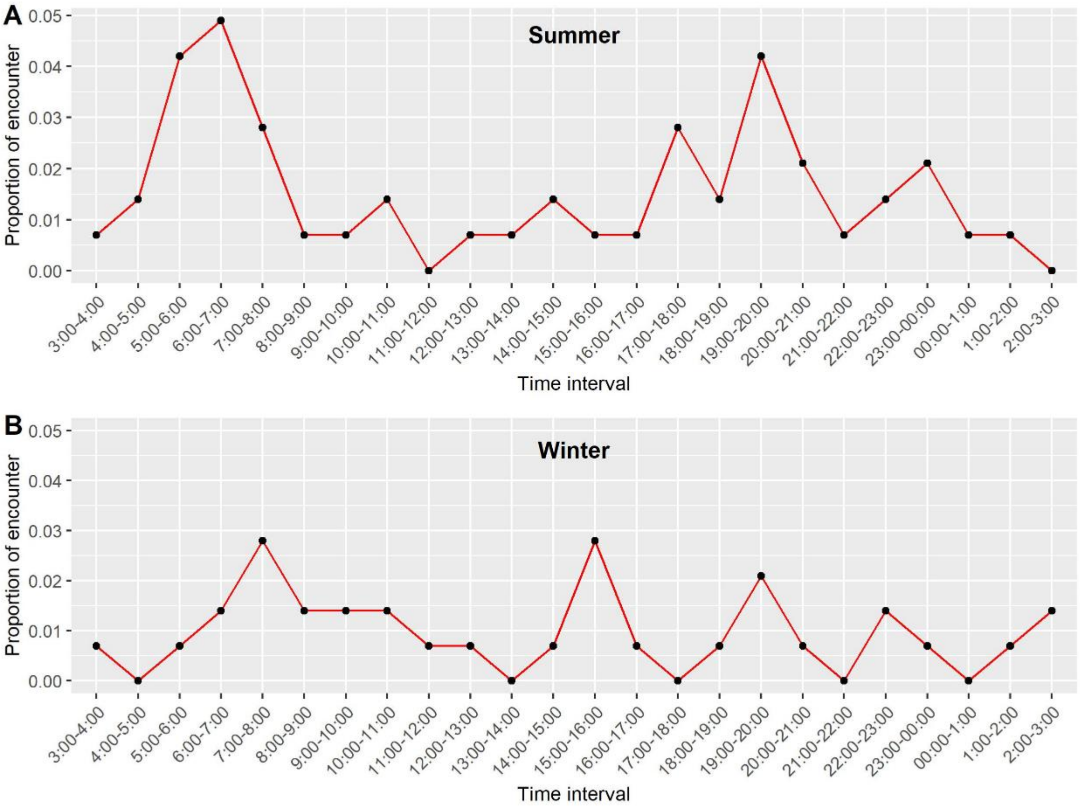

Figure 6.7. Frequency of occurrence (obtained from camera trap survey) between sloth bear and human at per hour interval of a 24-hour cycle during overall summer (**A**) and winter (**B**) in Sanjay Tiger Reserve, Madhya Pradesh, India, during 2017-2018.

It was found that human activity patterns varied significantly between seasons, which was the case for sloth bears. My result indicated that sloth bears used sites more extensively during winter regardless of human occurrence, with a distinct temporal avoidance. Studies elsewhere showed that carnivores could avoid human occurrence temporally (Tollefson et al., 2005) instead of avoiding spatially in a human-dominated landscape (Wang et al., 2015; Nickel et al., 2020). The marginal difference ($p<0.05$) in the activity of sloth bear between seasons indicated that sloth bears tended to avoid a more predictable pattern of human activity, despite already being strictly nocturnal and crepuscular in a human-dominated landscape (Bargali et al., 2012). Since human activity is not dependent on sloth bear activity, my study revealed that a significant shift of human activity between seasons influenced the likelihood of interaction between sloth bears and humans temporally and spatially.

However, despite distinct spatio-temporal avoidance, the conflict between sloth bears and humans in terms of human-mauling is prevalent in STR. It is attributed to the year-round immense anthropogenic pressure posed by human settlements inside the core area of STR. Similar observations were also made in human-dominated landscapes of central India (Bargali et al., 2005; Dhamorikar et al., 2017) and elsewhere (Garcia et al., 2016; Debata et al., 2017; Singh et al., 2018). Based on the camera trap survey, I found that during summer, the number of detections of sloth bears and humans per hour interval was higher (n=53) than in winter (n=33), indicating that during summer, there is a relatively high probability of conflict incidents inside the forest, as reported by Dhamorikar et al. (2017). I assumed that sites where sloth bears and humans were detected more frequently within a specific bin (1 hour in this case) of time are more prone to accidental encounters between them, while humans are engaged in different activities, especially in and around those sites. In previous studies, it has also been observed that most of the attacks happened during sudden close encounters with sloth bears (Dhamorikar et al., 2017; Singh et al., 2018), especially while collecting non-timber forest produce (NTFP) separately and silently inside the forest (Dhamorikar et al., 2017).

Identification of existing and future conflict hotspots

Finally, I have attempted to identify the conflict source sites and dispersion around source sites based on the present camera trap survey and records of conflict incidents obtained from the forest department. My primary aim was to identify areas inside the forest where humans and sloth bears occurred more frequently, assuming those areas had a high chance of conflict. Furthermore, I used records from the forest department to validate the model and future conflict predictions. Based on detections of sloth bears and humans obtained from camera traps and validating them with registered conflict records, I found that the GP model produced a reasonably good prediction (Gini coefficient = 0.92) of conflict source and dispersion. The GP model was found to be especially robust when fewer data points are provided and can potentially out-compete any other spatial statistics used in different contexts and scenarios of model-based predictions (Stevenson et al., 2012). However, a distinct seasonal variation was observed in the current prediction of conflict hotspots. A higher number of source sites (n=9) of probable conflict were identified during summer in comparison to winter (n=5), indicating a greater chance of conflict in forest areas during summer, which was also earlier reported from the central Indian landscape (Dhamorikar et al., 2017). It could be attributed to the much higher number of detections of sloth bears (n=312) during summer compared to winter (n=180) and associated high temporal overlap with humans. I found that seasonal variation in the temporal overlap between sloth bears and humans was also reflected in the predicted source-dispersion modelling of conflict in STR. As probable source sites and hotspots were primarily associated with villages' locations, it was evident that the probability of conflict was also higher in areas adjacent to villages. Relocation of villages from the core zone of STR, especially those situated around the probable source sites, would be an initial step for managers to reduce conflict between sloth bears and humans. However, despite being a contentious and gradual process, voluntary relocation of villages from the core zone of Tiger

Reserves was proven to be one of the successful means of restoring wildlife population and its habitat (Lasgorceix and Kothari, 2009), with special reference to tiger conservation in India (Jhala et al., 2021). Also, elsewhere in India, communities living inside Tiger Reserves preferred to be relocated willingly due to the decreased productivity of forests, lack of basic amenities and human-wildlife conflict (Harihar et al., 2014). Thus, I believe that the findings of geographic profiling could be one of the means for forest managers to identify the villages that need to be relocated on a priority basis or rationalize resources or techniques for conflict mitigation. However, in between the process of relocation, the effort could be made to aware people living in conflict hotspots, regarding the avoidance of conflict scenarios based on findings of this study and studies carried out elsewhere in central India (Bargali et al., 2005; Dhamorikar et al., 2017, 2018). Although I could not identify the conflict sources and hotspots outside the core zone of STR, conducting awareness programs among the local people as mitigation measures is especially important in the buffer zone and other human-dominated forest areas outside the Tiger Reserve.

Caveats and limitations

My study had several limitations. I could not broaden my understanding of conflict prediction in the buffer zone of STR due to logistic constraints. Also, camera traps were not deployed during the monsoon, which restricted us from drawing inference during this season. I have deployed camera traps on the trails and forest roads to maximize the capture of tigers and other co-predators, including sloth bears. Previous studies showed that the placement of camera traps on trails or forest roads significantly increased the detection of certain mammal species and bears as well (Wearn et al., 2013; Kolowski and Forrester, 2017; Tanwar et al., 2021). On the other hand, human activities inside the forest, such as the collection of NTFPs, are not restricted to the roads and trails but anywhere in the forest, depending upon resource availability and accessibility. In general, people do not walk alone on forest roads or trails, especially during dark hours, as moving in groups would reduce the chance of being attacked by bears (Garrote et al., 2017). Despite knowing these facts, I assumed that in the locations where humans and bears were frequently detected, the chance of conflict could also be high in and around those locations. Although widely used, the efficacy of trail-based camera traps in estimating anthropogenic pressure has rarely been assessed. However, Miller et al. (2017) showed how wildlife monitoring cameras (on-trail) can also efficiently capture human activities and could be used for long-term wildlife-human interaction-based studies. My study indicated fairly accurate conflict prediction through the GP model by using only camera trap data; however, I recommend careful consideration while deploying camera traps (on-road and off-road) in future similar studies.

6.5 Conclusion

The present study revealed that interaction between sloth bears and humans was primarily governed by the seasonal variation of spatio-temporal overlap, especially the significant changes in human activity between seasons. From the species perspective, fine-scale segregation of temporal and spatial activity was observed to co-exist with humans. However, extensive anthropogenic pressure throughout STR exacerbated the situation and led to conflict scenarios. Therefore, radio-telemetry-based research and monitoring through camera traps are recommended to understand the spatio-temporal activity of sloth bears in a human-dominated landscape. In this study, the application of novel methods like GP models provided an opportunity to predict source-dispersion of conflict precisely, even utilizing secondary data sets obtained from camera traps. It would be insightful for forest managers to reduce conflict incidents derived from the sloth bear-human interface with a more area-specific targeted approach and promote coexistence in a human-dominated landscape.

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CHAPTER 7: SYNTHESIS AND MANAGEMENT RECOMMENDATIONS

7.1 Synthesis

The sloth bear is one of the least studied bears in the world, despite their broad geographic distribution in the Indian sub-continent. Limited information is available regarding sloth bears' ecological and conservation aspects except for a few, such as feeding ecology and human-sloth bear conflicts. Given its 'Vulnerable' status on the IUCN Red List and declining population trend, conservation action planning is necessary to safeguard the population of sloth bears, especially in human-dominated landscapes. However, an in-depth understanding of sloth bears' ecology is a prerequisite to devising conservation plans. In this context, my study attempted to fill the existing knowledge gap regarding the ecological and conservational aspects of sloth bears in a human-dominated landscape.

Firstly, the occupancy or the probability of site use of sloth bears in the core zone of STR was assessed as a pilot study and found that sloth bears were widespread in the study area. Contrary to my expectations, no fine-scale factors adequately explained the occurrence of sloth bears. However, more detailed observations regarding the abundance of sloth bears and associated governing factors revealed the influences of various environmental and anthropogenic factors on the abundance of sloth bears. Sloth bear, being an individually unidentifiable species, it was challenging to assess its absolute abundance without employing spatial capture-recapture methods, which is a gold standard for population estimation. I interpreted the abundance estimates of sloth bears as a relative measure, given the methodology I used in this study. However, in the absence of absolute abundance, precise, ecologically realistic relative abundance estimates would help generate baseline information. Apart from the knowledge of abundance, I found broadly similar governing factors which influenced the abundance as well as the seasonal space use pattern by sloth bears. Environmental factors such as forest and habitat types were mainly associated with forage availability, influencing the abundance of sloth bears and their space use patterns. Subsequently, findings from the diet profile of sloth bears were related to their space use patterns and abundance in particular forest or habitat types. I also found that sloth bears selected dens for day resting and parturition in areas with higher terrain complexity and sufficient concealment cover.

Since the overarching theme of my study was the conservation of sloth bears in human-dominated landscapes, understanding human dimensions in the ecology of sloth bears is necessary. I found that a relatively high number of sloth bears occur in areas further away from human habitations in STR. However, in a more detailed view, sloth bears used areas close to human habitations during winter but avoided those areas in summer, which could be attributed to the availability of opportunistic foraging in the villages. Food plant species for sloth bears are abundant in STR, but the diversity is low, and some of the abundant foods (fruits and flowers of *M. longifolia*, fruits of *B. cochinchinensis* etc.) were not realised by sloth bears, perhaps due to the competition with humans. The absence or fewer occurrences of such food items in the diet of sloth bears also indicated the same pattern. Moreover, the least occurrence of some important food plants, especially species of *Ziziphus* (such as *Z. mauritiana*) in the forest, led bears to the villages during winter, which increased the chance of human-bear interaction. Sloth bears also appeared to seek more concealed and inaccessible (to humans) den sites, perhaps as an outcome of human disturbance. In this camera trap-based study, sloth bears' photo captures were positively associated with human photo captures, which is an outcome of the placement of camera traps on forest roads but also indicated the (indirect) interaction between sloth bears and humans in STR. I dedicated the last objective of this study to understand the human-sloth bear interaction more precisely. Given the persistent human pressure, I assessed the interaction between humans and sloth bears with respect to their seasonal spatial and temporal overlap in STR. I found that sloth bears co-occur with humans through fine-scale seasonal spatio-temporal segregation. However, the conflict between sloth bears and humans also prevailed in STR, despite the spatio-temporal segregation. Hence, I predicted the conflict hotspots based on conflict (attacks on humans) records and camera trap data, which would be helpful in prioritising areas for conflict mitigation.

Lastly, based on this present study, I proposed some key management recommendations and the scope of future studies on sloth bears in human-dominated landscapes, which were mentioned below.

7.2 Key recommendations

1. Sloth bears strongly avoided degraded barren lands (mostly due to human pressure/livestock grazing) close to the villages, streams, and rivers. Habitat restoration activities in such areas with special reference to strategically planting

fruiting trees and shrubs (e.g. species of *Ziziphus*), which are less common in the forest, may reduce the seasonal human-bear interaction.

2. Since denning is intricately related to bears' physiological stress and reproductive success, it is imperative to identify the preferred sloth bear dens (day resting and natal den). Subsequently, necessary protection for those dens from human disturbance should be provided by imposing restrictions on human movement at potential denning habitats and raising awareness amongst local people explaining the importance of conserving such habitats. Highly used dens located near human settlements should be targeted first to undertake protection measures, and human activities (stone querying, livestock grazing) should be restricted.

3. In other human-dominated landscapes of central India, species of *Ficus* was one of the preferred food plants for sloth bears; however, in STR, the rarity of this species may have contributed to the less consumption of the fruits of *Ficus* sp. by sloth bears. Hence, strategic plantation of such key species could help reduce the resource competition between humans and sloth bears, as fruits of Ficus sp. are generally not collected by the local people.

4. The restriction of movement of humans is only possible if the primary focus is targeted toward the voluntary relocation of villages from the core zone of STR, which is a gradual and contentious process but hopefully is underway. However, my findings of geographic profiling in human-sloth bear conflict prediction could be one of the means for forest managers to identify the villages that need to be relocated on a priority basis or rationalise resources or techniques for conflict mitigation.

5. In between the process of village relocation, the effort could be made to aware people living in conflict hotspots to avoid conflict scenarios based on the findings of my study and studies carried out elsewhere in central India. Although I could not identify the conflict sources and hotspots outside the core zone of STR, conducting awareness programs among the local people as mitigation measures is especially important in the buffer zone and other human-dominated forest areas outside the Tiger Reserve.

7.3 Scope of future studies

As mentioned before, the shortfall of studies on the ecology of sloth bears poses a challenge to its conservation. Hence, long-term studies on sloth bears, based on radio-telemetry, camera traps and genetics, should be carried out, especially in human-dominated landscapes. In this study, I demonstrated the application of the N-mixture model to estimate the relative abundance of sloth bears from camera trap data. Site-structured models are rarely effective in detecting population changes over time, especially if knowledge of certain aspects of the species' ecology (e.g., home range, population density) is limited. In such cases, future studies to monitor sloth bears' population changes should emphasise the spacing between camera traps based on a more thorough understanding of their ranging patterns. Also, site-structured models should require a spatially extensive sampling area with several spatial replicates at each sampling unit (depending on the feasibility and size of the sampling unit) to assess the abundance with precision in the case of a low-density population of sloth bears. In addition, I highly recommend carrying out density estimation by applying a spatially explicit capture-recapture framework through genetics and comparing results with site-structured models and other suitable camera trap-based methods (USCR, SPA, and distance sampling). Since sloth bears' home range varies considerably, the knowledge of the ranging patterns of sloth bears would also assist in choosing the appropriate scale in the case of scale-specific habitat/resource selection studies. Such multi-scale approaches are often overlooked but essential to understanding the species-habitat relationship. Den sites should be identified based on radio-telemetry, in addition to the knowledge of forest staff and local people. Subsequently, I also recommend monitoring dens through infrared camera traps to more precisely understand the seasonal use of dens by sloth bears. I could not adequately assess the difference in microclimatic conditions between used and available dens with respect to the environment; hence, future studies should focus on this particular aspect of denning ecology of sloth bears. The feeding ecology of sloth bears should also be studied based on DNA metabarcoding to identify the food species more accurately, leading to a complete diet profile of sloth bears. Subsequently, I also recommend quantifying food items available in the environment (e.g. fruiting trees and fruit production, ants and termites etc.) for an in-depth understanding of food habits through an available-used framework in human-dominated landscapes. Lastly, I recommend further increasing this study's conservational aspect by identifying the causal factors (socio-economic and environmental) of human-sloth bear conflicts in and around the predicted conflict hotspots and preparing probable mitigation measures. In human-dominated landscapes, the behaviour of sloth bears should also be closely monitored through radio-telemetry and intensive monitoring through camera traps to devise a species-specific conflict mitigation plan.

ANNEXURES

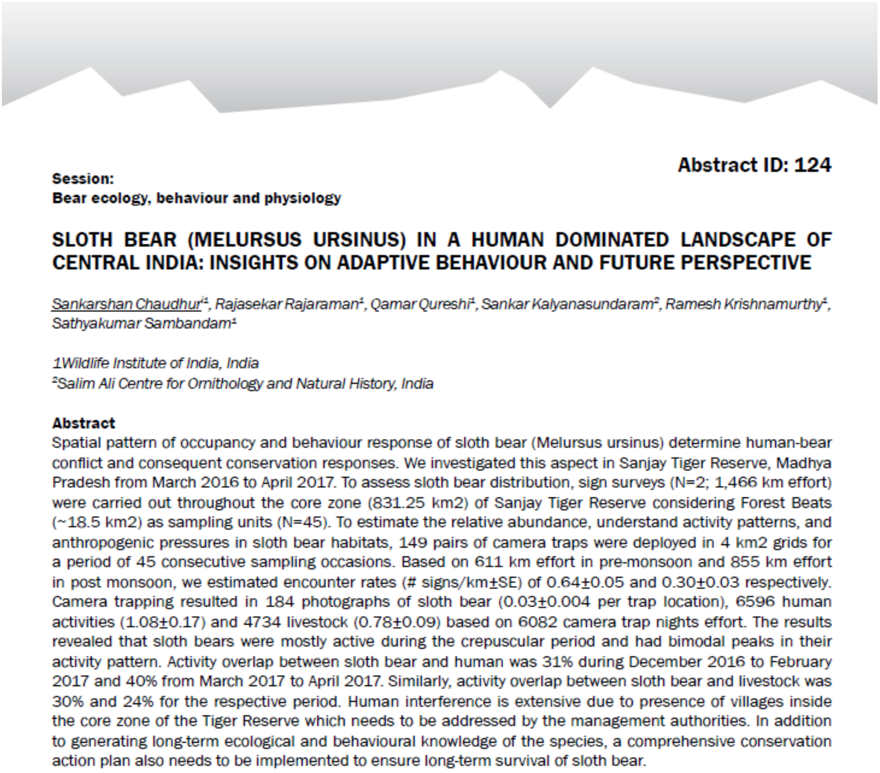
Annexure 1

Table 1. Hit score percentages (<20%) of the conflict source locations estimated from the Dirichlet Process Mixture (DPM) modelling for model validation, using only camera trap detection data of sloth bear and human detections in Sanjay Tiger Reserve, Madhya Pradesh, India, during 2017-2018.

No. of conflict sites	Latitude	Longitude	Hit score %
1	23.9634	81.67894	0.3044
2	23.97027	81.67593	1.1268
3	23.97027	81.67593	1.1268
4	23.97027	81.67593	1.1268
5	23.85187	82.08369	2.5324
6	23.85461	81.85903	3.7576
7	23.97154	81.6977	4.8512
8	23.95868	81.81871	6.4644
9	23.94338	81.84973	7.7692
10	23.98188	82.04964	8.3032
11	23.96554	81.8021	8.316
12	23.84112	81.84514	9.0092
13	23.99592	81.68338	10.636
14	23.99592	81.68338	10.636
15	23.97463	81.71072	11.0396
16	23.97463	81.71072	11.0396
17	23.99059	81.6718	11.9676
18	23.99059	81.6718	11.9676
19	23.99059	81.6718	11.9676
20	23.94011	81.75567	12.5264
21	24.02296	81.68533	12.5524
22	23.94726	81.74672	12.6992
23	24.03255	81.68866	13.68
24	23.87856	81.90701	14.4952
25	23.98263	81.65593	14.7516
26	23.86049	81.81064	15.1008
27	24.03144	81.67188	15.3176
28	24.00063	81.7003	15.5828
29	23.94742	82.00973	16.8996

Annexure 2

List of International Conferences Attended



Abstract ID: 124

Session:
Bear ecology, behaviour and physiology

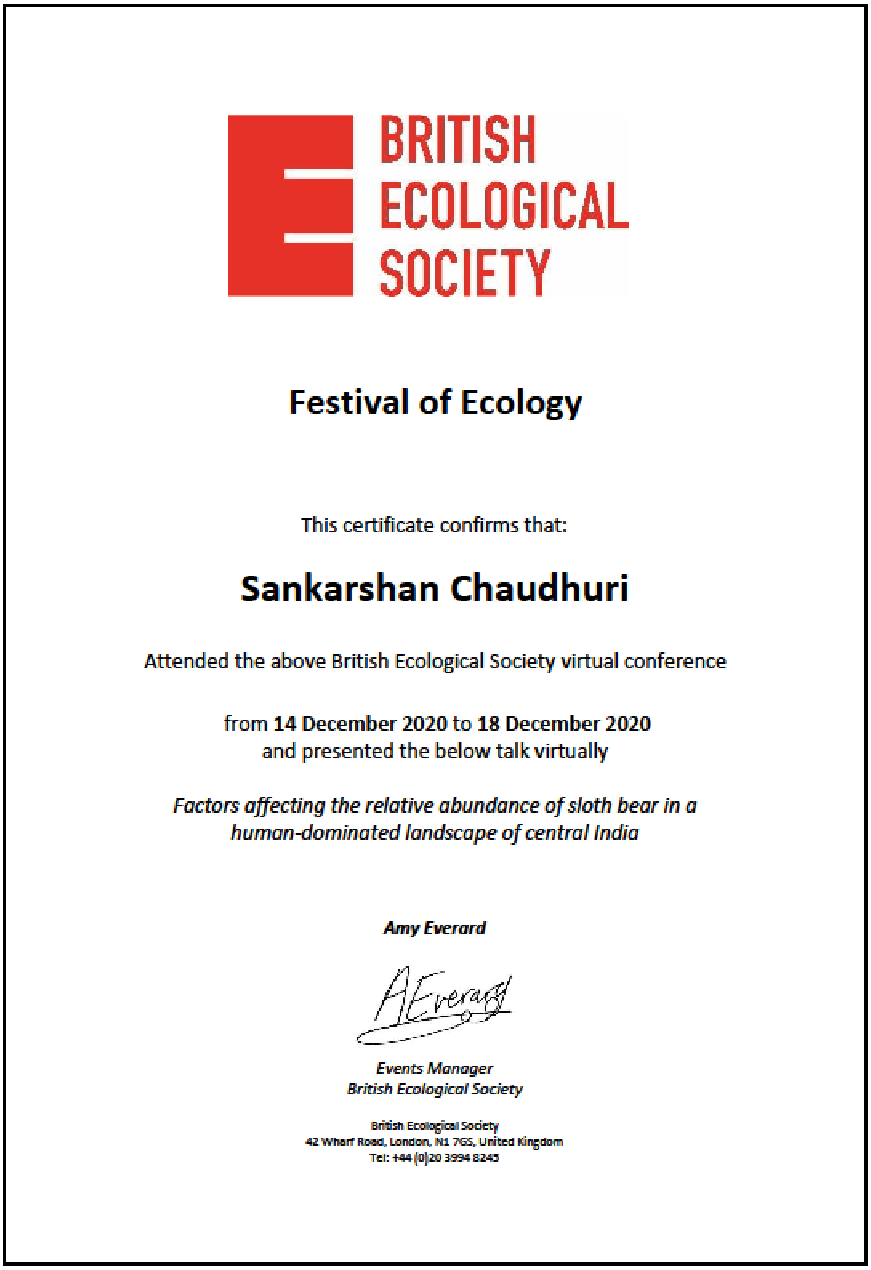
SLOTH BEAR (MELURSUS URSINUS) IN A HUMAN DOMINATED LANDSCAPE OF CENTRAL INDIA: INSIGHTS ON ADAPTIVE BEHAVIOUR AND FUTURE PERSPECTIVE

Sankarshan Chaudhuri[1], *Rajasekar Rajaraman*[1], *Qamar Qureshi*[1], *Sankar Kalyanasundaram*[2], *Ramesh Krishnamurthy*[1], *Sathyakumar Sambandam*[1]

1Wildlife Institute of India, India
[2]Salim Ali Centre for Ornithology and Natural History, India

Abstract

Spatial pattern of occupancy and behaviour response of sloth bear (Melursus ursinus) determine human-bear conflict and consequent conservation responses. We investigated this aspect in Sanjay Tiger Reserve, Madhya Pradesh from March 2016 to April 2017. To assess sloth bear distribution, sign surveys (N=2; 1,466 km effort) were carried out throughout the core zone (831.25 km2) of Sanjay Tiger Reserve considering Forest Beats (~18.5 km2) as sampling units (N=45). To estimate the relative abundance, understand activity patterns, and anthropogenic pressures in sloth bear habitats, 149 pairs of camera traps were deployed in 4 km2 grids for a period of 45 consecutive sampling occasions. Based on 611 km effort in pre-monsoon and 855 km effort in post monsoon, we estimated encounter rates (# signs/km±SE) of 0.64±0.05 and 0.30±0.03 respectively. Camera trapping resulted in 184 photographs of sloth bear (0.03±0.004 per trap location), 6596 human activities (1.08±0.17) and 4734 livestock (0.78±0.09) based on 6082 camera trap nights effort. The results revealed that sloth bears were mostly active during the crepuscular period and had bimodal peaks in their activity pattern. Activity overlap between sloth bear and human was 31% during December 2016 to February 2017 and 40% from March 2017 to April 2017. Similarly, activity overlap between sloth bear and livestock was 30% and 24% for the respective period. Human interference is extensive due to presence of villages inside the core zone of the Tiger Reserve which needs to be addressed by the management authorities. In addition to generating long-term ecological and behavioural knowledge of the species, a comprehensive conservation action plan also needs to be implemented to ensure long-term survival of sloth bear.



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participated and contributed a presentation titled

Patterns of human-sloth bear interaction and geographic profiling for conflict prediction in central India

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Antony J. Lynam
Tony Lynam
President, Society for Conservation Biology

Society for Conservation Biology

Annexure 3

Research Publications

Chaudhuri S, Bandyopadhyay M, Rajaraman R, Kalyanasundaram S, Sathyakumar S and Krishnamurthy R (2022). Spatio-Temporal Patterns and Source-Dispersion Modeling Towards Sloth Bear–Human Conflict Management in Central India. Front. Conserv. Sci. 3:850309. doi: 10.3389/fcosc.2022.850309.

Chaudhuri S, Rajaraman R, Kalyanasundaram S, Sathyakumar S, Krishnamurthy R. (2022). N-mixture model-based estimate of relative abundance of sloth bear (Melursus ursinus) in response to biotic and abiotic factors in a human-dominated landscape of central India. PeerJ (Accepted manuscript).

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Spatio-Temporal Patterns and Source-Dispersion Modeling Towards Sloth Bear–Human Conflict Management in Central India

Sankarshan Chaudhuri[1*], **Meghna Bandyopadhyay**[1], **Rajasekar Rajaraman**[1], **Sankar Kalyanasundaram**[2], **Sambandam Sathyakumar**[3] and **Ramesh Krishnamurthy**[1,4*]

[1] Department of Landscape Level Planning and Management, Wildlife Institute of India, Dehradun, India, [2] Ministry of Environment, Forest and Climate Change, Government of India, Salim Ali Centre for Ornithology and Natural History, Coimbatore, India, [3] Department of Endangered Species Management, Wildlife Institute of India, Dehradun, India, [4] Faculty of Forestry, University of British Columbia, Vancouver, BC, Canada

Edited by:
Alexandra Zimmermann,
University of Oxford, United Kingdom

Reviewed by:
Himanshu Shekhar Palei,
North Orissa University, India
Vijay Koli,
Mohanlal Sukhadia University, India

***Correspondence:**
Ramesh Krishnamurthy
ramesh@wii.gov.in
Sankarshan Chaudhuri
chaudhurisankarshan@gmail.com

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The impact of humans on biodiversity, in the form of the spatially extensive occurrence of humans and subsequent habitat degradation, leads to negative interactions between humans and native wildlife. However, knowledge of the spatial and temporal interface between humans and wildlife is necessary to understand the root cause of such negative interactions, yet considerably understudied in the context of human-dominated landscapes in south and south-eastern Asia. We took this opportunity, gaining insights on seasonal spatial interaction and spatio-temporal overlap between sloth bears (*Melursus ursinus*) and humans, and subsequently predicted the conflict source sites and dispersion (i.e., hotspots) based on the robust geographic profiling (GP) method in the Sanjay Tiger Reserve (STR), a human-dominated landscape of central India. Detection data of sloth bear and human were obtained from camera trap survey conducted for two years (2017–2018) and records of conflict incidents (2009–2019) were collected from forest department. We found that sloth bears can co-occur with humans independently of seasons, based on occupancy models. However, during summer, higher temporal overlap ($\Delta_4 = 0.46$) and lower spatial overlap (0.31) were observed between sloth bears and humans. Contrastingly, lower temporal overlap ($\Delta_4 = 0.29$) and higher spatial overlap (0.44) were observed between the same two during winter. The activity patterns of sloth bears and humans differed significantly across seasons and within the same species in different seasons. Our findings indicated that significant changes in human activity, especially during summer, increased the likelihood of sloth bear-human interaction and subsequent conflict incidents. The mapping of conflict source and dispersion (with high accuracy) also predicted a greater probability of conflict during summer, compared to winter, and thus showed the successful application of GP models in this field. Also, camera trap data alone were able to predict the occurrence of hotspots, demonstrating the use of camera trap records in the successful prediction of source-dispersion of conflict. This study would be useful for decision-makers to alleviate sloth bear–human conflict based on insights on seasonal variation of spatio-temporal overlap between the two and direct conservation efforts accordingly.

Keywords: camera trap survey, human-dominated landscape, interaction, conflict hotspot, *Melursus ursinus*, geographic profiling

1 INTRODUCTION

The ever-expanding human population and the resultant degradation and fragmentation of habitat have inadvertently affected wildlife and increased the proximity-driven interaction between humans and wildlife (Messmer, 2000; Messmer, 2009; Inskip and Zimmermann, 2009; Elfström et al., 2014; Joshi et al., 2016). Often, these human–wildlife interactions pose challenges to conservation initiatives due to the significant damage in the form of livestock depredation, human-mauling, crop loss, and retaliatory killing of the concerned species (Rajpurohit and Krausman, 2000; Madhusudan, 2003; Treves and Karanth, 2003; Bargali et al., 2005; Karanth et al., 2012). However, the identification of underlying reasons behind such conflict scenarios is necessary to address human–wildlife conflict (Sutherland et al., 2009). Among other species of wildlife, conflict with large carnivores (including bears) has a significant impact on the health and livelihood of humans worldwide (Treves and Karanth, 2003; Chapron et al., 2014). Human–carnivore interaction has been studied widely to understand the nature and extent of conflict and draw inferences on possible conflict mitigation (Redpath et al., 2013; Nyhus, 2016).

The expansion of anthropogenic activities has affected the global distribution of carnivores as well as their temporal and spatial use of the habitat, globally (Dirzo et al., 2014; Gaynor et al., 2018; Tucker et al., 2018). Also, humans impact directly or indirectly on the mortality and physiological fitness of carnivores (Rode et al., 2007; Darimont et al., 2015). However, it was observed that human–wildlife interaction could be quite complex and dependent on several factors (Tablado and Jenni, 2017). Wild animals, especially carnivores, can respond to increased or persistent anthropogenic activities in a variety of ways (Ordiz et al., 2011; Carter et al., 2012; Elfström et al., 2014; Tablado and Jenni, 2017; Suraci et al., 2019b), including strong fear responses (Suraci et al., 2019a). The response of carnivores towards human occurrences may be governed by the spatial extent and intensity of the anthropogenic pressure prevailing in the concerned landscape. In a human-dominated landscape, especially with year-round high occurrence of humans can affect the space use pattern of carnivores, leading to avoidance of areas where human disturbance is high (Chi and Gilbert, 1999; Fortin et al., 2016). Many wildlife species, including bears, consider humans as a perceived threat or predator as they elicit similar responses in the presence of natural predators (Joshi et al., 1999; Frid and Dill, 2002; Ciuti et al., 2012). However, bears were also found to be frequently attracted to human settlements for opportunistic foraging, especially crop raiding, livestock depredation, and feeding on garbage (food wastes) (Charoo et al., 2011; Morales-González et al., 2020; Prajapati et al., 2021). Despite these exceptions, it is likely that animals will avoid places where human pressure persists (Dröge et al., 2017). Similarly, if the temporal pattern of human occurrence is more predictable to the animals, it is also possible that animals would segregate their movement by increasing the time of activity when humans are least active (Suraci et al., 2019b), instead of avoiding spatially (Nickel et al., 2020), which would in turn facilitate co-existence (Fagen and Fagen, 1994; Fortin et al., 2016; Srivastava et al., 2020).

It is expected that human–wildlife conflict is a direct outcome of the high degree of human–wildlife interface (Pack et al., 2013). However, unlike in Europe and North American countries, in South and South-east Asian regions, the prevalence of human–wildlife conflict is primarily due to rich and endangered biodiversity and associated developmental activities in and around forest areas (Madhusudan and Karanth, 2002; Sodhi et al., 2010). A high degree of dependence on natural resources driven by poverty and an expanding human population impacts adversely on biodiversity (Sodhi et al., 2010; Chao, 2012) and subsequently increases conflict scenarios due to highly overlapping resource use (Treves et al., 2006). In this view, there is a paucity of studies that take into consideration the spatio-temporal interactions between humans and carnivores with global distribution, such as Asian bear species, to address the issues of conflict more effectively (Can et al., 2014; Anand and Radhakrishna, 2017).

The sloth bear (*Melursus ursinus*) is one of the four bear species found in the Indian subcontinent and endemic to this region (Prater, 1965; Johnsingh, 2003; Sathyakumar et al., 2012). Most of the sloth bear populations are restricted to the moist and dry deciduous forests of the Western Ghats and central Indian landscape (Yoganand et al., 2006). The sloth bear is a myrmecophagous (ant and termite eating) ursid, which is notably involved in conflict with humans in some parts of its distributional range, especially human-dominated landscapes (Rajpurohit and Krausman, 2000; Bargali et al., 2005; Akhtar and Chauhan, 2008; Garcia et al., 2016; Debata et al., 2017; Dhamorikar et al., 2017; Singh et al., 2018; Sharp et al., 2020; Prajapati et al., 2021). However, except for a few anecdotal evidences, attacks on humans by sloth bears are mostly defensive, rather than intentional or predatory in nature, as they perceive humans as threats or predators (Sharp et al., 2020). In human-dominated forests of central India, it was found that most of the victims belonged to tribal communities with high natural resource dependency and were attacked inside the forest while collecting non-timber forest produce (NTFP) (Dhamorikar et al., 2017), as well as adjacent to the village peripheries and agricultural lands (Dhamorikar et al., 2017; Singh et al., 2018). The seasonal and temporal patterns of these attacks varied between different landscapes (Bargali et al., 2005; Dhamorikar et al., 2017; Singh et al., 2018). Sloth bears are mainly nocturnal and crepuscular in activity (Yoganand, 2005; Bargali et al., 2012; Ramesh et al., 2013) but could be active throughout the daylight hours in some parts of its distribution (Joshi et al., 1999). It was believed that the majority of the conflict cases happened due to the overlapped activity between sloth bears and humans during the crepuscular period (Akhtar et al., 2004; Bargali et al., 2005). However, there is a shortfall of studies which describe the fine-scale interaction and overlap (spatial and temporal) between sloth bears and humans utilizing the spatially independent point locations (such as camera traps) in a human-dominated landscape. Also, to our knowledge, no research work has been conducted to predict probable conflict sources and hotspots based on occurrence records of sloth bears and humans.

In this study, we used the season-specific detection records of sloth bears and humans, obtained from a camera trap survey conducted in the Sanjay Tiger Reserve (STR), central India, to understand the interaction between sloth bears and humans. Firstly, based on detection and non-detection data of sloth bears and humans, two-species occupancy models (Richmond et al., 2010) were developed to assess the species interaction in terms of association or segregation. Furthermore, the extent of spatial and temporal overlap was estimated using the relative abundance index (RAI) (O'Brien et al., 2003; Rovero and Marshall, 2009) of sloth bears and humans. We expected that sloth bears would avoid humans spatially regardless of seasons, as evident from other studies (Paudel et al., 2022; Pokharel et al., 2022). However, we also assumed that sloth bears would be more nocturnal in disturbed habitats to avoid human encounters during daylight hours. Finally, we used geographic profiling/geo-profiling (GP) to identify conflict sources and dispersion (i.e., hotspots), a novel approach to tackle wildlife crime and predict conflict hotspots (Faulkner et al., 2018; Struebig et al., 2018), which was earlier only restricted to criminology (Rossmo, 2000). With this background, we aimed to understand the seasonal spatial interaction and spatiotemporal overlap between sloth bears and humans in the context of a human-dominated landscape. Subsequently, we also attempted to identify probable conflict source sites and hotspots that can serve as a basis for implementing sloth bear–human conflict mitigation strategies.

2 MATERIALS AND METHODS

2.1 Study Area

This study was conducted at the Sanjay Tiger Reserve (STR), which is spread over two districts, i.e., Sidhi and Shahdol, in the Indian state of Madhya Pradesh, central India (**Figure 1**). As per the biogeographic classification of India by Rodgers & Panwar (1988),

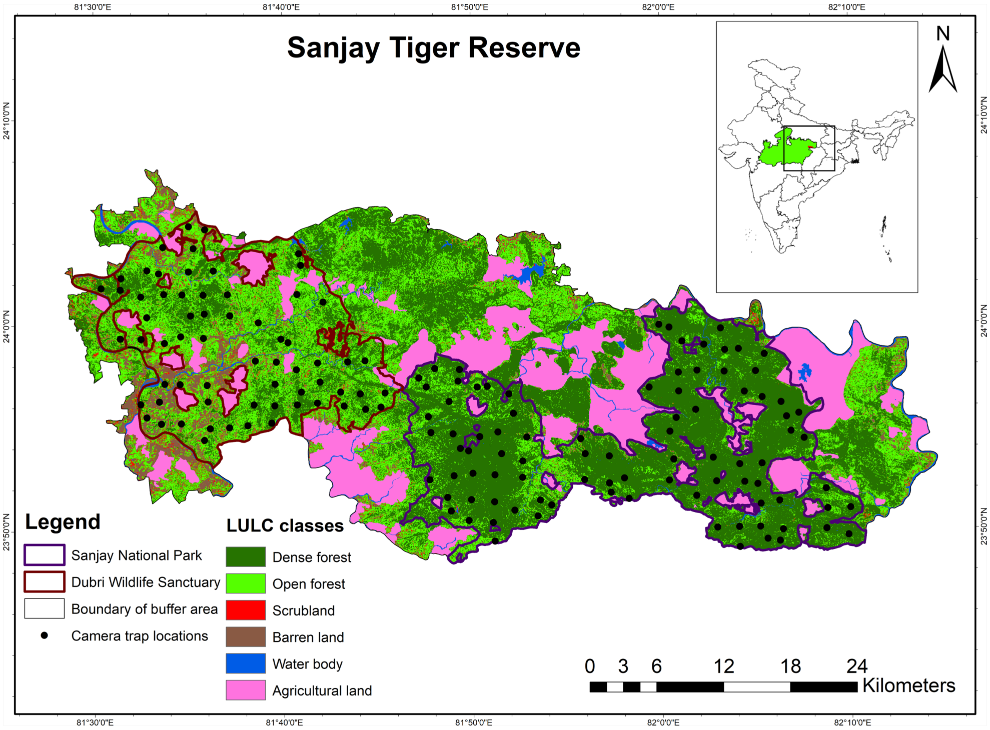

FIGURE 1 | Location of Sanjay Tiger Reserve (STR) in India, along with land use land cover (LULC) classes and camera trap locations.

STR comes under the “Central Highlands.” Sanjay National Park (SNP) and Dubri Wildlife Sanctuary (DWLS) together constitute the core area (831.25 km^2) of STR. Surrounding the core, an additional area of 812.58 km^2 was demarcated as a buffer zone for STR.

The forest type of STR is dominated by *Shorea robusta* (Sal). It falls under the sub-group "North Indian moist deciduous forests", with subtype "Moist sal-bearing forest" and subdivision "Moist peninsular sal forest", according to the revised classification by Champion & Seth (1968). Other major tree species found in STR include *Buchanania cochinchinensis* (Char), *Diospyros melanoxylon* (Tendu), *Madhuca longifolia,* (Mahua), *Lagerstroemia parviflora* (Sedha), and *Cassia fistula* (Amaltas). Land use land cover types of STR are categorized into six classes, i.e., dense forest, open forest, scrubland, barren land, water bodies, and agricultural land (**Figure 1**). The elevation of the STR varied from 239 to 770 m. The topography is generally undulating and hilly in the eastern part and moderately sloped to flat terrain in the western part with varying slopes, ranging from 0° to 61°. Temperature varies between 7.4 and 41.8°C with distinct seasonal variation viz. summer (March to June), monsoon (July to October), and winter (November to February). The mean annual precipitation in STR is 1,303 mm. Apart from sloth bear, STR harbors diverse mammal species such as the *Panthera tigris* (Tiger), *Panthera pardus* (Leopard), *Hyaena hyaena* (Striped hyena), *Canis lupus* (Indian wolf) *Cuon alpinus* (Asiatic wild dog), *Axis axis* (Chital), *Rusa unicolor* (Sambar), *Tetracerus quadricornis* (Four-horned antelope), etc. There are thirty-nine villages inside the core area of the Tiger Reserve. People living in these villages, along with their substantial livestock population, are highly dependent on forest resources, which pose immense anthropogenic pressure on the core habitat of STR.

2.2 Camera Trap Survey

Systematic camera trapping exercises were conducted over two consecutive years, from December 2016 to April 2017 (henceforth, 2017) and January 2018 to April 2018, in the core area of STR. The core area was gridded into 4 km^2 blocks (henceforth, sites), following the Indian tiger estimation protocol (Jhala et al., 2015). A total of 143 and 144 sites were selected for the deployment of camera traps during 2017 and 2018, respectively. Due to logistical constraints, we divided the core area into two blocks, viz., SNP (466.6 km^2) and DWLS (364.6 km^2) for the deployment of camera traps. First, we deployed camera traps at 86 sites during winter 2017, i.e., December 2016 to February 2017 in SNP, followed by 57 sites in DWLS during summer 2017, i.e., March 2017 to April 2017. During winter 2018 (January to February) and summer 2018 (March to April), camera traps were deployed in 57 sites in DWLS and 87 sites in SNP, respectively.

Double-sided automated camera traps (Cuddeback C1 and Cuddeback Ambush) were placed preferably on dirt roads, dry riverbeds, and animal trails to maximize the detectability of large carnivores (Karanth et al., 2011), including sloth bears, at a height of 30–40 cm above the ground. Furthermore, the placement of double-sided camera traps, opposite to each other, at each site would preferably increase the detection probability of animals (Pease et al., 2016). Each camera trap was set to take photographs at a 15-second interval upon detection of the animals. The survey duration of camera traps at each block was continued for 45 days, and camera traps were active on a 24 × 7 basis. However, in 2017, due to rampant forest fires at DWLS during summer, camera traps were active for an average period of 36 days. Each camera trap location was recorded using Garmin e-Trex handheld GPS units (Garmin Inc.).

After retrieving the camera trap data, detections of sloth bears and humans were considered independent events (or independent photographs) if two consecutive photographs of each species were taken at a ≥30-minute interval, to limit the risk of pseudoreplication (O’Brien et al., 2003; Xiao et al., 2018; Haswell et al., 2020). We chose a 30-minute interval for humans as we expected a substantial number of repeated photo-captures of the same individual (single and groups) within a short period, due to the curiosity of people detecting camera traps. The entire survey duration (45 days) was collapsed into 15 temporal replicates per site by considering each three days as a single temporal replicate and pooling the detection–nondetection records of species accordingly. Collapsing the survey duration into a smaller number of temporal replicates facilitated the reduction of overdispersion (variance > mean) and increased the temporal independence of each detection (Penjor et al., 2019). Lastly, we used camera trap data from two summers and two winters obtained from two blocks representing the entire core area of STR for each season (overall winter and summer).

2.3 Spatial Interaction

In the first step, we used two-species conditional occupancy models (Richmond et al., 2010) to assess if sloth bears show any spatial avoidance toward human occurrence in two different seasons (winter and summer). The two-species occupancy model estimates the conditional probability of occupancy and detectability of a less dominant species in the presence of a dominant species (Richmond et al., 2010). We interpreted the “probability of occupancy” of a species as the probability of “use” since we believed that the size of the site would potentially violate the closure assumption and independence between sites (MacKenzie et al., 2017, p447). If species A and B are to co-occur and A is dominant over B, the following parameters are estimated in the two-species model:

i. Ψ^A (Probability of occupancy of A)
ii. Ψ^{Ba} (Probability of occupancy of B when A is absent)
iii. Ψ^{BA} (Probability of occupancy of B when A is present)
iv. p^A (Detection probability of A when B is absent)
v. p^B (Detection probability of B when A is absent)
vi. r^A (Detection probability of A when both A and B were present)
vii. r^{BA} (Detection probability of B when A was present and detected)
viii. r^{Ba} (Detection probability of B when A is present but not detected).

In this study, we assumed that human (species A) activity does not depend on the occurrence of sloth bear (species B). Sloth bears probably perceive humans as a threat equivalent to predators (tigers and leopards), with which they are sympatric (Laurie and Seidensticker, 1977; Yoganand, 2005). Given the extensive anthropogenic pressure in STR, it is very likely that sloth bears would adjust their space use in accordance with human occurrence. In this view, we assumed sloth bears to be a subordinate species in the two-species occupancy model. However, considering humans as the dominant species in two-species occupancy models was not common [but see Parsons et al. (2016)], our intention was to assess the degree of spatial avoidance of sloth bears in response to the extensive occurrence of a perceived threat (i.e., humans) in the context of the human-dominated landscape.

The species interaction factor (henceforth, SIF) is a derived parameter estimated from the two-species occupancy model. A value of SIF >1 shows the degree of species aggregation rather than avoidance, and an SIF value <1 indicates spatial avoidance or segregation, where two species are less likely to occur with each other than expected under the hypothesis of species independence. An SIF value of ~1 indicates that both species can occur independently. Following Richmond et al. (2010), SIF can be calculated as:

$$SIF = \frac{\Psi^{A}\,\Psi^{B\,A}}{\Psi^{A}\left[\Psi^{A}\,\Psi^{B\,A} + (1 - \Psi^{A})\,\Psi^{B\,a}\right]}$$

We calculated SIF and compared it between seasons to assess the avoidance of space used by sloth bears in the presence of humans. We also carefully inspected the values of other parameter estimates (Ψ^{Ba}, Ψ^{BA}, p^{B}, r^{BA} and r^{Ba}), and compared them between seasons. Model comparison and selection were based on Akaike's Information Criterion (AIC) (Burnham and Anderson, 2002). We used the program Presence (Version 2.13.11) (Hines and USGS-PWRC, 2006) to estimate occupancy and related parameters of the two species.

2.4 Spatial and Temporal Overlap

2.4.1 Spatial Overlap

In the second step, the extent of spatial overlap between sloth bears and humans was estimated for summer and winter as we were interested in assessing the interaction on a finer scale, which might not always be discernible from occupancy models. In order to do so, the relative abundance index (RAI; O'Brien et al., 2003; Rovero & Marshall, 2009) was calculated at each camera trap location for both species using the following formula:

RAI = (No. of independent photographs/total no. of days during which each camera trap was operating) × 100.

Considering each location of the camera trap as spatially independent of other locations, we calculated the spatial overlap by applying Pianka's overlap index (Pianka, 1974). This method takes account of the proportion of RAI values of a pair of species with respect to different camera trap locations. The output value (estimate of overlap) ranges between 0, i.e., no overlap and 1 or complete overlap. Although it was developed initially to estimate the niche overlap, several studies have applied Pianka's index to estimate spatial overlap between co-occurring species and human activities (Ramesh et al., 2012b; Yang et al., 2019; Mori et al., 2020; Zhao et al., 2020). We also estimated 95% confidence interval (CI) of the spatial overlap estimate from 10,000 bootstrapped replicates. All analyses related to spatial overlap estimation were carried out in R (R Core Team, 2019) using the package "spaa" (Zhang et al., 2013).

2.4.2 Temporal Overlap

To assess the activity pattern of sloth bears and humans for summer and winter, we considered the same camera trap locations that were used to compute the spatial overlap. Following the methodology described by Ridout & Linkie (2009), a non-parametric kernel density function was fitted to the circular time data obtained from the camera trap photographs of both species, which would produce the pairwise (sloth bear–human) activity patterns. Then, the coefficient of overlap (delta, Δ) was estimated from the area under the curve of two different density functions (sloth bear and human). Values of Δ ranged between 0 and 1, where 0 denotes no overlap and 1 denotes complete overlap (Linkie & Ridout, 2011; Meredith & Ridout, 2014). We calculated Δ_4 when the number of independent events of both sloth bear and human, were >75, except for one season (winter 2018), when, number of sloth bear photographs was <75 and hence, Δ_1 was calculated, as mentioned by Linkie & Ridout (2011). The 95% CI was generated by 10,000 bootstrapped samples of overlap coefficient to check the precision of the estimates (Linkie and Ridout, 2011). Analyses regarding the estimation of overlap coefficient were carried out in the package "overlap" (Meredith and Ridout, 2021), implemented in R (R Core Team, 2019).

One-sample Watson's U^2 test (Watson, 1961) is a goodness of fit test, which was performed to determine whether the distribution of detection time of each species significantly deviated from the uniform distribution of activity. Similarly, we performed two-sample Watson's U^2 tests (Watson, 1962) to assess if the activity distributions of sloth bears and humans were significantly different from each other between summer and winter. We also tested for the difference in activity of sloth bears and humans separately between seasons through two–sample Watson's U^2 tests. Both one-sample and two-sample Watson's U^2 tests were performed using the software "Oriana" (Kovach, 2011).

2.5 Geographic Profiling for Sloth Bear–Human Conflict

2.5.1 Sloth Bear–Human Conflict Incidents

Records of conflict incidents between sloth bears and humans in the form of human injuries and deaths were obtained from the forest department of STR for 10 years (2009–2019). We retained the records only for the core area of the STR, as our camera trapping exercise was conducted in the core area only. However, GPS locations of these incidents were generally not recorded by the forest department; only the name of the concerned forest beat and compartment was mentioned. As compartments are the smallest (average size of 2.12 km^2) administrative units of our study area (and all other Tiger Reserves), we generated centroids of compartments of forest beats where conflict incidents occurred and used this information for further analysis.

2.5.2 Geographic Profiling

Geographic profiling (GP) is a technique used to estimate the probable sources of spatial encounters (human–sloth bear encounters in this case) when the sources are unknown (villages or sloth bear territories). It works on the Bayesian framework based on a distance decay process (Verity et al., 2014). We used the dirichlet process mixture (DPM) (Verity et al., 2014; Faulkner et al., 2017) model to produce a geographic profile of sloth bear–human encounters using the package Rgeoprofile 2.1.0 in R (https://github.com/bobverity/Rgeoprofile). GP has been proven to be advantageous over conventionally used prediction models such as kernel density and ensemble modeling for mapping of invasive species as well as conflict or risk assessment (Stevenson et al., 2012; Struebig et al., 2018). The process consists of two parts: the incident locations where the conflict occurred and the source locations from where the conflict might have arisen. The model divides the incident locations into “n” clusters. Then the source sites are predicted using the Gibbs sampler (Geman and Geman, 1984) within a Markov Chain Monte Carlo framework until it converges on the posterior distribution of sigma (σ). Sigma (σ) is the standard deviation of the dispersion distribution, and it means how far we expect the sources to travel from the incident sites. There are two approaches to testing the model efficiency: one is by using the hit score percentage (HS%, ranging from 0 to 100%), and the second is using the Gini coefficient (G, ranging from 0 to 1) (Gini, 1921; Faulkner et al., 2018). The GP model provides an optimal search process of the source locations or anchor points. The hit score percentage (i.e., the proportion of the area covered by the "conflict/encounter" divided by the total area) was used to test the efficiency of the model, with a lower hit score value depicting more accuracy of the search process. The Gini coefficient was used to assess the model performance; the higher the Gini coefficient, the more accurate the geographic profile (Faulkner et al., 2018).

We produced a geographic profile of sloth bear–human encounters using a) conflict data of sloth bear attacks on humans and b) camera trap data where both the sloth bears and humans were detected together (but at different times). We used the compartment centroids for the conflict data and camera trap data consisting of only sloth bears or humans were excluded. Duplicate points were removed from the data as the DPM model tends to fit very low sigma values with repeated locations (Struebig et al., 2018). The mean sigma (σ) used for the analysis was 2 km [assuming 87% of the dispersal events occur within 2 km of the source; following Faulkner et al. (2017)] and we ran the full data set with locations of conflict and camera trap, using 10 chains with a burn-in of 10,000. Firstly, the resultant model will identify the probable conflict source sites, followed by conflict hotspots or dispersion around the source sites. Next, we compared any seasonal shift in the conflict-probable areas in STR using the summer and winter camera trap locations separately, where sloth bear and human both were detected. In this study, we assumed the camera trap data where both the sloth bears and humans were detected to be a probable conflict site. Therefore, to test the efficacy of camera trap data (n = 88) alone to predict the future probable conflict areas, model validation was conducted. We generated a geographic profile based solely on camera trap data (where both sloth bears and humans were detected). Sloth bear attack sites (n = 57) were then overlaid on the geographic profile layer to check whether the conflict sites were falling over the predicted probable conflict areas generated by the camera trap data.

3 RESULTS

A total of 179 and 311 independent detections of sloth bears were obtained from camera traps in winter and summer, respectively, during the entire survey period of 12,430 trap-nights (winter = 6,435 trap-nights; summer = 5,995 trap-nights). Detection of sloth bears was substantially higher in SNP (winter = 122; summer = 235) than in DWLS (winter = 57; summer = 76), irrespective of the season (**Figure 2**). Similarly, the detection of humans was found to be higher in SNP (winter = 3,396; summer = 2,565) than in DWLS (summer = 1,749; winter = 1,862) (**Figure 2**).

3.1 Spatial Interaction

During winter and summer, SIF values (SIF value ± Standard Error) were found to be 0.98 ± 0.007 (95% CI: 0.97–0.99) and 1.00 ± 0.03 (95% CI: 0.93–1.07), respectively (**Table 1**). Thus, no difference in species interaction was observed between seasons. The closeness of the SIF value toward 1 indicated no distinguishable pattern of spatial avoidance shown by sloth bears toward human occurrences. The probability of use of sites by sloth bears in the presence of humans was found to be less [Ψ^{BA} (SE) = 0.71 (0.04)] than when humans are absent [Ψ^{Ba} (SE) = 1.00 (0.00)] during winter. However, during summer, the probability of use of sites by sloth bears did not differ much [Ψ^{BA} (SE) = 0.79 (0.04); Ψ^{Ba} (SE) = 0.77 (0.17)] in the presence or absence of humans.

3.2 Spatial and Temporal Overlap

3.2.1 Spatial Overlap

A higher spatial overlap (0.44) between sloth bears and humans was observed during winter than in summer (0.31). A high variation in spatial overlap between sloth bears and humans was observed between seasons in DWLS compared to SNP (**Table 2**).

3.2.2 Temporal Overlap

It was found that, during summer, the temporal overlap between sloth bears and humans was higher in comparison to winter (**Table 3**; **Figure 3**). Sloth bears were found to be more crepuscular and nocturnal (Watson’s $U^2 = 0.09$; $p < 0.05$ for winter; Watson’s $U^2 = 0.51$; $p < 0.005$ for summer) irrespective of the seasons, whereas human activity was found to be primarily diurnal (Watson’s $U^2 = 2.25$; $p < 0.005$ for winter; Watson’s $U^2 = 3.85$; $p < 0.005$ for summer; **Table 4**). During summer, the activity of humans started before 06:00 and extended as long as 22:00 (**Figure 3**). However, during winter, human activities mostly started after 07:00 and continued till 20:00 (**Figure 3**). Watson’s

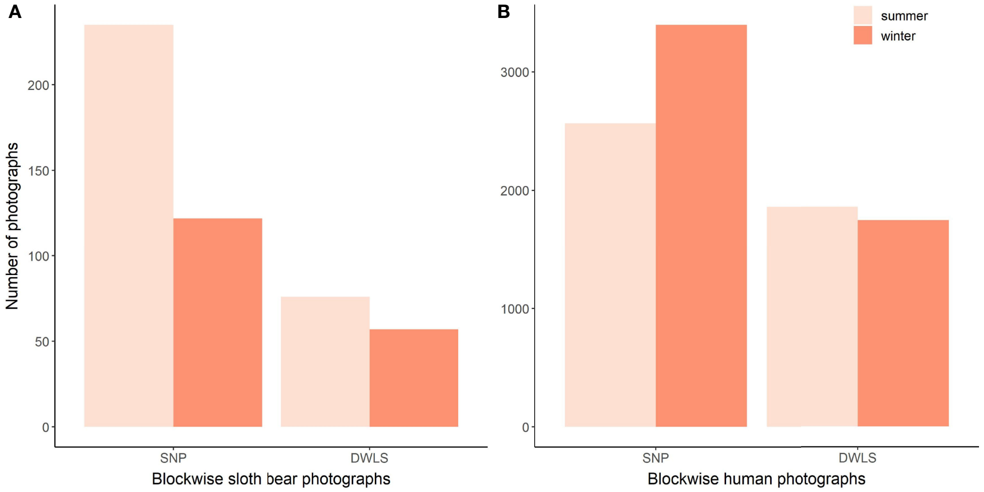

FIGURE 2 | Block-wise photographic captures of sloth bears **(A)** and humans **(B)** in winter and summer, obtained from camera trap surveys during 2017–2018 in the core zone of Sanjay Tiger Reserve, Madhya Pradesh, India.

TABLE 1 | Values of species interaction factor (SIF), probability of use (Ψ^{BA}) of sites by sloth bears (denoted as "B") in the presence of humans (denoted as "A"), probability of use (Ψ^{Ba}) of sites by sloth bears in the absence of humans (denoted as "a") and detection SIF (delta, δ), along with associated standard error (SE), calculated for two seasons (winter and summer) in the Sanjay Tiger Reserve, Madhya Pradesh, India, during 2017–2018.

Season	SIF ± SE	Ψ^{BA} ± SE	Ψ^{Ba} ± SE	δ ± SE
Overall winter (2017 & 2018)	0.98 ± 0.007	0.71 ± 0.04	1.00 ± 0.00	1.05 ± 0.06
Overall summer (2017 & 2018)	1.00 ± 0.03	0.79 ± 0.04	0.77 ± 0.17	1.06 ± 0.07

TABLE 2 | Spatial overlap index values of species pair (sloth bear and human), with associated 95% confidence interval (CI), during 2017–2018 in the Sanjay Tiger Reserve, Madhya Pradesh, India.

Season	Area	Spatial overlap between sloth bear and human with 95% CI
Winter 2017	SNP	0.41 (0.21–0.68)
Summer 2017	DWLS	0.17 (0.10–0.33)
Winter 2018	DWLS	0.51 (0.24–0.78)
Summer 2018	SNP	0.38 (0.18–0.62)
Overall winter (2017 & 2018)	SNP + DWLS	0.44 (0.27–0.65)
Overall summer (2017 & 2018)	SNP + DWLS	0.31 (0.16–0.49)

two-sample U^2 test revealed that sloth bear and human activity patterns significantly differed (Watson's $U^2 = 9.84$, $p < 0.001$ for winter; Watson's $U^2 = 11.13$, $p < 0.001$ during summer; **Table 4**) from each other for both the seasons. However, between seasons, the activity of sloth bears and humans also differed significantly ($U^2 = 0.22$, $p < 0.05$ for sloth bears; $U^2 = 19.61$, $p < 0.001$ for humans).

3.3 Sloth Bear–Human Conflict Prediction Through Geographic Profiling

3.3.1 Sloth Bear–Human Conflict Incidents

A total of 57 records of attacks on humans by sloth bears occurred in the core area of STR from 2009 to 2019. Centroids of compartments were generated for 57 conflict incidents.

3.3.2 Geographic Profiling for Conflict

The first model was run to produce the overall geographic profile of sloth bear–human encounters in STR using locations (n = 57) of bear attacks on humans and 88 camera trap locations that detected both sloth bears and humans. The top 10% of the probable source sites (n = 8) are mentioned in **Table 5** and **Figure 4**. The Gini coefficient of the model was 0.97. For the second model, the seasonal shift in the probable conflict sites was assessed using 81 and 66 camera trap locations for summer and winter, respectively. The model identified 9 sites and 5 sites as probable sources of conflict in summer and winter, respectively, details of which are provided in the supplementary material (**Supplementary Table 1**; **Supplementary Figures 1**, **2**). The Gini coefficient for the summer and winter models was 0.90 and 0.99, respectively.

TABLE 3 | Estimates of temporal overlap (Δ_4) between sloth bear and human, with associated 95% confidence interval (CI), during 2017–2018 in the Sanjay Tiger Reserve, Madhya Pradesh, India.

Season	Area	Temporal overlap (Δ_4) between sloth bear and human with 95% CI
Winter 2017	SNP	0.343 (0.282–0.404)
Summer 2017	DWLS	0.402 (0.323–0.483)
Winter 2018	DWLS	0.183 (0.110–0.262)*
Summer 2018	SNP	0.460 (0.405–0.515)
Overall winter (2017 & 2018)	SNP + DWLS	0.288 (0.238–0.337)
Overall summer (2017 & 2018)	SNP + DWLS	0.459 (0.414–0.505)

**due to less number of detections (< 75) of sloth bears, Δ_1 was calculated instead of Δ_4.*

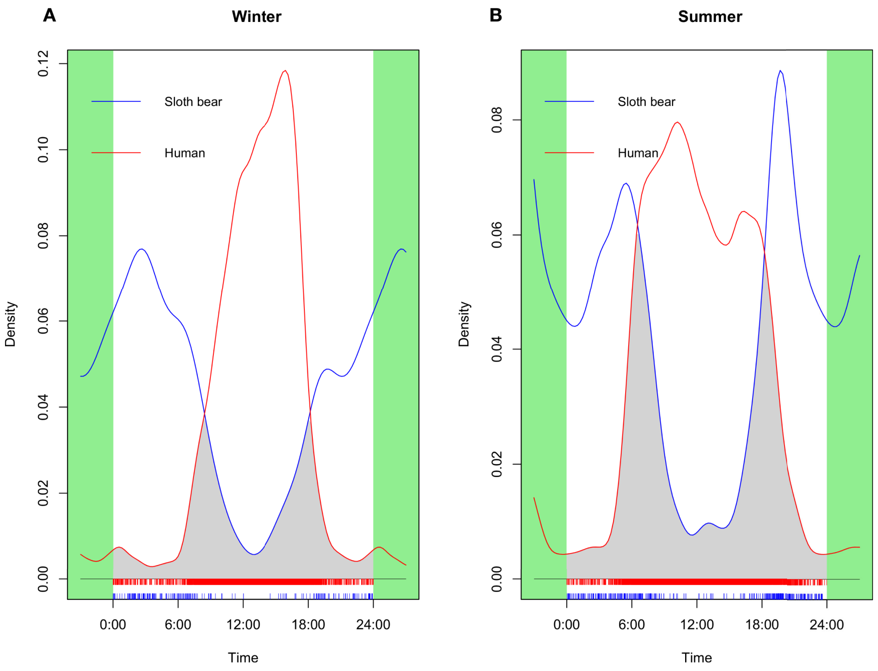

FIGURE 3 | Activity overlap and overlap coefficient (Δ_4), i.e., gray shaded area, between sloth bear (blue solid line) and human (red solid line) for overall winter **(A)** and summer **(B)** in the Sanjay Tiger Reserve, Madhya Pradesh, India, during 2017–2018.

A third model was run using 88 camera trap locations where sloth bears and humans were detected together, to test the efficacy of camera trap data for conflict prediction (**Figure 5**). A profile based on camera trap data alone produced a similar conflict prediction and could identify 21% of the sloth bear attacks on humans (12 attack sites out of 57) by searching less than 10% of the study area and 50% of these conflicts (29 attack sites out of 57) by searching just 20% of the study area (**Supplementary Table 2**). Thus, it validated the model performance (Gini coefficient = 0.91).

4 DISCUSSION

We reported spatial interaction and spatio-temporal overlap between sloth bears and humans with respect to seasons, i.e., winter and summer, in STR. Sloth bears independently co-occurred with humans in terms of space use, which was in contradiction to our expectations. However, distinct seasonal variation of spatial and temporal overlap between sloth bears and humans was observed at a finer scale, which, we believe, perhaps enables sloth bears to co-exist with humans in a human-dominated landscape. Finally, in the context of extensive human activities in sloth bear habitat and resultant conflict incidents, we identified the probable conflict source sites and hotspots in STR based on available conflict records and detections of sloth bears and humans from camera traps.

4.1 Spatial Interaction and Spatio-Temporal Overlap Between Sloth Bears and Humans

Two-species occupancy models showed that sloth bears could independently occur with humans irrespective of seasons, which, to our knowledge, is not an entirely novel situation. In Mudumalai Tiger Reserve, the occupancy of sloth bears was uninfluenced by the distance to the nearest human settlements

TABLE 4 | Circular statistics (include one-sample and multi-sample tests) of seasonal patterns of temporal activity of sloth bears and humans during 2017–2018 in the Sanjay Tiger Reserve, Madhya Pradesh, India.

Season	Variables	Sloth bear	Human
Winter	Mean vector	01:45	13:40
	95% confidence interval	12:48–02:42	13:35–13:46
	Watson's U^2 one-sample (p-value)	0.09 ($p < 0.05$)	2.25 ($p < 0.005$)
	Watson's U^2 two-sample (p-value)	9.84 ($p < 0.001$)	
Summer	Mean vector	00:18	12:09
	95% confidence interval	11:23–01:13	11:59–12:19
	Watson's U^2 one-sample (p-value)	0.51 ($p < 0.005$)	3.85 ($p < 0.005$)
	Watson's U^2 two-sample (p-value)	11.13 ($p < 0.001$)	

TABLE 5 | Hit score percentages of the probable source locations estimated from the Dirichlet Process Mixture (DPM) modeling using conflict (sloth bear attacks on humans from 2009 to 2019) and camera trap locations (where sloth bears and humans were both photo-captured) as incident sites in the Sanjay Tiger Reserve, Madhya Pradesh, India, during 2017–2018.

No. of source sites	Latitude	Longitude	Hit score %
1	23.98	81.68	0.00
2	24.03	81.58	0.05
3	23.87	82.09	0.17
4	23.86	81.85	0.25
5	23.96	82.04	0.54
6	23.95	81.81	1.06
7	23.92	81.60	3.93
8	23.88	81.95	6.57

(Ramesh et al., 2012a). Sloth bears can be habituated to persistent human presence and are attracted to human settlements in search of abundant seasonal fruits (*Ziziphus mauritiana*, *Syzygium cumini*), cultivated crops (groundnut, maize), and even garbage (food wastes) (Akhtar et al., 2004; Palei et al., 2020; Prajapati et al., 2021), which also increased the sloth bear–human conflict (Bargali et al., 2005; Prajapati et al., 2021). However, increased human activity or encounter with humans would facilitate the shifting of temporal activities of many mammalian species, including carnivores (Gaynor et al., 2018), instead of spatial avoidance (Nickel et al., 2020), or could induce habituation towards human activity (Albert and Bowyer, 1991; Barnes and Wilker, 2000; Elfström et al., 2014). On the other hand, bears show strong avoidance of human activities both spatially and temporally (Chi and Gilbert, 1999; Tollefson et al., 2005; Ladle et al., 2018). We believe the spatial distribution of humans or anthropogenic activities could be the governing factor of such spatial avoidance or spatial co-occurrence. In the human-dominated landscape, it is not always possible to avoid humans spatially as it would cost to miss out on potential resource-rich areas, which would negatively affect the foraging success and overall fitness of the animal. In STR, human occurrence and livestock grazing in sloth bear habitat are extensive. We found high occupancy ($\Psi^{A} = 0.96 \pm 0.02$ in winter; $\Psi^{A} = 0.84 \pm 0.03$ in summer) of humans in STR, which indicated the substantial level of anthropogenic pressure throughout the study area, regardless of seasons. Despite this extensive human pressure, the probability of use of sites by sloth bears was also found to be high ($\Psi^{BA} = 0.71 \pm 0.04$ in winter; $\Psi^{BA} = 0.79 \pm 0.04$ in summer; **Table 2**), which indicated that irrespective of seasons, sloth bears are utilizing sites independently of humans. It was also found that detection SIF (delta, $\boldsymbol{\delta}$) was slightly higher than 1 ($\boldsymbol{\delta} = 1.05 \pm 0.06$ in winter; $\boldsymbol{\delta} = 1.06 \pm 0.07$ in summer; **Table 2**), indicating some sort of association in detection between sloth bears and humans. It could be attributed to the relatively high preference for trail use by wide-ranging carnivores (Harmsen et al., 2010; Karanth et al., 2011; Kays et al., 2017; Nickel et al., 2020), apart from humans, which would facilitate animals to cover a larger distance in an energetically efficient way (Dickie et al., 2017).

However, spatial overlap estimation based on Pianka's overlap index (Pianka, 1974) revealed seasonal variation between sloth bears and humans. The intensity of space use by sloth bears was lower (0.31) during summer than during winter (0.44), which indicated that bears used sites less frequently during summer in the presence of humans than in winter. Given the widespread distribution of human activities during winter and summer, it was expected to have more overlapped areas between the two without any seasonal variation. Less spatial overlap between sloth bears and humans during the summer season could be attributed to comparatively higher temporal overlap during summer with respect to winter (see **Table 4**). In STR, human activities were found to be primarily diurnal regardless of seasons. However, during summer, the activities of humans started well before 06:00 o'clock in the morning and continued till late evening hours, which substantially overlapped with the crepuscular activity of sloth bears (Yoganand, 2005; Bargali et al., 2012; Ramesh et al., 2013). Increased human activity in the early morning and crepuscular hours during summer (especially from March to the first half of April) is primarily associated with the collection of flowers of Mahua, which is used to prepare alcoholic beverages (Kumar and Rao, 2007; Mewada, 2012). The selling of dry Mahua flowers

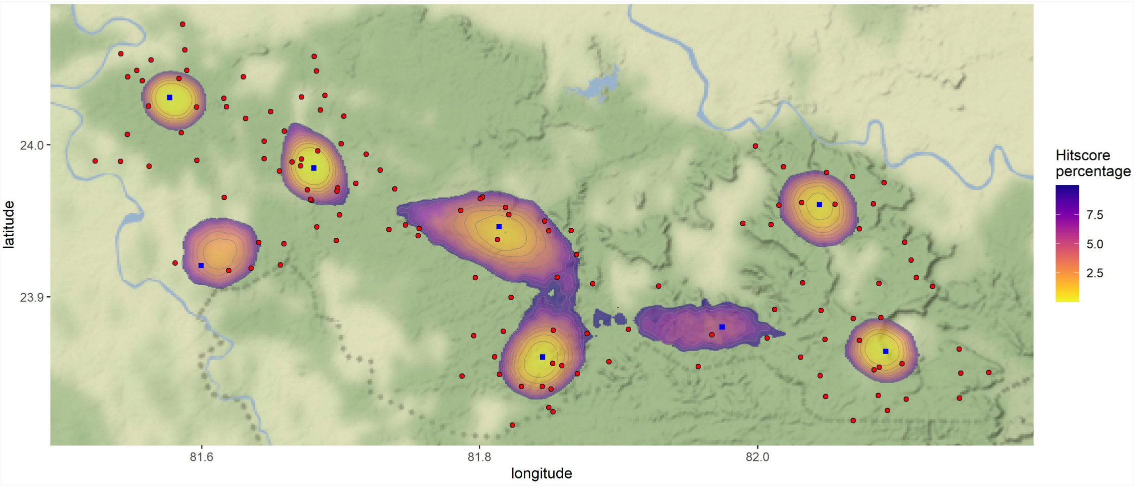

FIGURE 4 | Map showing the top 10% of the probable sloth bear–human conflict areas (probable source sites and dispersion or hotspots around source sites) in the Sanjay Tiger Reserve, Madhya Pradesh, India, based on locations of sloth bear attacks on humans (from 2009 to 2019) and camera trap surveys (locations where sloth bears and humans were both photo-captured) conducted during 2017–2018; blue square points (n = 8) were probable source locations (or anchor points) of conflict; red points were the camera trap locations which photo-captured both sloth bears and humans; lighter color shade indicated high conflict probabilities.

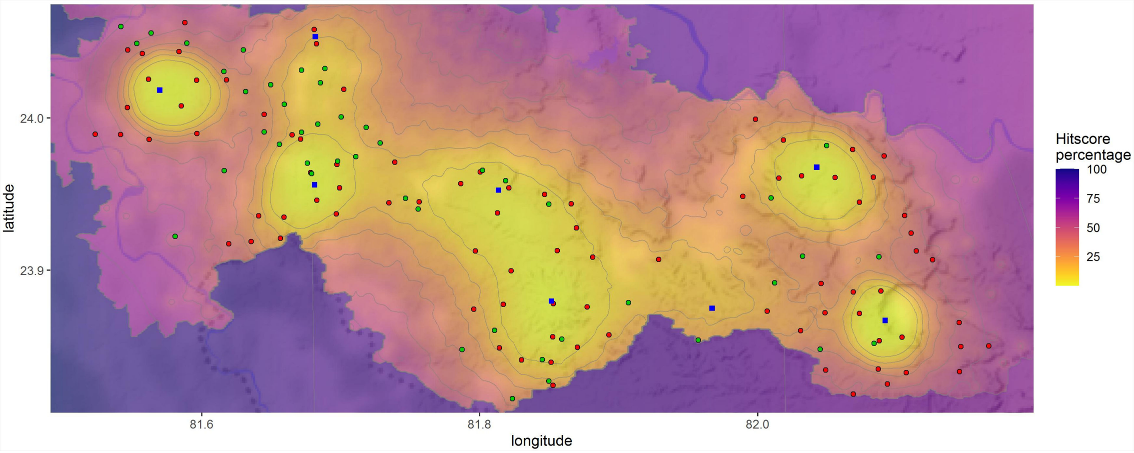

FIGURE 5 | Prediction of the source-dispersion of sloth bear–human conflict using camera trap surveys conducted in the Sanjay Tiger Reserve during 2017–2018, Madhya Pradesh, India; the lighter shade areas indicate high conflict probable areas and the blue squared points were the probable source sites (n = 8) of conflict hotspots; light green points were the conflict sites where attacks on humans by sloth bears occurred between 2009 and 2019; red points were the camera trap locations which photo-captured both sloth bears and humans; twenty-nine conflict sites fall within <20% of the search area and 12 conflict sites within <10%, thus validating the model performance.

significantly helps in the income generation of indigenous people (Islam and Quli, 2017). Hence, it is widely considered an important livelihood option for indigenous people in India (Kumar et al., 2018; Mishra and Poonia, 2019). The Mahua flowers also significantly contributed to the diet of sloth bears during the onset of summer (Bargali et al., 2004; Yoganand, 2005), which, coupled with its crepuscular activity, increased the interaction between sloth bears and humans (Akhtar et al., 2004). In STR, based on our camera trap survey, it was found that the frequency of occurrence of sloth bears and humans at per hour intervals was high during the early morning (05:00–07:00) and evening (19:00–20:00) in summer (**Supplementary Figure 3**). To avoid human encounters during the crepuscular period, it is possible that sloth bears used sites less intensively where chances of human encounters were high. Contrastingly, during winter, high spatial overlap could be associated with low temporal overlap between sloth bears and humans. In winter, human activities remained more concentrated during broad daylight hours for a comparatively short span of time, but did not start in the early morning hours, like summer (see **Figure 3**). In line with this, the morning (07:00–08:00) and early evening hours (15:00–16:00) showed the maximum frequency of occurrence of sloth bears and humans per hour interval, although the magnitude of the frequency of occurrence was less in winter compared to summer (**Supplementary Figure 3**).

We found that the activity patterns of humans varied significantly between seasons, which was also the case for sloth bears. Our results indicated that sloth bears used sites more extensively during winter regardless of human occurrences, with a distinct temporal avoidance. Studies elsewhere have shown that carnivores could avoid human occurrences temporally (Tollefson et al., 2005), instead of avoiding them spatially, in a human-dominated landscape (Wang et al., 2015; Nickel et al., 2020). The marginal difference ($p < 0.05$) in the activity of sloth bears between seasons indicated that sloth bears tended to avoid a more predictable pattern of human activity, despite already being strictly nocturnal and crepuscular in a human-dominated landscape (Bargali et al., 2012). Since human activity is not dependent on sloth bear activity, our study revealed that a significant shift in human activity between seasons influenced the likelihood of interaction between sloth bears and humans temporally and spatially.

However, despite distinct spatio-temporal avoidance, the conflict between sloth bears and humans in terms of human-mauling is prevalent in STR. It is attributed to the year-round existing immense anthropogenic pressure posed by human settlements inside the core area of STR. Similar observations were also made in the human-dominated landscapes of central India (Bargali et al., 2005; Dhamorikar et al., 2017) and elsewhere (Garcia et al., 2016; Debata et al., 2017; Singh et al., 2018). Based on our camera trap survey, we found that during summer, the number of detections of sloth bears and humans per hour interval was higher ($n = 53$) than during winter ($n = 33$), indicating that during summer, there is a relatively high probability of conflict incidents inside the forest, as reported by Dhamorikar et al. (2017). We assumed that sites where sloth bears and humans were detected more frequently within a specific bin (1 h in this case) of time are more prone to accidental encounters between them, while humans are engaged in different activities, especially in and around those sites. In previous studies, it has also been observed that most of the attacks occur during sudden close encounters with sloth bears (Dhamorikar et al., 2017; Singh et al., 2018), especially while collecting non-timber forest produce (NTFP) separately and silently inside the forest (Dhamorikar et al., 2017).

4.2 Identification of Existing and Future Conflict Hotspots

Finally, we have made an attempt to identify the conflict source sites and dispersion around source sites based on the present camera trap survey and records of conflict incidents obtained from the forest department. Our primary aim was to identify areas inside the forest where humans and sloth bears occurred more frequently, assuming that areas where sloth bears and humans were more likely to get exposed to each other, lead to conflict scenarios. Furthermore, records from the forest department were used to validate our model and future conflict predictions. We found that, based on detections of sloth bears and humans obtained from camera traps and validating them with registered conflict records, the GP model produced a reasonably good prediction (Gini coefficient = 0.92) of conflict source and dispersion. The GP model was found to be especially robust when fewer data points are provided and can potentially outcompete any other spatial statistics used in different contexts and scenarios of model-based predictions (Stevenson et al., 2012). However, a distinct seasonal variation was observed in the current prediction of conflict hotspots. More probable source sites ($n = 9$) of conflict were identified during summer, in comparison to winter ($n = 5$), indicating a greater chance of conflict in forest areas during summer, which was also earlier reported in the central Indian landscape (Dhamorikar et al., 2017). It could be attributed to the much higher number of detections of sloth bears ($n = 312$) during summer, in comparison to winter ($n = 180$, see **Table 1**), and associated high temporal overlap with humans. We found that seasonal variation in the temporal overlap between sloth bears and humans was also reflected in the predicted source-dispersion modeling of conflict in STR. As probable source sites and hotspots were mostly associated with villages' locations, it was evident that the probability of conflict was also higher in areas adjacent to villages. Relocation of villages from the core zone of STR, especially those situated around the probable source sites shown here, would be a primary step for managers to reduce conflict between sloth bears and humans. However, despite being a contentious and gradual process, voluntary relocation of villages from the core zone of Tiger Reserves was proved to be one of the successful means to restore wildlife population and its habitat (Lasgorceix and Kothari, 2009), with special reference to tiger conservation in India (Jhala et al., 2021). Also, elsewhere in India, communities living inside Tiger Reserves prefer to be relocated willingly due to the decreased productivity of forests, lack of basic amenities, and human–wildlife conflict (Harihar et al., 2014). Thus, we believe that the findings of geographic profiling could be one means for forest managers to identify the villages that need to be relocated on a priority basis or rationalize resources or techniques for conflict mitigation. However, in between the process of relocation, an effort could be made to make people living in conflict hotspots aware of the avoidance of conflict scenarios based on the findings of this study and studies carried out elsewhere in central India (Bargali et al., 2005; Dhamorikar et al., 2017; Dhamorikar et al., 2018). Although we could not identify the conflict sources and hotspots outside the core zone of STR, conducting awareness programs among the local people as mitigation measures is especially important in the buffer zone and other human-dominated forest areas outside the Tiger Reserve.

4.3 Caveats and Limitations

Our study had several limitations. Due to logistical constraints, we could not broaden our understanding of conflict prediction in the buffer zone of STR. Also, camera traps were not deployed in the monsoon, which restricted us to drawing inferences during this season. We have deployed camera traps on the trails and forest roads in order to maximize the capture of tiger and other co-predators, including sloth bears. Previous studies showed that placement of camera traps on trails or forest roads significantly increased the detection of certain mammal species and bears as well (Wearn et al., 2013; Kolowski and Forrester, 2017; Tanwar et al., 2021). On the other hand, human activities inside the forest, such as the collection of NTFPs, are not restricted to the roads and trails but can be conducted anywhere in the forest depending upon resource availability and accessibility. In general, people do not walk alone on the forest roads or trails, especially during dark hours, as moving in groups would reduce the chance of being attacked by bears (Garrote et al., 2017). Despite knowing these facts, we assumed that in the locations where humans and bears were frequently detected, the chance of conflict could also be high in and around those locations. Although widely used, the efficacy of trail-based camera traps in estimating anthropogenic pressure has rarely been assessed. However, Miller et al. (2017) showed how wildlife monitoring cameras (on-trail) can also efficiently capture human activities and could be used for long-term wildlife–human interaction-based studies. Our study indicated fairly accurate conflict prediction through the GP model by using only camera trap data, although we recommend careful considerations while deploying camera traps (on-road and off-road) in future similar studies.

4.4 Conclusion

This study revealed that the interaction between sloth bears and humans was primarily governed by the seasonal variation of spatio-temporal overlap, especially the significant changes in human activity between seasons. From the species perspective, fine scale segregation of temporal and spatial activity was observed to co-exist with humans. However, extensive anthropogenic pressure throughout STR exacerbated the situation and led to conflict scenarios. Therefore, radio-telemetry-based research, along with monitoring through camera traps, is recommended to understand the spatio-temporal activity of sloth bears in a human-dominated landscape. In this study, the application of novel methods like GP models provided an opportunity to predict the source-dispersion of conflict precisely, even using secondary data sets obtained from camera traps. It would be insightful for forest managers to reduce conflict incidents derived from sloth bear–human interface with a more area-specific targeted approach and promote coexistence in a human-dominated landscape.

DATA AVAILABILITY STATEMENT

The datasets presented in this study can be found in online repositories. The names of the repository/repositories and accession number(s) can be found in the article/**Supplementary Material**.

AUTHOR CONTRIBUTIONS

SK and RK conceived the project. RK, SK, SS, and SC designed the study. SC and RR collected the data. SC and RR carried out data preparation prior to analysis. SC and MB completed data analysis, prepared tables and figures and wrote the first draft of the manuscript. SS, RK, and SK supervised and critically reviewed the manuscript. All authors listed have made a substantial, direct, and intellectual contribution to the work and approved it for publication.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fcosc.2022.850309/full#supplementary-material

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N-mixture model-based estimate of relative abundance of sloth bear (*Melursus ursinus*) in response to biotic and abiotic factors in a human-dominated landscape of central India

Abstract

Reliable estimation of abundance is a prerequisite for a species' conservation planning in human-dominated landscapes, especially if the species is elusive and involved in conflicts. The importance of camera traps is globally recognized as a means of population estimation, although estimating the abundance of unmarked, cryptic species has always been a challenge to conservation biologists. This study explores the use of the N-mixture model with three probability distributions, i.e. Poisson, negative binomial (NB) and zero-inflated Poisson (ZIP), to estimate the relative abundance of sloth bears (*Melursus ursinus*) based on a camera trapping exercise in Sanjay Tiger Reserve, Madhya Pradesh from December 2016 to April 2017. We used environmental and anthropogenic covariates to model the variation in the abundance of sloth bears. We also compared null model estimates (mean site abundance) obtained from the N-mixture model to those of the Royle-Nichols abundance-induced heterogeneity model (RN model) to assess the application of similar site-structured models. Models with Poisson distributions produced ecologically realistic and more precise estimates of mean site abundance (λ=2.60±0.64) compared with other distributions, despite the relatively high Akaike Information Criterion value. Area of mixed and sal forest, the photographic capture rate of humans and distance to the nearest village predicted a higher relative abundance of sloth bears. Mean site abundance estimates of sloth bears obtained from the N-mixture model (Poisson distribution) and the RN model were comparable, indicating the overall utility of these models in this field. However, density estimates of sloth bears based on spatially explicit methods are essential for evaluating the efficacy of the relatively more cost-effective N-mixture model. Compared to commonly used index/encounter-based methods, the N-mixture model equipped with knowledge on governing biotic and abiotic factors provides better relative abundance estimates for a species like the sloth bear. In the absence of absolute abundance estimates, the present study could be insightful for the long-term conservation and management of sloth bears.

Introduction

Knowledge of the abundance of a species is fundamental for ecological studies and a prerequisite for successful conservation planning and decision-making processes in the field of conservation biology (Kremen, Merenlender & Murphy, 1994; Gaston & Rodrigues, 2003; Nichols & Williams, 2006; McCarthy & Possingham, 2007). Reliable abundance estimates can also be crucial for identifying the biotic and abiotic factors that govern the population size (Meents et al., 1983). Inaccurate estimates or estimates based on empirical studies can lead to erroneous management decisions and undermine the effort to conserve the concerned species in its natural environment. However, estimating the abundance of cryptic species through the traditional capture-mark-recapture (CMR) method is labor-intensive and not always logistically feasible. Exploration and subsequent implementation of cost-effective methodologies to monitor population abundance are thus required, given the limited resource availability in the field of wildlife conservation (Parker et al., 2011).

With the advent of science and technology, passive detectors, especially automated camera traps, have become indispensable in wildlife and conservation biology (O'Connell, Nichols & Karanth, 2011; Burton et al., 2015). However, despite the diverse applications of camera traps, their use has been confined to population estimation of individually identifiable or marked animals (Burton et al., 2015). Camera trap surveys provided successful population density estimates through spatial capture-recapture (SCR) models for marked animals (Karanth & Nichols, 1998; O'Connell et al., 2011). However, population estimation for unmarked or unidentifiable animals via camera trapping has posed challenges to ecologists and conservation biologists (Gilbert et al., 2021). Also, for unmarked species like Asian bears, population monitoring in a small area (<10,000km^2) requires spatially explicit methods (e.g., camera traps or genetics). However, these studies remain restricted due to resource and logistical constraints (Proctor et al., 2022).

Photographic capture rates (number of individuals photo-captured/effort in terms of trap-nights) or relative abundance indices provide surrogates of population abundance (Carbone et al., 2001; O'Brien, Kinnaird & Wibisono, 2003; Bengsen et al., 2011). However, inferences drawn from these index-based surveys do not meet the implicit assumption of equal detection probability (Jennelle, Runge & MacKenzie, 2002; Harmsen et al., 2010; Sollmann et al., 2013; Burton et al., 2015). Rowcliffe et al. (2008) proposed the random encounter model (REM) to estimate density by incorporating information on animal movement, photographic capture rate and the detection zone of camera traps. Despite its unique and robust approach, the limited application of REM is due to the actual random deployment of camera traps and the inability to account for spatial variation of abundance. Also, the need to extrapolate density from the camera trap detection zone to the entire surveyed area, and the lack of knowledge regarding the movement of elusive species, further restrict its applicability (Gilbert et al., 2021). The application of SCR frameworks for unmarked individuals, otherwise known as unmarked spatial capture-recapture (USCR; Chandler & Royle, 2013), is another promising approach to estimating population density. However, its limited applicability is due to the underlying computationally intensive (Bayesian) framework (Royle et al., 2014; Gilbert et al., 2021), tendency to produce imprecise estimates (Augustine et al., 2019) and judicious selection of priors on the spatial scale of animal detection at camera traps (Sun, Fuller & Royle, 2014). On the other hand, site-structured models such as the Royle-Nichols abundance-induced heterogeneity model (RN model; Royle & Nichols, 2003) and the N-mixture model (Royle, 2004) have been widely used to draw inferences on abundance from detection-nondetection data and count data, respectively, based on spatially or temporally replicated surveys. In the recent past, the latter was more widely applied to estimate the abundance of terrestrial mammals from count data (Keever et al., 2017; Xiao et al., 2018; Kafley et al., 2019; Kidwai et al., 2019; Penjor et al., 2019; Searle et al., 2020). The major advantages of the N-mixture model are its cost-effectiveness, the ability to bypass individual identification, and the ability to account for spatial variation of abundance as a function of covariates. The N-mixture model typically requires the following assumptions: population closure at the site, equal probability of detection of all individuals at a site, no false-positive errors (i.e., misidentification or double-counting of individuals), and independent detection of individuals across the sampling units (Royle, 2004). However, N-mixture models are sensitive to assumption violations, especially false-positive errors, which are not commonly addressed (Kéry & Schaub, 2011). For free-ranging unmarked species, one individual would likely be photo-captured several times during the entire sampling occasion (O'Brien, Kinnaird & Wibisono, 2003), which can overestimate abundance (Link et al., 2018; Nakashima, 2020). However, relaxing the assumption of population closure at the site for free-ranging species, it is justifiable to interpret the true abundance of the site as "relative abundance," which translates to the number of individuals utilizing the area at a given time (Kéry & Royle, 2015).

The sloth bear (*Melursus ursinus*), one of the four bear species found in the Indian subcontinent, is mainly confined to isolated patches of forested habitat both inside and outside of the protected areas (PAs) (Dharaiya, Bargali & Sharp, 2016). Despite the species' broad geographic distribution, especially across the moist and dry deciduous forests of the Western Ghats and central Indian landscape (Yoganand et al., 2006), information on population abundance based on a rigorous statistical framework is lacking (Dharaiya, Bargali & Sharp, 2016). However, the range-wide population abundance of sloth bears reportedly ranges between 10,000 and 20,000 (Garshelis et al., 1999; Chauhan, 2006; Yoganand et al., 2006; Sathyakumar et al., 2012). None of these estimates was reliable enough to monitor the population trends because of the differential data collection methods and types of data used in the analyses (Sathyakumar et al., 2012; Dharaiya, Bargali & Sharp, 2016). The sloth bear is an omnivorous generalist, long-ranging ursid and often comes in conflict with humans (Rajpurohit & Krausman, 2000; Bargali, Akhtar & Chauhan, 2005; Garcia, Joshi & Dharaiya, 2016; Debata et al., 2017; Dhamorikar et al., 2017; Singh, Sonone & Dharaiya, 2018). The ever-increasing human population, subsequent habitat fragmentation and human-sloth bear conflicts are significant concerns for conserving this species. One recent study (Gomez et al., 2021) also indicated a steadily increasing trend of sloth bear mortality due to poaching. So far, only one study (Garshelis, Joshi & Smith, 1999) has attempted to make robust density estimates of sloth bears, done in Royal Chitwan National Park (henceforth Chitwan), Nepal, based on a spatial mark-resight method. Reliable information on population abundance and identifying factors influencing abundance are crucial for the long-term conservation of sloth bears in human-dominated landscapes.

In this context, we explored the application of site-structured models with special reference to the N-mixture model (Royle, 2004) to quantify the relative abundance of sloth bears in Sanjay Tiger Reserve, Madhya Pradesh. The objectives of the present study were as follows:
(1) To demonstrate the use of the N-mixture model in estimating the relative abundance of sloth bears and compare the mean site abundance estimates obtained from the N-mixture model to

those of the RN model, and (2) to identify the environmental and anthropogenic covariates that govern the relative abundance of sloth bears.

Materials & Methods

Research Permissions and ethical considerations

Madhya Pradesh Forest Department (MPFD) issued all required permissions for our field surveys (Letter No: I/3129 dated 29/05/2014). Due to the non-invasive nature of sampling, we did not require ethical clearance for this study.

Study site

Sanjay Tiger Reserve (STR) is situated between 24.125122°N 81.546009°E (northwestern boundary) and 23.822641°N 82.175139°E (southeastern boundary) in the state of Madhya Pradesh in central India (Fig. 1). STR falls under the category "Central Highlands" as per the biogeographic classification of India (Rodgers & Panwar, 1988). The core and buffer areas of STR encompass 831.25 km^2 and 812.58 km^2, respectively. Furthermore, the core area of STR consists of two administrative parts, i.e. Sanjay National Park (SNP; 466.60 km^2) and Dubri Wildlife Sanctuary (DWLS; 364.60 km^2).

The terrain of STR is rugged and undulating, with plateaus and gorges in the eastern part (SNP) and flat land in the western part (DWLS). STR has an elevation range between 425m and 732m. The broad forest type of STR falls under sub-group 3C- "North Indian moist deciduous forests", with subtype (C2) "Moist sal-bearing forest" with subdivision (2e) "Moist peninsular sal forest" (Champion & Seth, 1968). Sal (*Shorea robusta*) is dominant in SNP and DWLS, occupying about 80% of the entire area, followed by Mixed, Bamboo mixed, Scrubland and Grassland (Fig. S1). Apart from sal, other frequently found tree species include tendu (*Diospyros melanoxylon*), mahua (*Madhuca longifolia*), char (*Buchanania cochinchinensis*) and sedha (*Lagerstroemia parviflora*). STR receives a mean annual precipitation of 1303mm between June and September, and the annual temperature varies between 7.4°C and 41.8°C. Other than the sloth bear, tiger (*Panthera tigris*), leopard (*Panthera pardus*), striped hyena (*Hyaena hyaena*), Indian wolf (*Canis lupus*) and Asiatic wild dog (*Cuon alpinus*) are the major carnivores found in STR.

Anthropogenic pressure is immense inside the core area of STR due to the presence of thirty-nine villages.

Camera trap survey

The camera trapping survey was conducted from December 2016 to April 2017 in the core area of STR. We divided the study area into 2km × 2km grids (henceforth, sites) following the All India Tiger Estimation protocol (Jhala, Qureshi & Gopal, 2015) and subsequently deployed camera traps in 143 locations (one at each site). We divided the core area into two blocks (SNP and DWLS), and camera traps were deployed in one block at a time due to the limited availability of camera traps. A pair of motion-triggered camera traps (Cuddeback C1 and Cuddeback Ambush) were deployed at each site opposite to one another to increase the detection (Pease, Nielsen & Holzmueller, 2016; O'Connor et al., 2017) at an approximate height of 30-40cm above the ground. We placed camera traps on forest roads, animal trails and dry riverbeds to maximize the photo-capture of sloth bears and other large carnivores (Karanth et al., 2011). The average distance between two adjacent camera trap locations was 2km. We set 15 second time-lapse between two consecutive photographs. Cameras were active 24 hours a day for an average trapping period of 42 days (range: 13-45 days) and checked once every 20-25 days to change batteries and memory cards. We collected the locations of camera traps by using Garmin e-Trex handheld GPS units (Garmin Inc.).

To avoid pseudoreplication, we considered two consecutive photographs of sloth bears or humans captured ≥30 minutes apart as independent events (O'Brien, Kinnaird & Wibisono, 2003; Xiao et al., 2018; Kafley et al., 2019). We set the temporal sampling unit (sampling occasion) as three days. We pooled the count of independent photographs of sloth bears from each site for each unit of three consecutive days, which yielded an encounter history of 15 sampling occasions. Reducing the sampling period by pooling data from a specific time (days) helped reduce overdispersion (Penjor et al., 2019).

Covariates

Local-scale spatial covariates

At each camera trap location, we measured habitat features within a 10m-radius plot, with a focus on fruiting trees (≥2m in height and ≥1m girth at breast height), which comprise≥1% of the diet of sloth bears (Ramesh et al., 2012; Rather et al., 2020). We also recorded the number of termite mounds and woody shrub species at every 10m and 5m-radius plot, respectively. We calculated the photo-capture rate of humans (number of photographs of humans at each location/number of days each camera trap was active) as one of the surrogates of anthropogenic pressure at each site. Table 1 provides the details of local-scale covariates.

GIS-based covariates

We prepared the Land Use Land Cover (LULC) layer for STR by processing a 30m spatial resolution Landsat-8 image (Image ID: LC81430432016297LGN00) dated 23 October 2016 in ERDAS IMAGINE 9.2 (Leica Geosystems) following a standard hybrid image classification protocol (Ruppert et al., 1997). The image classification resulted in six classes, i.e. dense forest, open forest, scrubland, agricultural land, barren land, and water bodies. The average accuracy of the resultant classified image was 88.89%, and the overall Kappa coefficient was 0.85. To classify the forest types in STR, we processed a 30m spatial resolution Landsat-8 image (Image ID: LC81430432018110LGN00) dated 2 May 2018, following the methodology mentioned above and with the support of 412 ground points. The classes included sal mixed, mixed, sal forest, bamboo mixed, grassland and non-forest. We categorized the different forest types following the broad classification by Champion & Seth (1968). Subsequently, we performed the accuracy assessment of the resultant classified image based on 392 ground points collected throughout the study area. This classified image's average accuracy and overall Kappa coefficient were 64.30% and 0.45, respectively.

Other covariates used in the analyses were the terrain ruggedness index (RI; Riley, 1999) and the distances to the nearest water source, metal road and village (see Table 1 for details). All analyses were carried out in ERDAS IMAGINE 9.2 (Leica Geosystems) and ArcGIS 10.2 (Environmental Systems Research Institute).

Relative abundance model of sloth bear

We developed the N-mixture model (Royle, 2004; Kéry & Royle, 2015) to estimate the relative abundance of sloth bears in STR within 3.52km × 3.52km grids, as this represents the minimum home range (i.e., 12.40 km^2) of sloth bear in central India (Yoganand, 2005). However, little is known about the home range size of the sloth bear in a human-dominated landscape, though studies elsewhere (Joshi, Garshelis & Smith, 1995; Ratnayeke, Manen & Padmalal, 2007) have indicated much smaller average home range estimates compared to those of the central Indian landscape (Yoganand, 2005). Since we deployed camera traps within smaller grids (2km × 2km), we used the data from all 1-4 camera traps falling inside each 3.52km × 3.52km grid and considered these as spatial replicates, following Xiao et al. (2018). We used the N-mixture model to estimate the relative abundance of sloth bears from independent photographic captures as a function of environmental and anthropogenic covariates (Table 1).

N-mixture models assume population closure (i.e., the number of individuals residing within the site is constant as no emigration or immigration occurs between sites). Due to the individually unidentifiable aspect and long-ranging behavior of the sloth bear, we believe that the closure assumption was likely to be violated. Hence, we relaxed the closure assumption by changing the interpretation from absolute abundance at a site to the number of individuals ever associated with a site during a given period (Kéry & Royle, 2015), otherwise known as relative abundance. We pooled independent detections of sloth bear for each site "i" from "R" sites where, i=1, 2, 3,..., R during a sampling occasion of "T", where t=1, 2, 3,…, T. The observed count (y_{it}) at "i" sites during "t" sampling occasions followed a binomial distribution, and the site abundance N_i followed a Poisson distribution (Kéry & Royle, 2015).

Due to the greater flexibility in modeling abundance with covariates and the ability to incorporate other effects (e.g. zero inflation, latent state), Poisson distribution is an integral part of the state process, i.e., abundance (Royle & Dorazio, 2008). On the other hand, binomial distribution accounts for imperfect detection or false-negative errors. The variation of observed counts (y_{it}) is an effect of imperfect detection of the actual (unknown) abundance N_i and its variability among sites. In simple algebraic form, we expressed the two processes as:

State process: $N_i \sim \text{Poisson}(\lambda_i)$; $\log(\lambda_i) = \beta_0$ + effects of covariates

Observation process: $y_{it} \mid N_i \sim \text{Binomial}(N_i, p_{it})$; $\text{logit}(p_{it}) = \alpha_0$ + effects of covariates

Where N_i is the latent abundance of site "i" (i=1,…., R), λ_i is the mean abundance of site "i", y_{it} is the count of the species at the site "i" and occasion "t" (t=1,…., T), and p_{it} is detection probability at the site "i" and occasion "t".

Studies have indicated that N-mixture models with Poisson and zero-inflated Poisson (ZIP) distributions can produce reliable relative abundance estimates if detection probability is modeled effectively with appropriate covariates (Barker et al., 2018; Kéry, 2018). We modeled the detection probability with the trapping period as we suspected that a more extended trapping period would positively influence the detection probability of sloth bears (O'Connor et al., 2017).

We also used local-scale spatial covariates (and no GIS-based covariates) likely to affect the detection probability, following Hofmeester et al. (2019). The detection probability would be constant if none of the covariates influenced it (Kafley et al., 2019).

After determining the effect of covariates on the detection process, we modeled the relative abundance of sloth bears as a function of environmental (forest types, LULC types, RI, distance to nearest water source, availability of fruiting trees, shrub density and number of termite mounds) and anthropogenic (human photographic capture rate, distance to nearest village and metal road) covariates with three probability distributions, i.e. Poisson, ZIP and negative binomial (NB). We calculated and extracted mean values of GIS-based environmental and anthropogenic covariates from each of the 12.40 km^2 grids. For local-scale spatial covariates where the number of camera trap locations was more than one, we calculated the average values of each covariate for the 12.40 km^2 grids. The selection of the most parsimonious models from the candidate model set followed the descending Akaike Information Criterion (AIC) values (Burnham & Anderson, 2002). We reported uninformative parameters but drew final inferences only from the models within ≤ 2 ΔAIC units (Burnham & Anderson, 2002) with statistically significant ($P<0.05$) parameters (Arnold, 2010). We did not consider model averaging in the case of a single top-ranked model or due to the presence of uninformative parameters in multiple top-ranked models (Arnold, 2010). However, due to the "good fit bad prediction dilemma" as mentioned by Kéry & Royle (2015), we further performed Goodness of Fit (GoF) tests with 1000 iterations of bootstrapping for each of the most parsimonious models of corresponding distributions. We also calculated overdispersion parameters ($\hat{c}$) for the same models (Kéry & Royle, 2015). We proceeded with further residual diagnostics and mapping of residuals if none of them passed the GoF test (when $P<0.05$) for the final checking of model adequacy (Kéry & Royle, 2015). We interpreted the result as an outcome of unstructured noise if no specific pattern of lack of fit was found (Kéry & Royle, 2015). Subsequently, we corrected it by inflating the predicted abundance's range (confidence interval) (Johnson, Laake & Ver Hoef, 2010; Kéry & Royle, 2015). We standardized all covariates using Z-transformation before analyses and subsequently performed Pearson's correlation tests to avoid multi-collinearity. A pairwise Pearson's correlation coefficient value of $r \geq \pm 0.7$ was considered strongly correlated (Dormann et al., 2013), and those covariates were not modeled together, as we were interested in seeing the effect of each covariate on the relative abundance of sloth bears. We carried out all analyses using the packages "Unmarked" (Fiske & Chandler, 2011) and "AICmodavg" (Mazerolle, 2015) implemented in R (R Core Team, 2019).

We compared the mean site abundance estimates with estimates obtained from another frequently used site-structured model, the RN model (Royle & Nichols, 2003), to understand the performance of the N-mixture model. Since modeling the relative abundance of the sloth bear with both local-scale and GIS-based covariates by applying the RN model and subsequent comparison of the model performance (of both N-mixture and RN models) was beyond the scope of this paper, we restricted the comparison between the null models of the two frameworks mentioned above. We chose the same probability distribution for the RN model, which was the most suitable for the N-mixture model. We used the program Presence (Version 2.13.11) (Hines, 2006) to estimate the mean site abundance of sloth bears by applying the RN model.

Results

Camera trap

A total of 191 and 5292 independent photographs of sloth bears and humans, respectively, were obtained from 5950 trap-nights (3870 and 2080 trap-nights in SNP and DWLS, respectively). Sloth bears were detected at 76 out of 143 (53.10%) camera trap locations (sites), and independent detections of sloth bear varied from 1 to 15 per site (Fig. 2). However, in the larger grids (12.40 km^2), the presence of sloth bears was recorded at 51 out of 73 (70.0%) grids. In SNP, a higher number (120) of sloth bear detections was observed compared to 71 detections in DWLS.

Detection process

The density of fruiting trees significantly influenced the detection probability of sloth bears for Poisson, NB and ZIP distributions (Table 2). The estimated detection probability (p) of the best detection model (modeled with the density of fruiting trees) was higher in Poisson (0.08 ± 0.01) than in ZIP (0.06 ± 0.02) and NB (0.01 ± 0.004) distributions (Table 2).

Relative abundance of sloth bears and effects of covariates

In top-ranked models, the relative abundance of sloth bears was significantly influenced by forest types (mixed and sal forest; environmental), distance to the nearest village (anthropogenic) and human photo-capture rate (anthropogenic) (Table 3 and Table 4). In addition to these four, the most parsimonious models also included the LULC types (agricultural land, scrubland and water body); however, none of these covariates significantly influenced the relative abundance of sloth bears (Table 4). Models with NB distributions were the most parsimonious in terms of lower overall AIC values, followed by ZIP and Poisson models (Table 3). Due to uninformative parameters, we did not carry out model averaging for any top-ranked models. Areas of mixed and sal forest predicted a higher relative abundance of sloth bears (Table 4 and Fig. 3; also see Fig. S2 and Fig. S3). The distance to the nearest village and human photo-capture rate also had significant positive effects (Table 4 and Fig. 3; also see Fig. S2 and Fig. S3) on sloth bear abundance.

However, the models within ≤ 2 ΔAIC units for Poisson, ZIP and NB distributions did not pass the GoF test. Overdispersion parameter estimates ($\hat{c}$) revealed a marginal lack of fit, which did not appear to differ among Poisson ($\hat{c}$ =1.31), NB ($\hat{c}$ = 1.19) and ZIP ($\hat{c}$ = 1.28) distributions. Residual diagnostics and mapping of residuals did not show any specific pattern of lack of fit. Hence, this finding is attributed to unstructured noise or overdispersion rather than a true lack of fit. The NB model (top-ranked) produced unusually high mean site abundance (λ = 10.80 ± 3.83) in comparison to mean site abundance estimates of ZIP (λ = 4.07 ± 1.56) and Poisson models (λ

= 2.60 ± 0.64) (Table 5). Spatial variation of relative abundance produced by the Poisson model was less (1 - 11; see also Fig. 4) than those of the ZIP (1 - 14) and NB models (4 - 37).

Comparing estimates with the RN model

The RN model (null model with Poisson distribution) produced a mean site abundance estimate of 1.62 ± 0.34, slightly lower than the N-mixture model's (2.05 ± 0.35) estimate. However, the mean site abundance estimate obtained from the N-mixture model was within the 95% confidence interval range (1.08 - 2.44) of the estimate produced by the RN model.

Discussion

Efficacy of the N-mixture model

We reported the first-ever estimates of the relative abundance of sloth bears using the N-mixture model. We also identified associated factors governing the relative abundance by using count data obtained from the camera trap survey in STR. The N-mixture model can produce reliable abundance (relative) estimates if the data meet the assumptions of the N-mixture model while accounting for imperfect detection (Royle, 2004). Also, its ability to model spatial variation of abundance and detection probability as a function of relevant covariates makes its inference more robust and reliable than inferences drawn from conventional index/encounter rate-based surveys (Gilbert et al., 2021). A growing number of studies have assessed the population abundance (relative and absolute) of unmarked individuals by using the N-mixture model from camera trap data (Brodie & Giordano, 2013; Keever et al., 2017; Xiao et al., 2018; Kafley et al., 2019; Penjor et al., 2019) as well as employing other field methods (Belant et al., 2016; Ficetola et al., 2018; Kidwai et al., 2019; Searle et al., 2020). Estimating the abundance of unmarked animals from camera trap data has always been challenging for ecologists. For the past two decades, very few studies have actually assessed the population abundance of Asian bears employing statistically rigorous methods; instead, many of the studies have heavily relied upon sign index or interview-based distributions due to the relatively less expensive nature and ease of application in the field (Garshelis et al., 2022; Proctor et al., 2022). Information on sloth bear abundance and its governing factors is lacking throughout the species' distributional range (Dharaiya, Bargali & Sharp, 2016). Most of the previous studies (Eisenberg & Lockhart, 1972; Laurie & Seidensticker, 1977; Santiapillai & Santiapillai, 1990; Ratnayeke, Manen & Padmalal, 2007; Akhtar, Bargali & Chauhan, 2008) regarding the abundance of sloth bears produced empirical estimates, except for Garshelis et al. (1999), where the authors first estimated abundance from sound analytical methods in Chitwan. However, in the case of this wide-ranging bear species, site-structured models (i.e., the RN model and the N-mixture model) may be reliable for estimating the population trend with careful considerations of model assumptions and study design (Morin et al., 2022).

It is well-advised that one should not interpret the abundance estimates obtained from site-structured models as a measure of absolute but relative abundance if the assumption of closure is likely to be violated (Barker et al., 2018; Link et al., 2018; Gilbert et al., 2021). In this context, mean abundance (lambda) estimated by the N-mixture model produced the number of individuals utilizing a site (12.40 km^2 grids in the present study) instead of the total number of individuals residing within the site. Site abundance estimates obtained from the N-mixture models would be consistently larger than the actual abundance of free-ranging animals like sloth bears, as each site contains multiple overlapping territories of different individual sloth bears. In reality, the actual abundance of animals comes from a considerably larger geographic extent than the actual surveyed area, leading to an overestimation of the mean site abundance and total predicted abundance (Joseph et al., 2009). In the present study, the best-selected model (with NB distribution) in terms of AIC produced substantially larger and ecologically unrealistic abundance estimates of sloth bears compared to ZIP and Poisson models. Mean abundance estimates obtained from NB and ZIP models showed that a large number (~11 and ~4 for NB and ZIP models, respectively) of bears were utilizing each site with considerably low detection probability, which, in general, is unlikely for a species like the sloth bear. In the Indian sub-continent, the lack of studies focusing specifically on the density or abundance of sloth bears and other bear species in a scientifically rigorous framework poses difficulties for comparing our findings to other studies. We expected that for a species like sloth bear, an elusive and solitary ursid, count data obtained from camera traps would be zero-inflated. The NB model performs poorly with a zero-inflated count dataset or when the focal species is not frequently found or is truly absent from a large number of sites surveyed (Joseph et al., 2009). In the present study, NB and ZIP variants of N-mixture models produced lower detection probabilities than Poisson models. Due to low detection probability, N-mixture models can produce positively biased abundance estimates with less precision than other methods (Dénes, Silveira & Beissinger, 2015; Dennis, Morgan & Ridout, 2015). On the other hand, the N-mixture model with a Poisson distribution can handle datasets with excessive true zeros (sites where the species is truly absent) given that the species naturally occurs in low densities (Joseph et al., 2009). In our study, the Poisson model produced a comparatively low mean site abundance (2.60±0.64) by modeling a substantial number (~30% of total sites surveyed) of unoccupied sites. We suggest interpreting the abundance estimates of sloth bears carefully with an ecologically realistic view rather than with strict adherence to the statistical properties of the models.

The mean site abundance of sloth bears produced from the N-mixture model (Poisson distribution) was comparable to estimates obtained from the RN model. Earlier studies have indicated that the RN model would not be a good choice when the focal species is common; in that case, the N-mixture model would be preferable (Dénes, Silveira & Beissinger, 2015; Kéry & Royle, 2015; Gilbert et al., 2021). The RN model reportedly produces imprecise estimates from camera trap-based surveys (covering comparatively small sampled areas), especially if the focal species is cryptic (Gupta et al., 2012; Kalle et al., 2014; Morin et al., 2022). However, there is evidence that the RN model can produce precise abundance estimates, enabling it to detect population trends (O'Brien et al., 2020). Estimates from the RN model can approximate the absolute density of the focal species if the study is conducted on a landscape scale in conjunction with an ecologically relevant grid size for camera trap deployment (Linden et al., 2017). On the other hand, simulation studies have shown that the N-mixture (Poisson and Binomial mixture) model may produce biased estimates of mean abundance in the context of false-positive detections (Nakashima, 2020). However, our findings from both models indicated the overall utility of site-structured models in estimating the relative abundance of elusive species like sloth bears. Although the number of camera trap locations/grid varied (1-4) in our study, the presence of >1 spatial replicates (camera trap locations) within most of the grids (>50%) might have improved the detection and robustness of mean site abundance estimates (Kolowski, Oley & McShea, 2021; Morin et al., 2022).

Applicability of other methods for unmarked population estimation

Due to the scarcity of spatially explicit studies, we could not directly compare relative abundance estimates of sloth bears with previous findings. Since the primary objective of this study was to understand the applicability of the N-mixture model, we considered other potential methods relevant to the abundance estimation of unmarked animals. However, we found that abundance estimation of unmarked species like Asian bears through camera traps rarely produces precise estimates for conservation planning unless supported by ancillary information obtained from statistically rigorous methods and covering a large number of sampling sites (Morin et al., 2022). Apart from the site-structured models presented here, USCR (i.e., the spatial count (SC) model; Chandler & Royle, 2013) and its extension (i.e., spatial presence-absence (SPA) model; Ramsey, Caley & Robley, 2015) can also be applied to estimate the absolute density of cryptic species. The application of the SPA model has been well-demonstrated to estimate the density of small carnivores through camera traps (Chatterjee, Nigam & Habib, 2020a,b). However, these models are computationally intensive (require Bayesian analysis) and sensitive to the appropriate choice of priors which could be challenging for a species like the sloth bear due to limited knowledge of its ranging pattern. Distance sampling through camera traps could also be a promising approach to estimating the density of unmarked animals (Howe et al., 2017), especially if the focal species is less abundant (Palencia et al., 2021). However, this method requires the random placement of camera traps which we could not implement, as this study followed the protocol of All India Tiger estimation (Jhala, Qureshi & Gopal, 2015). Random placement of camera traps might result in substantially low detections of sloth bears, which may not be conducive for further analysis. The N-mixture model would be particularly suitable regarding cost-effectiveness and logistic constraints related to camera placement. Moreover, if the study intends to model the spatial variation of abundance (relative or absolute), the N-mixture model would be more preferred than the other methods mentioned above. As long as the study objectives are clearly defined, and we correctly interpret the abundance estimates, the N-mixture model would be a suitable option for researchers and managers.

Effects of environmental and anthropogenic covariates on relative abundance

In the present study, mixed and sal forests positively influenced the relative abundance of sloth bears. In STR, we calculated the density of termite mounds and fruiting trees, utilizing a 10m-radius plot around each camera trap location. We recorded a high density (35/hectare) of termite mounds in mixed forest, followed by sal mixed and sal forest (27/hectare and 23/hectare, respectively). Similarly, the fruiting tree density was also higher (208/hectare) in the mixed forest in comparison to sal mixed (197/hectare) and sal forest (100/hectare). Due to fewer plots, we could not calculate meaningful density estimates of termite mounds and fruiting trees for the bamboo mixed forest and grasslands. In STR, a high number of fruiting trees and termite mounds in the mixed forest may explain the positive relationship between bear abundance and mixed forest. We found a similar observation from Panna National Park in central India, where the dense forest (comprised of miscellaneous tree species) had the highest densities of fruiting trees and prey insect colonies, and sloth bears preferred this habitat significantly (Yoganand, 2005). However, the limited number of surveyed plots (only at camera trap locations) to support our findings may not be enough to explain the effects of forest types and termite mounds on sloth bear abundance, for which more rigorous sampling is needed. In the North Bilaspur Forest Division, Akhtar et al. (2004) found that signs of sloth bears were more common in sal forest than in other forest types. We also found similar results in Chitwan but with distinct seasonal patterns (Garshelis, Joshi & Smith, 1999). The occurrence of bear signs (ground holes and mound holes) in sal forest was frequent in both landscapes, perhaps due to a relatively high number of termite mounds (Garshelis, Joshi & Smith, 1999; Akhtar, Bargali & Chauhan, 2004) and possibly the underground colonies of ants and termites. In the northern part of the Indian sub-continent, termite mounds are more abundant in sal forests (Chakraborty & Singh, 2020). However, underground colonies of termites and ants are difficult to quantify. Hence, we considered the number of termite mounds that are above-ground and visible as a surrogate for the availability of ants and termites (Akhtar, Bargali & Chauhan, 2004; Rather et al., 2020). We also could not find any significant relationship between the relative abundance of sloth bears and local-scale covariates; however, the availability of fruiting trees positively influenced the detection probability of sloth bears. In Nepal's Chitwan and Churia (a lowland forest outside the PAs) forests, the presence of fruiting trees and termite mounds positively influenced the sloth bear's occupancy and detection probability (Paudel et al., 2022; Pokharel et al., 2022). Das et al. (2014) also recorded a similar observation in southern India during their occupancy-based study. However, we believe that our camera trap-based study design may not be appropriate for drawing inferences on such local-scale covariates, despite their proven importance to the sloth bear's ecology. The ubiquitous nature of termites and ants in a relatively small area (Redford, 1987) may explain the insignificant relationship between bear abundance and termite mounds. Similarly, the availability of fruiting trees and termite mounds at a particular point (10m-radius plot, in our study) may not necessarily represent the entire sampling unit (12.40 km^2 grids). Thus, it could explain the apparent insignificant contribution of fruiting tree density and termite mounds to the relative abundance of sloth bears.

Sloth bears generally avoid human disturbances (Babu et al., 2015; Puri et al., 2015; Paudel et al., 2022; Pokharel et al., 2022). However, they can also tolerate a certain degree of human pressure or have habituated to living in disturbed and fragmented forests (Akhtar, Bargali &

Chauhan, 2004; Prajapati, Koli & Sundar, 2021). In the present study, we found that the photographic capture rate of humans and distance to the nearest village positively influenced the relative abundance of sloth bears. However, based on the current study design, it is not known whether a causal relationship between these variables (relative abundance of sloth bear and human photographic capture rate) exists. Moreover, to our knowledge, no other studies have shown that any such ecological relationship exists. We believe this finding may be purely attributed to the placement of camera traps on the forest roads and trails. Apart from sloth bears (and other carnivores), forest roads and trails are intensively used daily by local people living inside STR. We were interested in the relationship between human occurrence and sloth bear abundance as a matter of concern regarding the possible negative interactions between humans and sloth bears. However, to cope with extensive human pressure in STR, sloth bears have shown a fine-scale seasonal spatio-temporal segregation (Chaudhuri et al., 2022). On the other hand, sloth bears avoided areas close to villages in STR, which is in agreement with previous studies (Babu et al., 2015; Puri et al., 2015; Paudel et al., 2022; Pokharel et al., 2022). Such behavior could be due to the better availability of forage and shelter (day-resting den sites) in relatively undisturbed habitats.

Nevertheless, avoidance of human settlements does not necessarily curb the possibilities of conflict between humans and sloth bears, given the prevailing extensive anthropogenic pressure in STR. Sloth bears are also known to raid crops opportunistically and forage on fruits of *Ziziphus mauritiana*, *Mangifera indica* and *Syzygium cumini* found in the villages but less abundant inside the forest (Bargali, Akhtar & Chauhan, 2004; Palei, Debata & Sahu, 2020). Habituation to human settlements often leads to severe conflicts between humans and sloth bears (Bargali, Akhtar & Chauhan, 2005; Dhamorikar et al., 2017; Prajapati, Koli & Sundar, 2021), so there are significant conservation implications for this species in human-dominated landscapes.

Caveats and limitations

Our study has several limitations and caveats. We deployed camera traps to maximize the probability of capturing tigers and co-predators but did not specifically target a single species, i.e. sloth bear. We could not deploy camera traps intensively due to villages and agricultural fields, which resulted in an unequal number of camera traps (1-4) in each of the larger grids (12.40 km^2). Such design may have affected the detection of sloth bears as they are presumed to raid crops and forage on fruits in the villages situated inside the core area of STR. Currently, we restricted the camera trap survey to the core area of STR; however, expanding the sampling area to include the buffer zone could have improved the overall estimates of the sloth bear's relative abundance. We could not assess the seasonal variation of mean site abundance of sloth bears, as studies conducted elsewhere reported a distinct seasonal variation in sloth bear density (Garshelis, Joshi & Smith, 1999).

Conclusions

We demonstrated the application of the N-mixture model to estimate the relative abundance of sloth bears from camera trap data. Also, choosing an appropriate probability distribution is crucial to providing ecologically realistic estimates. The N-mixture and RN models with Poisson distribution could be suitable options for estimating the relative abundance of solitary, elusive carnivores, despite the better statistical properties of models with ZIP and NB distributions. However, limited knowledge of the ranging patterns of sloth bears in human-dominated landscapes and substantial variation in home range estimates throughout the species' distributional range could affect the choice of grid size (12.40 km^2 grids in the current study) for the N-mixture model. If the goal is to detect the population changes over time, site-structured models are rarely effective; especially if knowledge of certain aspects of the species' ecology (e.g., home range, population density) is limited (Morin et al., 2022). Future studies to monitor population changes of sloth bears should focus on the placement (i.e., spacing) of camera traps based on a more thorough understanding of their ranging patterns (Fuller et al., 2022). Also, a spatially extensive sampling area with several spatial replicates at each sampling unit (depending on the feasibility and size of the sampling unit) is required to assess the abundance with precision in the case of a low-density population of sloth bears. We highly recommend carrying out density estimation by applying a spatially explicit capture-recapture framework through genetics and comparing results with site-structured models (N-mixture and RN models) and other suitable camera trap-based methods (USCR, SPA, and distance sampling). However, we believe that, in the absence of absolute abundance estimates for an unmarked, habitat generalist, and conflict-prone species like the sloth bear, knowledge of an ecologically realistic relative abundance and associated factors governing the same are imperative for its long-term conservation in a human-dominated landscape.

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