# AN ASSESSMENT OF ABUNDANCE, HABITAT USE AND PREY SELECTION BY CARNIVORES IN KHANGCHENDZONGA BIOSPHERE RESERVE, SIKKIM

*Thesis submitted to the*

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*The Award of the Degree of*

**DOCTOR OF PHILOSOPHY**

**IN**

**WILDLIFE SCIENCE**

**By**

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Dedicated

to

My Parents…

भारतीय वन्यजीव संस्थान
Wildlife Institute of India

# Certificate

This is to certify that the thesis of Mr. Tawqir Bashir entitled **‘An Assessment of Abundance, Habitat Use and Prey Selection by Carnivores in Khangchendzonga Biosphere Reserve, Sikkim’** is an original piece of work submitted to Saurashtra University, Rajkot, Gujarat for the award of the **Doctor of Philosophy** in **Wildlife Science.**

Mr. Tawqir Bashir has put more than five term of research work embodied in this thesis under our guidance and supervision. This work presented in this thesis has not been submitted for any degree of any other University or Institution

Place: Dehradun
Date: 09-12-2013

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**APPENDIX I**

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**Tawqir Bashir**

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# *Executive Summary*

Sound ecological information is a prerequisite for any wildlife conservation action plan to be successful. Carnivores in this regard have being recognized as focal species for research and landscape level conservation planning. But, in the high altitude landscapes such as the Himalayas not much ecological information is available, particularly the eastern Himalaya because extreme weather conditions, rugged terrain and remoteness of the area has always challenged scientists for long-term research and monitoring. The present study conducted for about five years (2008-2012) was therefore designed to generate some baseline information on the carnivores of the least explored part of the eastern Himalayan biodiversity hotspot i.e., the Khangchendzonga Biosphere Reserve (BR~2,620 km²) with an attempt to assess their diversity, distribution, abundance, space use, food habits, activity patterns and the nature and extent of human-carnivore conflict, if any. An effort was also made to use this information in predicting suitable and priority areas for conservation action and in developing a protocol for long-term population monitoring of these species.

Reconnaissance surveys were conducted in seven watersheds of the Khangchendzonga BR and the *Prek chu* watershed (182 km²) was selected for intensive field work. Remote and ruggedness of the terrain limited all field activities to be carried out in the expedition mode. Trail monitoring (n=24; 1.5-7 km; 784 walks), sign surveys (n=32), camera trapping (n=27) and local interviews (n=72) were conducted to assess the diversity, distribution and abundance of carnivores. Continuous camera trapping was done from 2009-2011 in different habitats and elevation classes of the *Prek chu* catchment in a 2×2 km grid based design with a total effort of 6,910 effective camera days. A small camera trapping effort of 250 trap days was also conducted in the trans-Himalayan part of the Biosphere Reserve i.e., *Lhonak chu* catchment following the same grid based design. Camera trap captures were used to estimate relative abundances expressed as photo-capture rates (# captures/100 trap days = RAI) and absolute abundances of carnivore species. Species specific trapping designs and sampling sessions were used to estimate absolute abundance and density of carnivores, e.g., spatially-explicit capture recapture models (SECR- likelihood and Bayesian with MCMC simulation) for individually identifiable species and occupancy based abundance models (Royle's Biometrics-repeated count data) for individually non-identifiable species.

The study confirmed the presence of 19 species of carnivores from the KBR including 11 species of global conservation significance. Among these species highest overall photo-capture rate was observed for yellow-throated marten (5.76 ± 1.23) followed by red fox (4.33 ± 1.15), while lowest photo-capture rate was recorded for common leopard (0.036 ± 0.036) and jackal (0.038 ± 0.038). SECR models estimated the density of leopard cat as 17.52 ± 5.52 individuals/100 km$^2$ which was higher than the estimate for snow leopard (4.77 ± 1.81) and large Indian civet (10.98 ± 3.38), while the occupancy based models indicated yellow-throated marten as the most abundant carnivore of the study area (Average abundance/sample unit [$\lambda$] = 1.34 ± 0.31) followed by red fox (0.73 ± 0.24). In the trans-Himalayas, camera traps detected the presence of only two carnivore species including Tibetan wolf and red fox with photo-capture rates of 4.1 ± 2.05 and 3.2 ± 2.71, respectively.

In order to understand the factors that govern the co-existence of carnivores in the area their spatial, dietary and temporal patterns of recourse use were also assessed. Occupancy modelling was used to assess the pattern of space use by carnivores in relation to 11 site and two sampling covariates. The estimated site occupancy was highest for yellow-throated marten ($\Psi$ = 0.348 ± 0.106) as it was detected in 13 out of 41 sampling sites, while the lowest site occupancy was estimated for large Indian civet ($\Psi$ = 0.157 ± 0.0576). Elevation was determined as the best predictor for the occupancy (intensity of space use) of snow leopard and stoat with positive influence, and for masked palm civet, large Indian civet and leopard cat but with negative influence. Presence of *krummholtz* habitat for red fox, tree cover for yellow-throated marten and Asiatic golden cat, while both broadleaved and coniferous forests were determined as the best predictors for the occupancy of Asiatic black bear. Dietary patterns of carnivores assessed through scat analysis revealed blue sheep as the main prey of snow leopard occurring in 44.7% of the scats, while pika was found to be the main prey for red fox (59.9%) and rodents for yellow-throated marten (56.2%). Jacob's selectivity index however indicated that snow leopard preferred Himalayan tahr (0.623), red fox preferred pika (0.989), yellow-throated marten preferred rodents

(0.955) and galliformes (0.889), and wild dogs preferred dzo (0.63) and serow (0.591) more than their availability. In addition, the analysis of leopard cat and jackal scats indicated rodents as their main prey. The Levins' standardized niche breadth and body weight of the carnivores showed an almost linear relationship ($R^2 = 0.391$) indicating that larger species tend to be more generalist with a wider dietary spectrum. The activity profile of carnivores deduced through the analysis of camera trap captures revealed a bimodal activity for snow leopard mainly during later part of the day (dusk and midnight). Strict nocturnal activity pattern was observed for large Indian civet, masked palm civet, leopard cat and red fox, while diurnal activity was observed for yellow-throated marten, stoat and wild dog. The activity profile of black bear and golden cat was however arrhythmic.

Based on the ecological information generated on the species habitat suitability (HS) modelling with the help of Remote Sensing (RS) and geographical information system (GIS) was carried out to predict the extent of suitable areas for the occurrence of these carnivores in the entire KBR landscape. Presence only habitat suitability modelling [ENFA] was used to predict the suitable areas and develop habitat suitability maps. Among the carnivores red fox (1.78), snow leopard (1.67), and Asiatic golden cat (1.59) exhibited high global marginality values indicating that these occupied a relatively small portion of the set of environmental conditions available in KBR, however the lowest values of global tolerance index exhibited by large Indian civet (0.18), masked palm civet (0.21) and snow leopard (0.21) indicated their low tolerance towards deviations from the optimal conditions. The predicted distribution of suitable areas for the occurrence of carnivores in the extent of KBR was generally fragmented with large Indian civet representing an extreme case (maximum fragmentation). The ENFA predicted just 135.7 km$^2$ area as highly suitable for the occurrence of snow leopard out of the total 2,620 km$^2$ of KBR landscape. The proportion of predicted highly suitable area was however minimum for masked palm civet (25.72 km$^2$) followed by large Indian civet (39.11 km$^2$) and leopard cat (77.13 km$^2$), however it was maximum for yellow-throated marten (315.59 km$^2$). High Boyce index of the HS models (ranging from 0.48 to 0.96) indicated their ability to discriminate between suitable and unsuitable areas and thereby validated their power of appropriate prediction.

In order to assess the nature and extent of human-carnivore conflict local interviews were conducted using a semi-structured questionnaire during which 233 affected households belonging to 50 villages were sampled in 2009. A total of 296 instances of conflict [including crop raiding (207), livestock killing (75), and attack on human (14)] were recorded. Number of incidents of conflict were found positively correlated with the elevation of the village and negatively correlated with the distance of the village from dense forest ($R^2 = 0.59$). Most of the conflict incidents took place between September and November, 2009. Black bear was found to be the most conflicting carnivore species exclusively responsible for the crop damage and attacks on humans; however livestock and poultry depredation was also done by yellow-throated marten (23%) and jackal (15%) besides black bear (55%). In total 18 types of agricultural harvests were found to be raided by bears including maize (*Zea mays*) as the most frequently raided (73% of total incidents) crop. Overall an increasing trend was observed in the occurrence of conflict incidents over the years, although highest frequency was recorded in 2009.

The adequacy of different monitoring methods (sign surveys, trail monitoring [sightings] and camera trapping) for detecting carnivore species was also compared. In this regard, camera trapping proved to be the most suitable method as it was successful in detecting more carnivore species than other methods in all the sampled habitat of the study area. Moreover, the results of the power analysis showed that a sampling effort of at least 2400 effective camera days/annum for a minimum of 14 years is required for detection 3% annual decline in snow leopard population with 80% power. Moreover, a total expense of ~ 2,500,000 INR was calculated to be required for monitoring snow leopard, leopard cat and large Indian civet populations in the *Prek chu* catchment of KBR. Priority areas for the monitoring and conservation of carnivores in KBR were also identified based on the prior information generated during habitat suitability modelling. This indicated the occurrence of few highly important areas mainly in the south-western part of the Biosphere Reserve including *Lampokhri, Relli, Gochela, Basecamp, Pangding, Danpheveer, Kasturi, Tsoka* and *Sachen*, while few in the northern trans-Himalayan part including *Zanak, Rasum, Dolma* and *Rongsa*. Moreover, few areas along the eastern buffer zone of the KBR including *Panchpokhri, Thepala, Narkhola, Tung* and *Lingdem* were also identified as important. Besides this, present study did not provide any strong evidence for direct threats such as illegal hunting for pelt/parts or habitat degradation to carnivores in KBR, although some conservation concerns such as retaliatory killings of carnivores, increasing intensity of livestock and pack animal grazing, occasional hunting of galliformes and ungulates, and outbreak of fatal diseases affecting the ungulate prey populations were also recorded, which need to be managed.

This study provides the first scientific information on the ecology of carnivores in the Khangchendzonga BR and is of high significance to managers for efficient conservation and management of these ecologically important species and their habitats in such intricate habitats where accessibility and data collection is a limiting factor. Regular monitoring and strict vigil of the most suitable habitats predicted during this study and management of other conservation concerns is necessary to safeguard the future of these species and their habitats.

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# Chapter 1

# INTRODUCTION

## 1.1 BACKGROUND

Humans have inarguably had a great influence on animal populations around the world, and thereby drastically reduced the geographic range and populations of thousands of species of animals, and driven many to extinction (Thomas 1956; Butler 2006). Today, it is not uncommon for humans to intervene in the natural world to help manage wildlife populations to keep species from becoming extinct. A crucial first step in managing wildlife populations is simply to keep track of them. Population monitoring programmes are critical for assessing biodiversity, identifying species in decline or at risk of extinction, tracking the spread of invasive and pest species, evaluating the effects of management and general accumulation of knowledge (Engeman 2005; Marsh and Trenham 2008). Although, the actual impetus for population monitoring may be curiosity, basic research, or interest in long-term population trends (Witmer 2005), its goal is generally to develop a scientifically defensible estimation of the status and trends in wildlife resources and to determine whether management practices are sustaining those resources or should be changed (Gibbs et al. 1999).

Understanding the processes that determine the distribution and abundance of organisms is a fundamental component of any ecological study (Krebs 1994). The description and analysis of species distributions play an important role in conservation by elucidating the status of a species or identifying areas for conservation action (Wikramanayake et al. 1998). Moreover, abundance estimates of species need to be accurate and unbiased as these also help in assessing their status and population trends, and are crucial for determining conservation priorities (Silveira et al. 2003). Since, the loss and degradation of habitat has been identified as the primary threat to the world’s mammals affecting 40% of species (Schipper et al. 2008), information on the patterns of space use within a landscape can provide better understanding of the species resource utilization at a finer scale and hence help in conservation of critical habitats. Dietary analysis on the other hand is also considered as an important step in understanding an animal’s ecology because diet directly reflects resource use and can also provide insight into habitat utilization and competitive interactions (Litvaitis 2000). Information of food selection is hence critical in understanding the life history strategies and developing sound conservation recommendations (Miquelle et al. 1996). However, as a matter of concern interactions between humans and wild animals have emerged as the greatest threats to the survival of species throughout the world (Distefano 2005), particularly in tropics and developing countries. Such interactions have resulted in large economic losses and even led to widespread elimination or reduction of populations of many species (Woodroffe and Ginsberg 1998; Woodroffe 2000). Investigations on such vulnerable issues are also crucial for sustainable management and conservation of the wild heritage.

## 1.2 WHY CARNIVORES

Carnivores are frequently used as biodiversity surrogates for landscape level conservation planning (Dalerum et al. 2008), since these are recognized as the monarchs of all biological communities, represent the apex of the food chain, and play a vital role in maintaining the essential balance of the ecosystem (Terborgh et al. 1999). Ecologically, carnivores have been shown to play an important role in regulating populations of prey species and maintaining healthy diverse ecosystems (Berger et al. 2001; Terborgh et al. 2001). In the absence of carnivore species, ecosystems frequently experience trophic cascades and ultimately degradation of biodiversity (Terborgh et al. 2001). Top carnivores are often recognized as keystone species that have disproportional importance in the ecological balance and organization of their communities (Mills et al. 1993) and due to their large home ranges they can even serve as umbrella or flagship species for landscape level conservation (Gittleman et al. 2001).

Comprising of 287 extant species in 123 genera belonging to 16 families (Macdonald 2001), there are increasing concerns about the status and distribution of terrestrial carnivore populations throughout the world (Schaller 1996). Changes in land-use practices, habitat loss and fragmentation, sanctioned human persecution, declines in natural prey, disease, illegal poaching, and increased competition within carnivore guilds have brought about a general decline in several carnivore populations with some species now occupying a fragment of their former range (Gese 2001; Karanth and Chellam 2009). The persistence of these environmentally sensitive and ecologically pivotal species is an indication of the integrity of entire ecosystems (Noss et al. 1996) and these can hence act as effective focal species to evaluate the degree of both fragmentation and connectivity across large landscapes (Crooks and Sanjayan 2006). Moreover, mammalian carnivores can also serve as useful tools for the study of ecological disturbances or for conservation planning and reserve design (Soulé and Terborgh 1999).

## 1.3 MOUNTAIN ECOSYSTEMS AND HIMALAYAS

Mountains are a special world (Schaller 1980) not only because of their panoramic splendour and complexity that has inspired and challenged humans for centuries (Seastedt et al. 2004), but because of their role as a lifeline to the human race. Spreading over one-fifth of the earth’s landscape (Karmokolias 1997) and supplying water to half of the human population (Messerli and Ives 1997; Viviroli et al. 2003), mountain ecosystems provide resources that directly support the lives of about two billion people on earth (Karmokolias 1997; Mountain Agenda 1998). Beyond their common characteristics of having high relative relief (or very marked topographic variation) and steep slopes, mountain ecosystems are remarkably diverse (Ives et al. 1997) and biologically unique providing habitats for an impressive diversity of wildlife that supports an abundance of rare plants and animals.

Himalaya is the youngest, most fragile and dynamic mountain system supporting a rich biodiversity (Mani 1974) and covering ca. 12% of the India’s geographical area (Ghosh 1996). Biogeographically, it is categorized into four provinces viz., North-western, Western, Central and Eastern (Rodgers et al. 2000) and spreads over the six Indian States of Jammu and Kashmir, Himachal Pradesh, Uttarakhand, Sikkim, West Bengal and Arunachal Pradesh (Ghosh 1996). Its location, topography and climate have endowed it with rich and diverse life forms (Rau 1975). This is evident from the fact that of the 372 mammalian species in India as many as 241 species (65%) are recorded in the Himalaya, and as many as 29 (37%) mammalian species listed under Schedule I of Indian Wildlife (Protection) Act (1972) occur in the Himalaya (Ghosh 1996). In spite of its enormous biological, ecological, geo-hydrological, socio-cultural and aesthetic significance, it is being depleted at an alarming pace. As high-elevation ecosystems have become subject to greater human use over the years, carnivores have been much severely affected by such activities than other groups of animals (Johnsingh 1986) either due to persecution, retaliatory killings or depletion of prey base (Mishra 1997; Bagchi and Mishra 2006; Johnsingh et al. 2007; Datta et al. 2008a). Studies have also shown that hunting practices have become severe in higher altitudes relative to their wildlife abundances (Rao et al. 2005; Aiyadurai et al. 2010). Therefore, it is becoming increasingly important to protect the mountain biome and conserve its wilderness. Contrary to this, less attention has been given to variation in species richness in high mountains such as Himalaya, the reason being the remoteness of the area and difficult terrains to work upon (Schaller 1977). Hence, it is crucial to understand and monitor the distribution patterns and status of carnivores from one of the world's most diverse and fragile ecosystem such as the Himalayas.

## 1.4 REVIEW OF LITERATURE ON CARNIVORES IN HIMALAYAS

The diversity and endemism of carnivores in Himalaya is exceptional, however scientific knowledge on many of these is still lacking. Most of the information on carnivores of the Himalayas is based on status surveys. These mainly include surveys on snow leopards in Indian Himalayas and trans-Himalayas (Mallon 1984; Schaller 1988; Fox et al. 1991; Jackson et al. 2006), black bears and brown bears (Sathyakumar 2001; Sathyakumar and Choudhury 2007), wolves (Fox and Chundawat 1995; Habib et al. 2013) and on mustellids and viverids of Northwestern and Western Himalayas (Sathyakumar 1999). In north-east India a number of status surveys and documentations have been done on large and small carnivores (Choudhury 1997ab; Datta 1999; Datta et al. 2008 ab; Jha 1999 ab, 2000; Dada and Hussain 2006; Mishra et al. 2006), red panda (Kakati 1996) and Malayan sun bear (Sethy and Chauhan 2011). Apart from this few carnivore status surveys have also been conducted in other Himalayan countries mainly on snow leopard such as in Nepal (Jackson and Ahlborn 1984), China (Schaller et al. 1988; Turghan et al. 2011), Bhutan (Fox and Jackson 2002), Afghanistan (Habib 2007) and Pakistan (Hussain 2003; Din and Nawaz 2011). A status survey on the brown bears of Pakistan (Nawaz 2007) and on the small carnivores of Nepal has also been conducted (Appel et al. 2013).

Few long-term ecological studies have also been conducted in different parts of the Himalayan range on snow leopard (Oli 1991; Chundawat 1992; Jackson 1996; Ale 2007), common leopard (Chauhan 2008; Kumar 2011), red panda (Pradhan 1998), brown bear (Rathore 2008) and black bear (Charoo 2012; Sharma 2012). Moreover, a couple of studies on the ecology of sympatric carnivores have been carried out in the temperate ecosystem in central Bhutan (Wang and Macdonald 2009) and north-east India (Selvan et al. 2013).

In addition to these long-term studies, some short-term studies on different ecological aspects of carnivores have also been carried out. These generally include studies on distribution, abundance and food habit of snow leopard in Sagarmatha National Park (Shrestha 2008), upper Mustang (Upadhyay 2010) and Kangchenjunga Conservation Area (Karmacharya et al. 2011) Nepal, and Baltistan, Northern Pakistan (Anwar et al. 2011). Food habits of common leopard have also been studied Pauri Garhwal, Uttarakhand (Das 2006) and Dhorpatan Hunting Reserve, Nepal (Aryal and Kreigenhofer 2009). Besides, Aryal et al. (2010) also studied the food habits of red fox in Dhorpatan Hunting Reserve, while Manjrekar (1989) and Sathyakumar and Viswanath (2003) investigated the food habits of Asiatic black bears in Kedarnath Wildlife Sanctuary and Dachigam National Park, Kashmir, respectively. Moreover, a short-term study focusing on the distribution of red panda has also been carried out in Dhorpatan Hunting Reserve, Nepal (Sharma and Belant 2009).

Studies addressing the assessment of human-carnivore conflict and livestock depredation by carnivores have also been conducted. Jackson et al. (1996) studied the livestock depredation by snow leopard in Nepalese Himalaya, while a similar study focusing on Tibetan wolves as well was conducted in the trans-Himalayas of Kibber Wildlife Sanctuary (Mishra 1997) and recently in upper Spiti landscape (Suryawanshi et al. 2013). A study on status and conflict perception of snow leopard was also done by Khatiwada (2004) in Langtang National Park, Nepal, while Ikeda (2004) studied the economic impacts of livestock depredation by snow leopard in Kanchenjunga Conservation Area, Nepal. In the Indian trans-Himalayas, studies on predation on livestock by the snow leopard in Spiti region (Bagchi and Mishra 2006) and mitigation of conflict with large carnivores in Ladakh (Sathyakumar and Qureshi 2003; Namgail et al. 2007; Maheshwari et al. 2012) have also been carried out in the recent past. A similar recent study on snow leopard predation on livestock and wild prey and its conservation implications has been conducted in northern Nepal (Wegge et al. 2012). In addition, assessment of human-black bear conflict has also been done in the Indian Himalayas (Chauhan 2003; Charoo et al. 2011) and Nepal (Yadav et al. 2009). Wang and Macdonald (2006) and Sangay and Vernes (2008) also studied the patterns of livestock predation by large mammalian carnivores, and on dholes (Johnsingh et al. 2007) in Bhutan. Moreover, a single study predicting the patterns, perceptions and causes of human-carnivore conflict has also been conducted in and around Machiara National Park, Pakistan (Dar et al. 2009).

## 1.5 EASTERN HIMALAYA AND ITS BIOLOGICAL IMPORTANCE

Extending to about 720 km, the eastern Himalaya is positioned near to the Tropic of Cancer in terms of its latitude and geographically proximal to the Bay of Bengal. It includes the portion between the Namche Barwa peak (7,750 m) [where the River Brahamputra curves southwards] and the River Teesta in the west covering parts of Nepal, Bhutan, the north-east Indian states of West Bengal, Sikkim, Assam and Aurnachal Pradesh, southeast Tibet (China), and northern Myanmar in its entire expanse. The eastern Himalaya is fragile and susceptible to degradation due to the compound effect of higher rainfall, steeper terrain, narrower extent and fragmentation. A prolonged monsoon from June to September with heavy (80%) precipitation and little (20%) precipitation in winter is the characteristic of the eastern Himalaya (Miller 1987). In humid eastern Himalaya the upper limit of forest varies from 4,600-4,900 m. Due to the heavy rainfall, the ‘treeline’ is higher at 4,000 m, the *Krummholtz* (stunted trees in between treeline and alpine zone) and alpine scrub zones are more wide-ranging, and the permanent snow line lies at 5,500 m.

Being positioned at the convergence of two biogeographic realms, *viz.*, Palaearctic and Oriental/Indo-Malayan (Mani 1974; Corbet 1978; Corbet and Hill 1992), the area has a high potential of sustaining many primitive as well as newly evolved species including both Himalayan and Peninsular Indian elements (Khoshoo 1991). Due to its enormous wealth in terms of biodiversity, the eastern Himalaya is well recognized as one among the 34 global biodiversity hotspots (Myers et al. 2000; Mittermeier et al. 2004) as well as one of the several Global 200 Ecoregions (Olson and Dinerstein 1998). The ecosystem profile of the eastern Himalayan region also includes two endemic bird areas (Stattersfield et al. 1998) and several centres for plant diversity (WWF/IUCN 1995). It also harbours the greatest number of endemic species in India (MacKinnon and MacKinnon 1986; Myers 1988), and along with contiguous regions in China and Southeast Asia it is considered to have constituted the evolutionary cradle for flowering plants (Takhtajan 1969). Similar pattern of high diversity and endemism is also relevant in case of its faunal distribution. The eastern Himalayan habitat supports a high diversity of birds and is one of the most diverse regions in the orient (Crosby 1996). Supporting 22 restricted range bird species of which 19 are endemic (Stattersfield et al. 1998; BirdLife International 2001; Jathar and Rahmani 2006), the eastern Himalaya also represents one of the largest concentrations of globally threatened birds in Asia (Acharya and Vijayan 2010). Among the mammalian species the pygmy hog (*Sus salvanius*), Namdapha flying squirrel (*Biswamoyopterus biswasi*), slow loris (*Nycticebus bengalensis*), hoolock gibbon (*Bunopithecus hoolock*), Arunachal macaque (*Macaca munzala*) and many such other primates, black musk deer (*Moschus fuscus*), leaf deer (*Muntiacus putaoensis*), takin (*Budercos taxicolor*), red panda (*Ailurus fulgens*), sun bear (*Ursus malayanus*), clouded leopard (*Neofelis nebulosa*), Asiatic golden cat (*Pardofelis temmincki)* and many other lesser carnivores are found exclusively in the eastern Himalaya (Datta et al. 2003; Rodgers et al. 2000; Macdonald 2001).

### 1.5.1 Sikkim: an important biodiversity rich area

Sikkim is a landlocked Indian State nested in the Himalayas and situated at the western extremities of the eastern Himalaya. Spread over a geographical area of 7,096 km² and extending about 114 km north-south and 64 km east-west, it is wedged in between the Himalayan kingdoms of Nepal in the west, Bhutan and Chumbi valley in the east, Darjeeling district of West Bengal in the south and a large expanse of trans-Himalayas (Tibetan Plateau) in the north. This thumb-shaped State of Sikkim is the smallest in area after Goa, characterised by mountainous terrain with elevation ranging from 280 to 8,586 m (Mt. Kanchenjunga). The climate ranges from sub-tropical in the south to tundra in the northern parts, with temperature seldom exceeding 28° C in summer and dropping below 0°C in winters.

The unique biogeographic location of Sikkim at the junction of the eastern and the central Himalaya, and a meeting point in between the sub-tropical broad-leaved forests and trans-Himalayan cold desert has endowed this State with extremely diverse and endemic flora and fauna. This is depicted by the occurrence of about 72% of the Himalayan rhododendron species in Sikkim (Singh et al. 2003a). Moreover, in terms of mammals about 40% of the mammalian fauna of Sikkim are Palaearctic in origin and routed through southern China and Tibet. These include carnivores such as *Cuon alpinus, Vulpes vulpes montana, Martes foina, Mustella sibirica, Lutra lutra, Ursus thibetanus* and *Panthera uncia*. The species with Oriental/Indo-Malayan origin account to about 50% of the mammalian fauna and mainly include carnivore species such as *Viverra zibetha, Arctictis binturong, Arctonyx collaris, Paguma larvata, Ailurus fulgens, Catopuma temminckii, Prionailurus bengalensis, Pardofelis marmorata* and *Neofelis nebulosa* (Chattopadhayay et al. 2006).

### 1.5.2 Khangchendzonga BR: ecological information and knowledge gap

The Khangchendzonga Biosphere Reserve (KBR) is an important high altitude wildlife landscape and a repository of many rare and endangered flora and fauna, primarily due to its location and remarkable variations in altitude and rainfall (Singh et al. 2002). It falls among the most important protected areas in the eastern Himalaya (Singh et al. 2002). The world's third highest peak (Mt. Kanchenjunga [8,586 m]) also resides within the Biosphere Reserve. Its adjoining wilderness area holds an array of wildlife habitats that support rich biodiversity and forms an integral part of the people living in the region and provides life-support systems for several million people living in the hills and plains south of the region. In the 35 years history of conservation and management of Khangchendzonga National Park (KNP) and 12 years after the creation of Khangchendzonga Biosphere Reserve (BR), 10 major long term studies had been carried out in and around the Biosphere Reserve.

### 1.5.3 Studies carried out in Khangchendzonga BR

Economics of livestock grazing was studied and its impact on the environment was assessed in North Sikkim, particularly in the *Lhonak* and *Rangyang* watersheds of Khangchendzonga NP (Paljor 1995). Impact of anthropogenic pressure on birds and butterfly communities and their habitats were studied in *Prek* and *Churong* watersheds of KBR along a very popular (*Yuksam-Dzongri*) tourist trekking corridor by Chettri (2000a) focussing on bird species richness, community structure and their relationship with vegetation variables and anthropogenic factors (Chettri 2000b; Chettri et al. 2001, 2005ab), patterns of fire wood extraction and their effects on the vegetation (Chettri et al. 2002; Chettri and Sharma 2006), assessment of traditional knowledge on firewood and fodder values of local communities (Chettri and Sharma 2009) and woody tree, bird and butterfly diversity indices of warm temperate and cool temperate forests (Chettri 2010). Another long-term study (1999-2005) on effect of anthropogenic pressures on the natural resources was carried out in Khangchendzonga BR by Chettri (2005) with more emphasis on the buffer zones of KNP and in the fringe villages (Chettri et al. 2006). At the similar study sites, the impact of grazing on plant diversity and productivity along a tourist trekking corridor in the Khangchendzonga BR by Singh (2000) focussing on vegetation dynamics and livestock-rangeland linkages (Singh et al. 2002), livestock grazing and its management implications (Singh et al. 2003b) and composition, economic use, and nutrient contents of alpine vegetation (Singh and Sundriyal 2005), and on the socio-economic aspects such as the economic benefits and conservation linkages of tourism in context of KBR were also assessed simultaneously by Maharana (2000) focussing on ecotourism evaluation of the sacred *Khecheopalri* Lake inside KBR (Maharana et al. 2000a) along with the assessment of the usability of the calculating method (Maharana et al. 2000b) have been studied. Moreover, a study on the assessment of the landscape change, resource status, human dimension and conflict prevention has also been conducted in the Khangchendzonga BR (Krishna et al. 2002).

Diversity of vascular plants in the entire KBR was assessed and documented through a major study carried out from 1999 to 2004 (Maity 2004 [Maity and Chauhan 2002; Chauhan et al. 2003] thereby describing the wild flowers of KBR (Maity 2007). In the cultural landscape of Khangchendzonga NP, research on ethnography of the local inhabitants and the detailed investigation on the available literature on them in India and England had been carried out by Arora (2004). Ethno-medicinal plant use by the indigenous *Lepcha* tribe of *Dzongu* valley bordering the Khangchendzonga BR has been documented by Pradhan and Badola (2008) which also included an assessment of the population status and conservation needs of rare *Rhododendron maddenii* (Badola and Pradhan 2010). Occurrence of new species of *Panax* was also reported from *Uttarey* and *Yuksam* situated at the western part of the KBR (Sharma and Pandit 2009). Diversity of wild mushrooms in Barsey Rhododendron Sanctuary forming the southern border of KBR has also been assessed and the conservation needs have been discussed by Das (2010). Moreover, an important long term ‘action research’ on interrelationship between alpine vegetation, landscape parameters, and land use patterns in the alpine zone of Khangchendzonga BR has also been carried out by Tambe (2007) that focussed on ecology and community structure of alpine vegetation

(Tambe and Rawat 2010), economics, equity and ecology of the pastoral systems (Tambe and Rawat 2009) and assessment of landscape characteristics and changes in the Khangchendzonga NP (Tambe et al. 2012). Adjacent to the KBR at the Teesta Basin in Sikkim, long term ecological studies have been carried out on the distribution and resource use pattern of reptiles (Chettri 2007) and on the distribution pattern of bird communities along the elevation gradient (Acharya 2008).

### 1.5.4 Information gap and genesis of the present study

In spite of such rich biodiversity the mega fauna of Khangchendzonga BR has not been completely explored. Apart from a few rapid survey carried out for assessing the status of Himalayan tahr (Sharma and Lachungpa 2002) and Takin *Budorcas taxicolor* (Lachungpa 1999) by the Department of Forests, Environment and Wildlife Management (DFEWM), Government of Sikkim and on Southern Kiang (Shah 1994) in the protected areas of Sikkim and a single short term study on the ungulates of Tso Lhamo cold desert (Chanchani et al. 2010) has been conducted in the past. Moreover, a couple of short preliminary surveys on the status weasels and cats have also been carried out (Jha 1999a, 2000). Besides these, no long-term surveys or studies dealing with the aspects such as the status, distribution and ecology of mammals, particularly carnivores and their prey populations in Khangchendzonga NP and BR were conducted. Though the wildlife resources of the area have been under continuous threat and exploitation from centuries as reported in one of the earliest literature, the Sikhim Gazetteer (Gammie 1894), no initiative has been taken to assess and document their status over the years. This has created a gap in the scientific knowledge on the mammalian fauna of the area particularly on those existing higher in the trophic level (carnivores), which is in fact essential for the effective conservation and management of the ecosystem as a whole. This has generated a strong need and scope for research on understanding basic ecological aspects of carnivore communities that inhabit the high-elevation habitats of Khangchendzonga BR.

With this background, present study was initiated to generate baseline information on few ecological parameters (diversity, abundance, habitat use, food habits, and activity patterns) of carnivores in Khangchendzonga BR. An attempt was also made to model the suitable habitat of carnivores in the extent of KBR landscape and assess the extent of human-carnivore conflict in the area. This would help in detecting the significant changes in their status by comparing with baseline data, in developing a long term monitoring protocol for the region and subsequent management planning and legislations.

## 1.6 STUDY OBJECTIVES

Considering the need for baseline information the present study was formulated with the following objectives:

1. To assess the distribution, abundance and density of major carnivores in KBR.
2. To estimate the factors determining co-existence of major carnivores in KBR.
3. To develop a habitat suitability model for major carnivores in KBR.
4. To quantify the nature and extent of human-carnivore conflict in KBR and recommend mitigation measures.

## 1.7 STUDY PERIOD

The study period extended from 1st February 2008 to 31st May 2012. This study was a part of a research project, ‘Developing Spatial Database on the Mammal Distributions and Monitoring Programme for large Carnivores, Prey populations and their Habitats in Khangchendzonga Biosphere Reserve, Sikkim’ initiated by the Wildlife Institute of India, Dehradun in collaboration with the DFEWM, Government of Sikkim and Saurashtra University, Rajkot, Gujarat. All field equipment and logistic support required for this study were provided by the Wildlife Institute of India, Dehradun.

## 1.8 INSTRUMENTS/EQUIPMENTS USED

The following equipments were used to measure/quantify different geographical, climatic and ecological parameters:

1. Altimeter (Pete Bros) with 25m accuracy for Altitude (up to 9,000m)
2. Sunnto compass with 1o accuracy for sighting angle and aspect categories. Aspect was measured on an eight point scale (North: 337°-22°, Northeast: 23°-67°, East: 68°-112°, Southeast: 113°- 157°, South: 158°-203°, Southwest: 204°-247°, West: 248°-292° and Northwest: 293°-336°) using a Sunnto compass
3. LASER Range finder (Bushnell) for measuring sighting distances
4. Slope and canopy cover by ocular estimation
5. GARMIN Etrex H Handheld GPS for geographic position
6. Mercury thermometer with the range between -10°C and 50°C for temperature

7. Rain-gauge for rainfall
8. Binoculars - 10 × 50 for field identification
9. Spotting scope for scan/trail sampling
10. Nikon D 300 SLR Camera with 70 - 300 mm lens for photo documentation
11. Camera traps-- Deer Cam (Deercam Scouting Camera, Non Typical, Inc., Park Falls, WI, USA), Wild-view (wildview xtreme2, Grand Prairie, Texas, USA), Stealthcam (Stealthcam, LLC, Grand Prairie, Texas, USA), Moultrie (Moultrie Feeders, Alabaster, Alabama, USA) and Cuddeback Attack (Cuddeback digital, Green Bay, WI, USA) to observe and collect ecological data on cryptic, elusive and nocturnal animals.

## 1.9 ORGANIZATION OF THESIS

The thesis has been organized into eight chapters. As a general introduction, Chapter 1 describes the background of this study, identifies the research gap, described the need of this study, the study objectives, the review of existing literature, the study period and the Instruments/equipments used in the study. Chapter 2 describes the study area in detail including physiography, climate, biological and socio-cultural aspects. Chapter 3 discusses some general methods used during this study. Chapter 4 discusses the diversity, distribution and abundances of carnivores in the study area. Chapter 5 attempts to discusses the resource use by carnivores at spatial (occupancy modelling), dietary (food habits) and temporal (activity patterns) scales. Chapter 6 deals with the delineation of suitable habitats of carnivores at the extent of the entire Khangchendzonga BR using expert based and presence only habitat suitability modelling approaches. Chapter 7 attempts to assess the extent of human-carnivore conflict along the fringes of Khangchendzonga BR and other parts of Sikkim as a whole. Chapter 8 finally attempts to develop a long-term monitoring protocol for assessing the status of carnivores based on the results of the present study, highlights the major research findings on different carnivore species, major threats and future research directions. Conservation implications and recommendations are also discussed in this final chapter so that the baseline information generated in this research can adequately fulfil the long term conservation needs of this enormously rich yet ecologically fragile landscape of eastern Himalaya.

# Chapter 2

# STUDY AREA

## 2.1 BACKGROUND

The Biosphere Reserves represent characteristic ecosystems in different biogeographic regions and consider human communities as their integral component. In the Indian scenario where mountainous regions cover an area close to 100 million hectares and home a rich biodiversity, only few areas come under Biosphere Reserve management. The protected area network in the Indian Himalayan region consists of 7 biosphere reserves including Khangchendzonga Biosphere Reserve (KBR), 31 National Parks and 111 Wildlife Sanctuaries (Mathur et al. 2002). For the conservation and management of high-elevation ecosystem of the Sikkim Himalaya, Khangchendzonga National Park (NP) was created in 1977 with an area of 880 km² which was later expanded to 1,784 km². The conservation and management efforts in Khangchendzonga NP were further strengthened by an addition of 836 km² as buffer zones to constitute the Khangchendzonga Biosphere Reserve (BR) in 2000 (Sharma and Lachungpa 2002).

## 2.2 LOCATION AND AREA

The Khangchendzonga BR is located in a landlocked Indian State of Sikkim nested in the Himalayas between 27$^{o}$ 30′ to 27$^{o}$ 55′ N and 88$^{o}$ 02′ to 88$^{o}$ 37′ E and spreads across the districts of North, West, and a small portion of South Sikkim (Figure 2.1 a, b). It is the highest protected area in India and the third highest in the world. Khangchendzonga BR is connected to the adjacent Khangchendzonga Conservation Area in eastern Nepal, Barsey and Maenam Wildlife Sanctuaries in Sikkim and Singalila BR in Darjeeling district of West Bengal, with a number of corridors (Tambe 2007). According to the biogeographic classification of India (Rodgers et al. 2000), KBR is categorized under Biogeographic province 2C (central Himalaya), while its northern part under Biogeographic province 1B (trans-Himalaya—Tibetan plateau) characterized as a high-altitude cold desert with its biota of Palaearctic affinity.

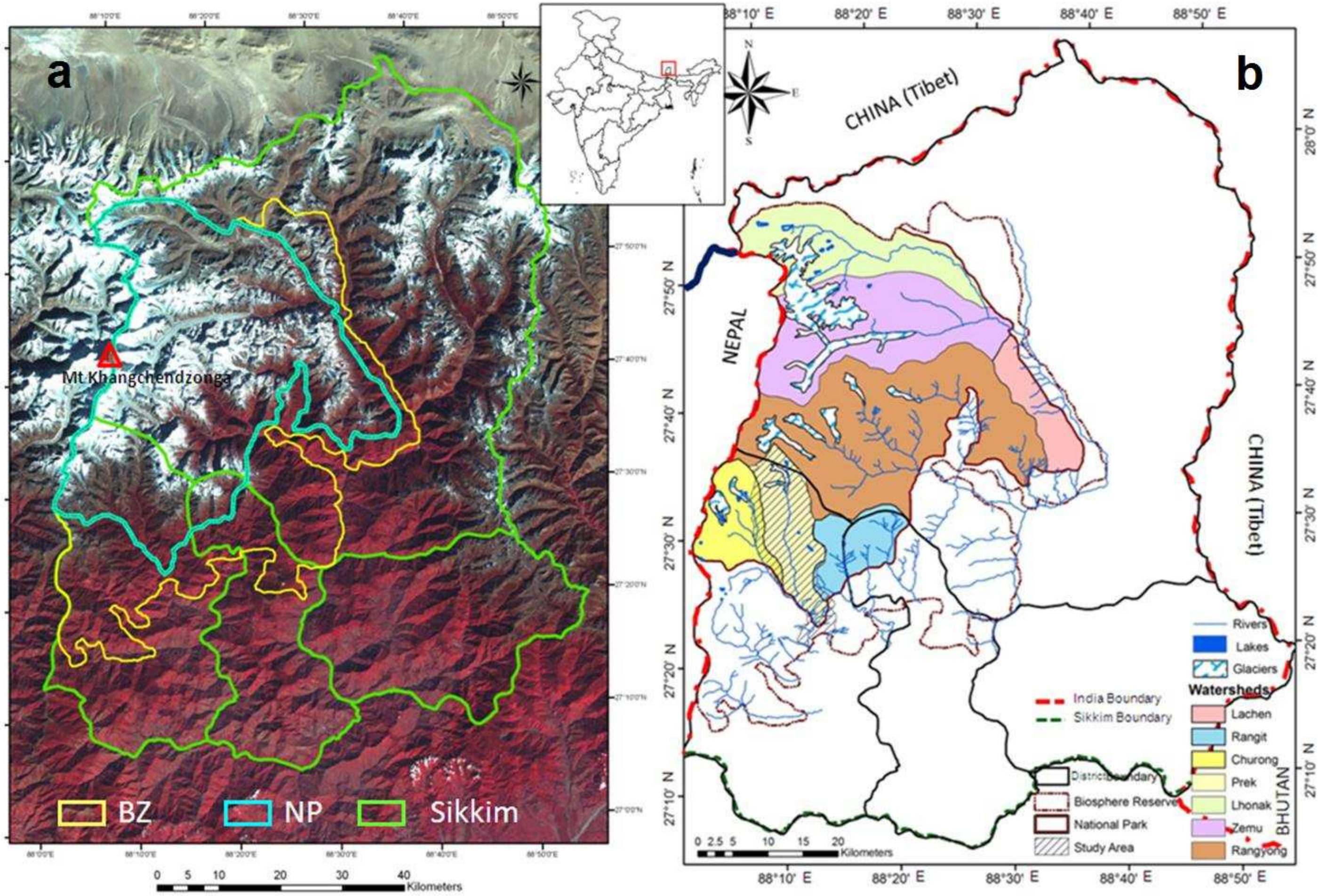
The Khangchendzonga BR covers an area of 2619.92 km$^2$, and constitutes almost 36.92% of the total geographic area of Sikkim (Krishna et al. 2002) which is the highest in proportion compared to rest of the Indian States. The Singalila range separates Sikkim from Nepal and forms the western border of Khangchendzonga BR. This ridge is dotted with highest peaks in Sikkim such as *Kokthang* (6,148 m), *Rathong* (6,680 m), *Pandim* (6,693), *Siniolchu* (6,889 m), *Kabru* (7,340 m), etc., and reaches the zenith at the summit of Mt. *Kanchenjunga* (8,586 m [Plate 2.1]). The huge *Kanchenjunga* massif is buttressed by five great ridges running almost in five different directions (Tambe 2007) and contains a number of peaks between 6,000 m to 8,000 m among which few were mentioned above. The entire landscape is enormously rich in biodiversity, highly important as hydrological, environmental and recreational resources and also represents a unique amalgamation of different cultures of several ethnic communities along with their traditional livelihood practices.

## 2.3 PHYSICAL ATTRIBUTES

### 2.3.1 Physiography

***2.3.1.1 Elevation***- The elevation range of Khangchendzonga BR varying from <1000 m to 8,586 m within an aerial distance of approximately 50 km makes this landscape a unique natural heritage hotspot in the world. About 70% of the Biosphere Reserve lies above 3,000 m and 50% above 4,000 m. The different elevation classes that occur inside KBR differ from 1,001-2,000 m to 6,001-7,000 m (about 98% of the entire KBR). Areas below 1,000 m (0.65%) and above 7000 m (0.48%) are insignificant in comparison with the whole Biosphere Reserve. Among the six elevation classes (1,001 m to 7,000 m, 1,000 m interval); the largest area is covered by 4,001-5,000 m class (29%) followed by 5,001-6,000 m class (20%), 2,001-3,000 m class (20%) and 3,001-4,000 m class (19%). Very small area of 1,001-2,000 m (8%) class indicates the steep rise of the terrain in Khangchendzonga BR (Figure 2.2 a).

**Figure 2.1 Geographic location of Khangchendzonga Biosphere Reserve in Sikkim. a) Boundary of KBR overlaid on False Colour Composite LANDSAT imagery, b) Different watersheds in KBR and *Prek chu* watershed as the study area.**



***2.3.1.2 Aspect-*** Although the Greater Himalaya generally runs in an east-west direction, the chief ridge of Khangchendzonga range is aligned in north-south inclination with west-east running transverse spurs (Tambe et al. 2012). The major valleys in the southern and south eastern part of the Biosphere Reserve are oriented north-south thereby creating east and west aspects. On the other hand, the valleys in the central and northern portions are east-west oriented having mostly north and south aspects. Primarily the KBR landscape can be categorized into eight aspect categories such as North (11%), North-East (15%), East (17%), South-East (15%), South (14%), South-West (12%), and West (8%), North-West (7%), respectively (Figure 2.2 b).

***2.3.1.3 Slope-*** In Khangchendzonga BR 0-10° to 60-70° slope classes are found (Tambe 2007). Among all other classes 20-30° class (29%) and 30-40° class (29%) are more prevalent. Gentle slope classes (0-10° and 10-20°) are located at the Northern trans-Himalayan part and along the glaciers. Interestingly, the central and eastern parts of the Biosphere Reserve are steeper as compared to the south-western and northern portions of the glaciated valleys (Figure 2.3 a) as they are not dominantly sculptured by valley glaciers (Tambe 2007).

***2.3.1.4 Land cover-*** The land cover in Khangchendzonga BR can be primarily classified into 14 categories such as perpetual snow (23%), rocks (20%), alpine meadows (5%), moraine vegetation (0.27%), subalpine thickets (14%), sub-alpine fir forest (7%), temperate forest (21%), bamboo thickets (0.68%), Rhododendrons (2%), temperate scrub (2%), sub tropical wet hill forests (1%), moist mixed deciduous forest (0.50%), water bodies (0.56%) and forest gaps (0.5%). Dissolving the minor classes into broad land cover categories, six distinct classes can be visualized (Figure 2.3 b) as mixed subtropical forest (2%), mixed temperate forest (25%), subalpine forest (23%), alpine zone (6%) as forest classes and rock and snow cover (44%) as non-forest class.

### 2.3.2 Drainage and watersheds

The *Tista* and the *Rangit* are the two major river systems that have their origin in and around Khangchendzonga BR. The *Tista* originates from the *Tista Kangse* glacier in the trans-Himalaya of Sikkim and conducts across the inner Himalaya before meeting the Brahmaputra in Bangladesh, while the *Rangit* originates from various glaciers in south western part of KBR. For simplicity, the area of KNP has been divided into seven watersheds or river sub-systems (Figure 2.1 b) namely *Lhonak* (15%), *Zemu* (23%), *Lachen* (5%), *Rangyong* (36%), *Rangit* (6%), *Prek* (8%) and *Churong* (7%).

Among these watersheds, remnants of melted glaciers are only in *Lhonak* (Smith and Cave 1911), while in *Zemu* and *Prek*, active glaciers such as *Zemu* (26 km long) and *Onglaktang* are present. As the *Rangit, Prek, Churong* and *Lachen* rivers flow north-south others flow from west to east (Tambe 2007). A total of 73 glacial lakes occur within KBR covering an area of about 3.34 $km^2$. All of these lakes [Plate 2.1] are above 4,000 m and are locally known as *Pokhri* or *Tsho* (Tambe 2007).

### 2.3.3 Climate

The varying aspects and slopes have led to the occurrence of different climatic zones in Khangchendzonga BR. These include: warm subtropical (<1,000-1,600 m), warm temperate (1,600-2,500 m), cool temperate (2,500-3,300 m), subalpine (3,300-4,000 m), alpine (4,000-5,000 m) and cold deserts (>5,000 m). With monsoon and winter being the two major seasons, the climate can be categorized as cold and wet. The rainy season extends from May to September in the summers and from November to March in the winters. Spring and autumn occur for a very short time during the months of April and October respectively [Plate 2.1]. The annual precipitation decreases from 2,750 mm in the south-eastern part to 750 mm in the north with the average being 2,143 mm (Anonymous 2000). In most of the places, July is the wettest month. The relative humidity is quite high, being more than 70% throughout the year. The mean minimum and mean maximum temperatures vary inversely with altitude. Generally, January is the coldest month and August the warmest (Tambe 2007).

### 2.3.4 Geology and soil

Geologically, the Khangchendzonga BR is composed of rocks that are mainly high grade gneisses consisting of quartz and feldspar with streaks of biotite (GSI 1984; Singh and Sundriyal 2005). In alpine zone, the soils are loose and acidic in nature (pH varies from 4.93 to 5.41) chiefly compose of sand (52-62%) followed by gravel (15-25%), silt (8-12%) and clay (11-14%), with fairly good organic carbon, nitrogen and phosphorus contents (Singh and Sundriyal 2005). About 11% of the total KNP is covered by moderately shallow coarse loamy soil which is excessively drained on steep to very steep (>30°) slopes. Further, sub types of soil depicted as loamy- skeletal isofrigid lithic cryorthents in the trans-Himalayan portion of KNP and coarse-loamy thermic typic haplumbrepts or loamy-skeletal thermic entic hapludolls and others in most of the greater Himalayan parts have also been described (Anonymous 2000).

**Plate 2.1 (A) The Kanchenjunga massif and adjacent glacial moraines; different seasons in KBR (B) winter, (C) spring, (D) summer or monsoon, (E) autumn; glimpses of seven watersheds of KBR (F) *Churong* (G) *Prek* (H) *Rangit* (I) *Rangyang* (J) *Lachen* (K) *Zema* and (L) *Lhonak*; some glacial lakes situated in *Prek chu* catchment: (1) *Gochella pokhri* (2) *Somiti pokhri* (3) *Lam pokhri* (4) *Lakshmi pokhri* (5) *Neer pokhri***

**Figure 2.2 a) Different elevation classes [1,000 m interval] and, b) different aspect classes present in Khangchendzonga Biosphere reserve. The intensive study area (*Prek chu* catchment) is also indicated.**

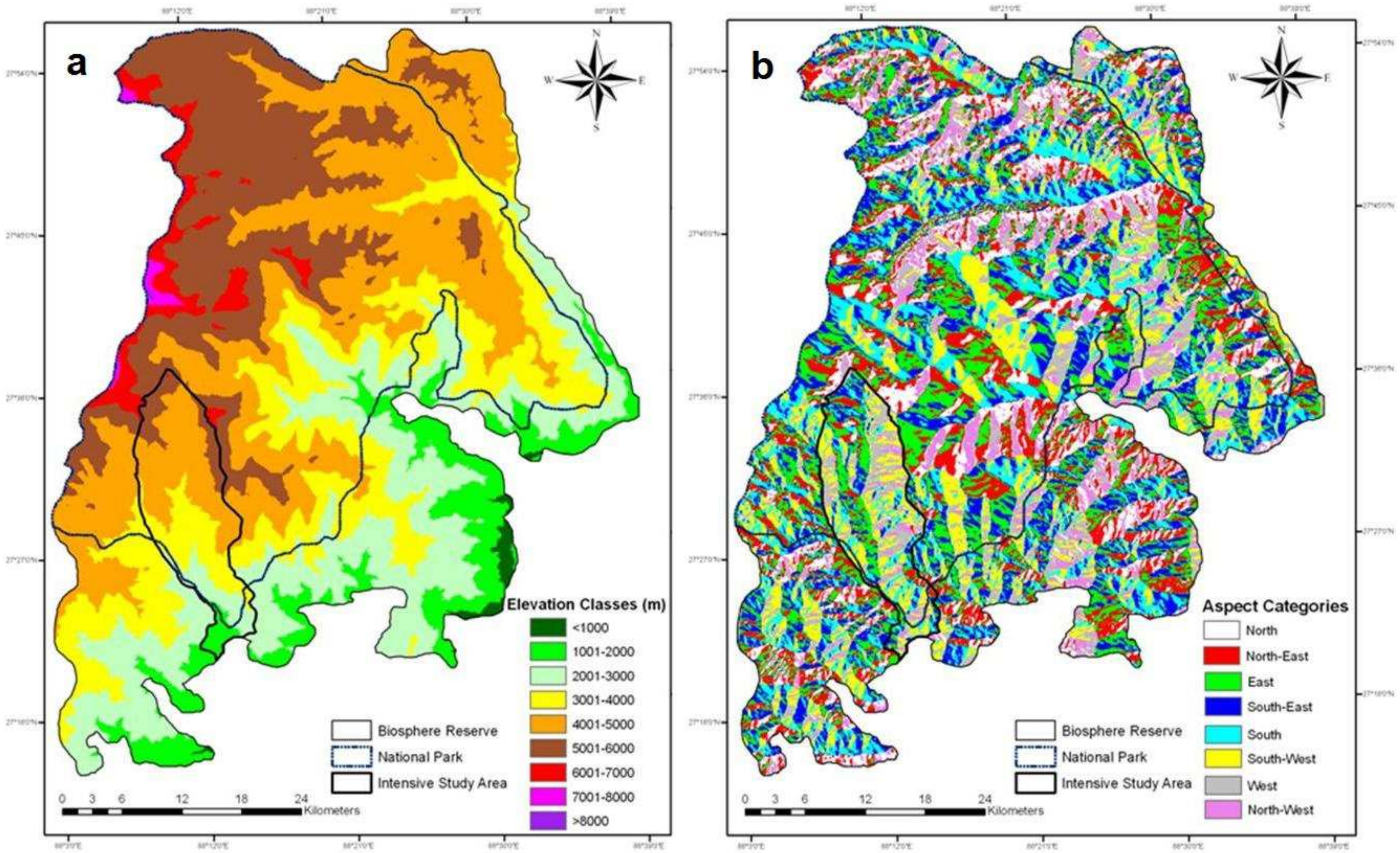
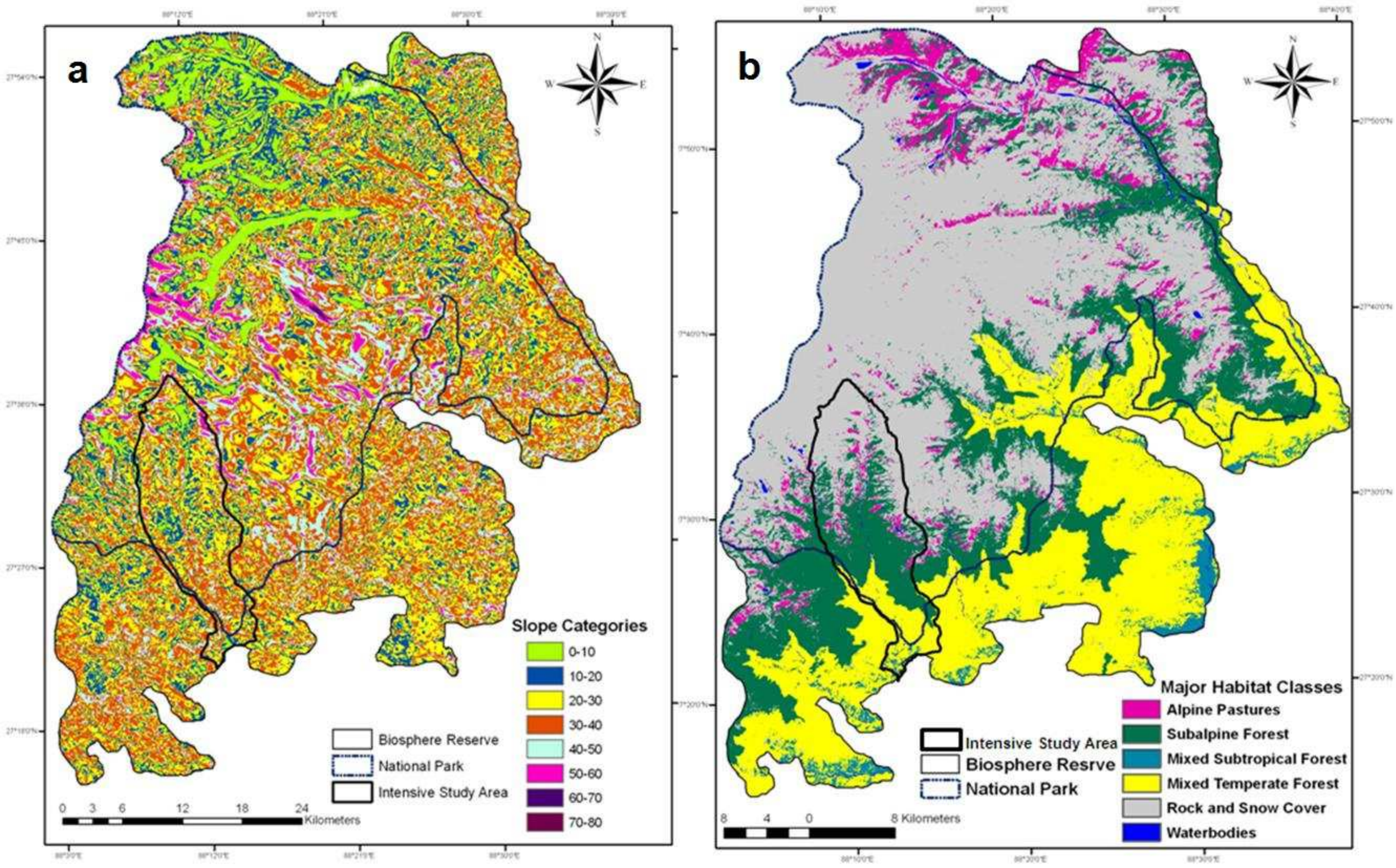


**Figure 2.3 a) Different slope classes [10° interval] and, b) major land cover classes present in Khangchendzonga Biosphere reserve. The intensive study area (*Prek chu* catchment) is also indicated.**



## 2.4 BIOLOGICAL ATTRIBUTES

### 2.4.1 Flora

The vegetation classification of the area as stated by Champion and Seth (1968) demonstrates the presence 18 forest types in states of Assam (including NEFA, Nagaland, Sikkim and Bhutan) and West Bengal. These include: 8B/C1 East Himalayan sub-tropical wet hill forest, 11B/C1(a, b, c) East Himalayan wet temperate forest, 12/C3(a, b) East Himalayan moist temperate forest, 12/DS1 Montane bamboo brakes, 12/DS2 Himalayan temperate parkland, 12/DS3 Himalayan temperate pastures, 12/1S1 Alder forest, 13/C6 East Himalayan dry temperate coniferous forest, 13/CE1 Larch forest, 13/C7 East Himalayan dry juniper/birch forest, 13/1S1 *Hippophae/Myricaria* scrub, 14/C2 East Himalayan sub-alpine birch/fir forest, 14/DS1 sub-alpine pasture, 15/C1 Birch/*Rhododendron* scrub, 15/C2/E1 Dwarf *Rhododendron* scrub, 15/C3 Alpine pastures, 16/C1 Dry alpine scrub, 16/E1 Dwarf juniper scrub.

The Khangchendzonga BR inhabits around 1,580 species of vascular plants encompassing 106 pteridophytes, 11 gymnosperms and 1,463 species of angiosperms. The angiosperms are represented by 1,207 species of dicots and 256 species of monocots, distributed under 598 genera and 138 families, while the gymnosperms are represented by 5 families categorized into 9 genera and 11 species. The dominant families are Asteraceae, Rosaceae, Orchidaceae, Poaceae, Scrophulariaceae, Ericaceae and Primulaceae (Maity and Maiti 2007). Botanical studies carried out in alpine pastures of KBR have revealed that the pasturelands have high species richness with as many as 202 species of plants belonging to 38 families of which majority (90%) are dicots and rest (9%) monocotyledons. Studies have also shown that plant diversity and density increases from April until August and decreased thereafter (Singh and Sundriyal 2005). Life-form distribution patterns show that >50% of the species are chamaephytes (dwarf-shrub), hence signifying high adaptation by the plants.

Recent study on ecology of alpine vegetation in KBR revealed the presence of total 585 species of angiosperms belonging to 67 families and 243 genera (Tambe and Rawat 2010). Compared to the Western Himalaya, proportions of alpine scrub and sedge meadows are higher, whereas herbaceous formations and grassy meadows are limited in extent. The alpine vegetation based on numerical classification has been segregated into 11 types with the extensive ones being *Juniperus indica* scrub, Rhododendron scrub, *Kobresia duthiei* moist meadow, *Kobresia nepalensis* moist meadow, *Kobresia pygmaea* dry meadow, and *Anaphalis xylorhiza* mixed meadow (Tambe and Rawat 2010). In KBR presence of several medicinal and aromatic plants such as *Aconitum ferox* (*Bikh*), *Aconitum spicatum* (*Bikhma*), *Allium prattii* (*Dandu*), *Berginia pupurascens* (*Pakhanbhed*), *Lomatogonium* spp. (*Mahaguru, Sharmaguru*), *Picrorhiza kurrooa* (*Kurki*), *Orchis latifolia* (*Panchamle*), *Nardostachys grandiflora* (*Jatamanshi*) have been identified in the alpine meadows including many other important medicinal herbs (Tambe and Rawat 2006; Tambe 2007). In addition, 36 species of *Rhododendron*s have been reported KBR particularly in subalpine and alpine area of the west KBR (Singh et al. 2003a).

### 2.4.2 Fauna

The faunal diversity of Sikkim comprises of about 125 species of mammals, 550 species of birds, 600 species of butterflies, 33 species of reptiles, 16 species of amphibians and 48 species of freshwater fishes (Hajra and Verma 1996; Lachungpa et al. 2003; Tambe 2007; Chakraborty 2011 [Figure 2.4]). Mammalian diversity of Khangchendzonga BR is represented by 42 species [Plate 2.2] belonging to seven orders and 16 families (Sathyakumar et al. 2011a). These mainly include 19 species of carnivores and eight species of ungulates. Of the 42 species recorded, 18 are of high global conservation significance, categorized as critically endangered (1), endangered (4), vulnerable (4) and near threatened (9) on the IUCN Red list (IUCN 2012). A total of 21 species recorded are characteristically high altitude fauna, although some of them occur over a wide altitudinal range. Ungulates such as blue sheep (*Pseudois nayaur*), Tibetan Argali (*Ovis ammon*), musk deer (*Moschus* spp.), Himalayan tahr *Hemitragus jemlahicus*, serow *Capricornis thar*, goral *Naemorhaedus goral*, barking deer *Muntiacus muntjac* and wild pig *Sus scrofa* are reported to be present in the Biosphere Reserve (Sharma and Lachungpa 2002). Other mammals present in the Khangchendzonga BR include primates such as Assamese macaque *Macaca assamensis* and central Himalayan langur *Semnopithecus entellus*, rodents and lagomorphs such as parti-coloured flying squirrel *Hylopetes alboniger*, orange-bellied squirrel *Dremomys lokriah*, Himalayan crestless porcupine *Hystrix brachyura*, Sikkim rat *Rattus sikimmensis*, and other small mammals such as pikas *Ochotona* spp. Presence of Chinese pangolin *Manis pentadactyla* from the lower part of subtropical forests in *Yuksam* has also been reported (Chakraborty 2011).

Being a part of the Endemic Bird Area (EBA) in the eastern Himalaya and one among the 11 Important Bird Areas (IBA's) in Sikkim (Islam and Rahmani 2004), over 213 species of birds have been recorded in KNP (Chettri 2000ab; Chettri et al. 2005b), including seven species of galliformes (Sathyakumar et al. 2011 b) as blood pheasant (*Ithaganis creuntus*), satyr tragopan *Tragopan satyra*, Himalayan monal *Lophophorus impejanus*, kalij pheasant *Lophura leucomelanos*, snow partridge *Lerwa lerwa*, Tibetan snowcock *Tetraogallus tibetanus* and hill partridge *Arborophila torqueola* [Plate 2.2]. Besides these, few species of reptiles were also recorded during this study including king cobra *Ophiophagus hannah* (Bashir et al. 2010), mountain pit viper *Ovophis monticola*, red necked keelback *Rhabdophis subminiatus*, eastern trinket snake *Orthriophis cantoris*, variegated mountain lizard *Japalura variegata*, Sikkimese rock skink *Asymblepharus sikimmensis*, etc. Also, previous studies on butterflies have reported the occurrence of 195 species in Yuksam-Dzongri area inside National Park (Chettri 2000b; Chettri 2010).

**Figure 2.4 Contribution of different groups to the faunal diversity of Sikkim**

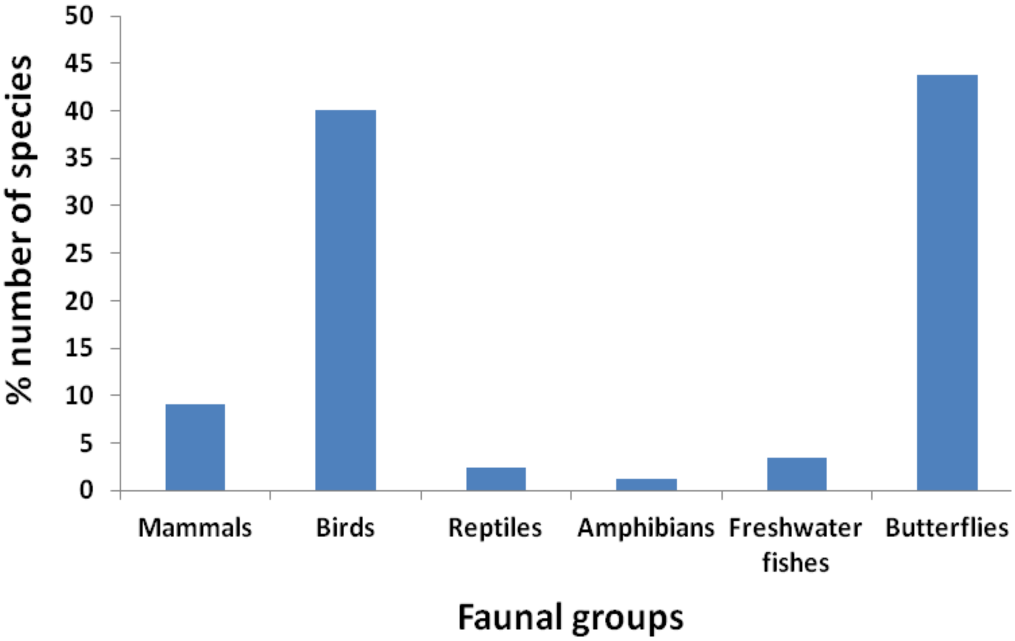


**Plate 2.2 Camera-trap photographs of some mammals present in KBR (1) snow leopard (2) clouded leopard (3) golden cat (4) jungle cat (5) leopard cat (6) wild dog (7) red fox (8) black bear (9) red panda (10) large Indian civet (11) yellow throated marten (12) masked palm civet (13) stone marten (14) Siberian weasel (15) pale weasel (16) Assamese macaque (17) central Himalayan langur (18) wild pig (19) barking deer (20) brown goral (21) Himalayan serow (22) Himalayan tahr (23) blue sheep (24) musk deer (25) Himalayan crestless porcupine (26) Grey-faced flying squirrel (27) hoary-bellied squirrel (28) orange-bellied squirrel (29) pika (30) Sikkim rat; photographs of some galliformes present in KBR (A) blood pheasant (B) Himalayan monal (C) Satyr tragopan (D) kalij (E) hill partridge (F) snow partridge**

## 2.5 SOCIO-CULTURAL ASPECTS

The Khangchendzonga landscape is sacred to both the Hindus and Buddhists and is believed to have been blessed by Guru Padmasambhava, a revered sage of the 8th century AD. Apart from a lone village (*Tshoka*) made up of 10 households, the remaining 9,500 households with a total population of nearly 30,000 live in 29 villages outside but adjacent to KNP (Census of India 2001). Several indigenous communities' *viz., Lepcha, Limbu, Gurung, Mangar, Chettri, Bhutia*, *Sherpa, Lachenpa,* and Tibetan *Dokpa* have inhabited in this landscape for quite a few centuries. High degree of variation in altitude, the exceptionally high biodiversity, and the existence of nine major ethnic communities on its fringes make this Biosphere Reserve a global natural and cultural heritage site. The cultural diversity of the local communities living adjacent to KBR has given rise to a variety of livelihoods including large cardamom farming in the sub-tropical belt and livestock rearing in the temperate and alpine belt (Tambe 2007). Traditionally, the *Gurung*s and *Manger*s were the shepherds, the *Bhutia*s were the traders and yak herders, the *Lepcha*s and the *Limbu*s were the hunter-gatherers and shifting cultivators, the *Chettri*s and *Bahun*s were the agro-pastoralists rearing cattle, and the Tibetan *Dokpa*s were the nomadic yak herders in the trans-Himalaya.

Keeping in view the large extent of KBR, intensive field work was majorly carried out in ***Prek chu*** (*chu* = river) **catchment** area. Selection of *Prek chu* catchment was based after investigating aspects such as availability of habitat types, accessibility, logistic support and perception of local communities. Besides, this catchment area represented all habitat types, elevation, aspect and slope classes of KBR except the trans-Himalayan characteristics, hence was most feasible to sample. Apart from the *Prek* catchment, shot term surveys were carried out in other catchments particularly in ***Lhonak*** valley which represents the trans-Himalayan cold desert habitat of KBR.

## 2.6 *Prek chu* CATCHMENT AREA

The *Prek chu* has its origin in the *Onglaktang* glacier and while flowing north and south it joins the *Churong chu*, after passing through *Thangsing, Lamune, Gochela, Areylungchok* and *Lampokhri* areas. Its main tributaries are *Yangzee chu* and *Thansing khola*. The confluence of *Prek chu* and *Rathong chu* is situated near *Sachen*, and southwards to this confluence the river is known as *Rathong chu* which ultimately meets the *Rangit* River. It is located [27° 37’N, 88° 12E - 27°21N, 88° 17E] in the western part of Sikkim. The *Prek chu* valley opens up in the upper reaches and is spread over an area of 144 km$^2$ which is about 7% of the total area of KBR. This catchment along with the buffer parts near *Yuksam-Khecheopalri* (totaling 182 km$^2$) was delineated as the intensive study area (Figure 2.1 b). The immensely popular *Yuksam-Dzongri-Gochela* trek lies in this watershed encompassing elevation from 1,780 to 5,010 m (Chettri et al. 2001).

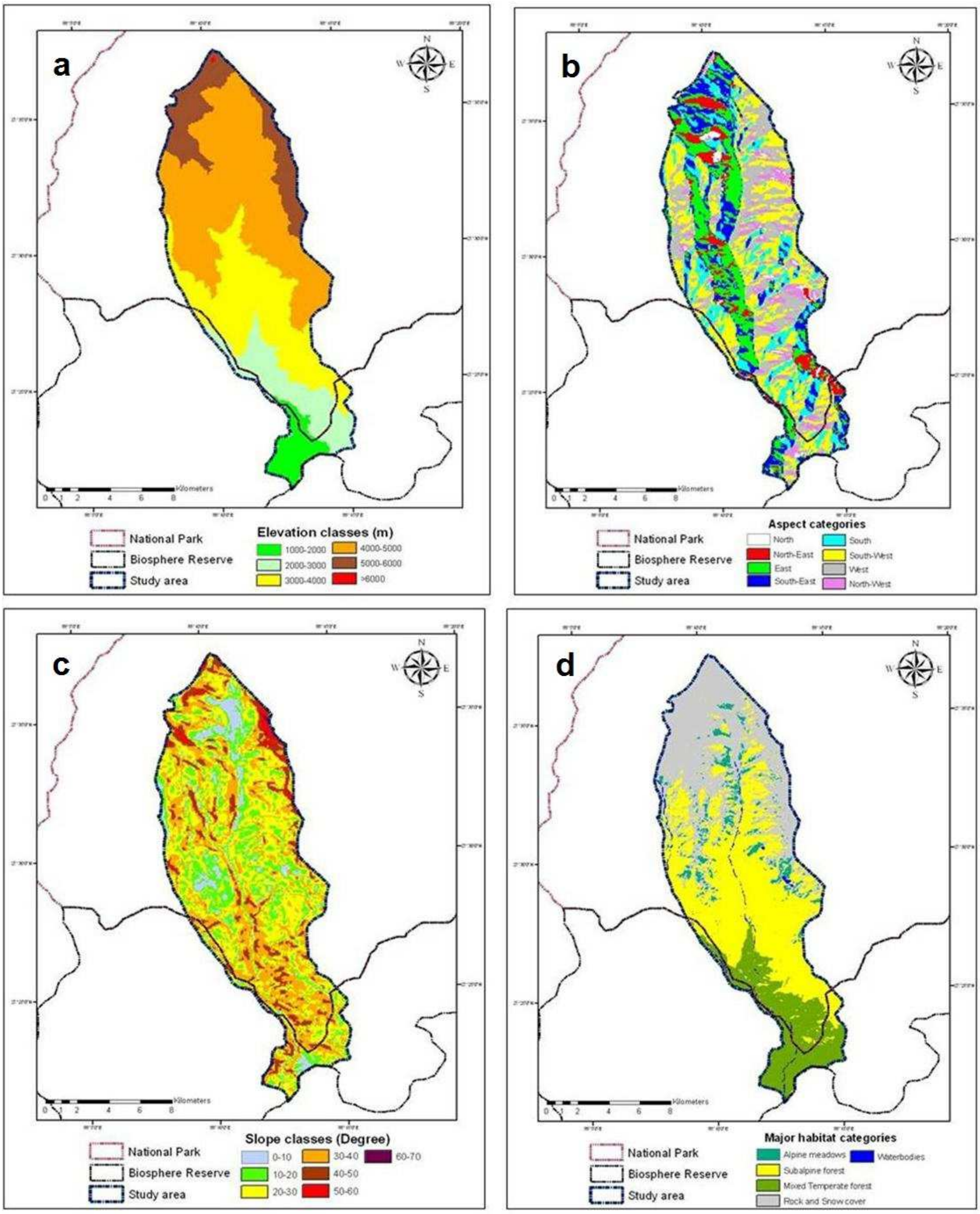
### 2.6.1 Physiography of *Prek chu* catchment

The highest and the lowest elevations of the *Prek chu* watershed are 6,691 m (summit of *Pandim*) and 2,200 m respectively, with a mean of 3,562 m (Tambe 2007). The following elevation classes (Figure 2.5 a) are observed – 1,001-2,000 m (5%), 2,001-3,000 m (13%), 3,001-4,000 m (25%), 4,001-5,000 m (44%) and 5,001-6,000 m (13%). *Prek chu* watershed has the maximum number of 16 glacial lakes (22% of the total glacial lakes in KBR). The various aspect categories (Figure 2.5 b) found in the *Prek chu* catchment include North (2%), North-east (6%), East (15%), South-east (12%), South (14%), South-west (24%), West (21%) and North-west (6%). The slope classes in this watershed (Figure 2.5 c) can be categorized into six types as: 0-10° (7%), 10-20° (19%), 20-30° (34%), 30-40° (28%), 40-50° (10%) and 50-60° (2%).

### 2.6.2 Major habitats in *Prek chu* catchment

The *Prek chu* watershed can be divided into following habitat classes (Figure 2.5 d) *viz.,* mixed sub-tropical (1%), mixed temperate (16%), sub-alpine (36%), alpine pastures (5%), rock and snow cover (41%) and water bodies (1%). The alpine zone can be subdivided into *Juniper* scrub habitats, alpine meadows and marsh meadows. The sub-alpine zone can be further sub-divided into *Fir-Birch-Rhododendron* forest and the stunted Rhododendrons of *Krummholdtz* zone which in many areas are extended up to higher elevation (3,700 m to 4,200m) and intermingled with alpine meadows. Mixed temperate forests are present up to 3,000 m from where the *Fir-Birch-Rhododendron* forests start. In between, a transition zone dominated by *Tsuga demosa* is present from 2,700 m to 3,100 m in and around the *Tshoka* village inside the National Park [Plate 2.3]. Mixed subtropical forests are present at the lower elevations near *Yuksam* and extend up to 1,900 m.

**Figure 2.5 a) Different elevation classes [1,000 m interval], b) different aspect classes, c) different slope classes, and d) different habitat categories, present in the *Prek chu* catchment of Khangchendzonga BR.**



### 2.6.3 Precipitation and temperature in *Prek chu* catchment

The annual rainfall in the *Prek chu* catchment varied from 233 to 2,340 mm and the mean lied around 1,130 mm (Figure 2.6) during the year 2009 [present study]. July was the wettest month while January was the driest. The watershed has a typical monsoon climate with heavy rainfall during June-September and very less precipitation during the winter months. The average minimum and maximum temperatures varied inversely with altitude and January was the coldest and August the warmest month. Alpine zone was the coldest according to the average minimum temperature and the temperate zone the warmest (Table 2.1). The relative humidity is depicted as more than 60% all through the year in temperate, subalpine and alpine zones, and in June-July it crosses 90% in subalpine areas (Chettri 2000a; Tambe 2007). However, during the study period, the weather was generally clear throughout the winter season, but with heavy snowfalls at the end of March during 2009, 2010 and 2011 and at the beginning of May in 2009 and 2012, respectively.

**Figure 2.6 Monthly variations in rainfall in *Prek chu* catchment at *Yuksam***

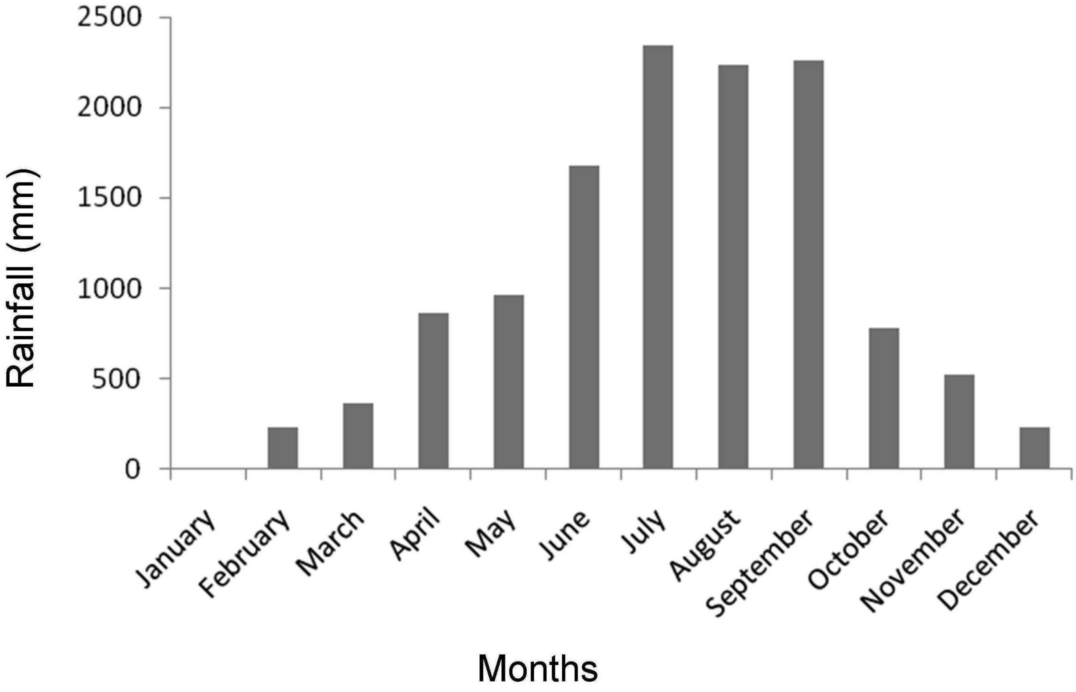


**Table 2.1 Average Minimum (Av. max) and Maximum (Av. min) Temperature (°C) in *Prek chu* catchment**

| Habitat | Av. max | Av. min | max | min |
|---|---|---|---|---|
| Alpine | 4.30 | -8.67 | 12.8 | -16.11 |
| Subalpine | 11.07 | -2.52 | 15 | -8.89 |
| Temperate | 16.55 | 4.84 | 33.9 | -1.11 |

### 2.6.4 Socio-cultural profile of *Prek chu* catchment

Subba being the major ethnic people followed by *Bhutias*, *Lepchas*, *Nepalis* and *Tibetan* Refugees (at *Tshoka*); Hinduism and Buddhism are two main religions practiced in this area. The oldest Buddhist monastery of Sikkim is situated at *Dubdi* (1710 AD) near *Yuksam* [first capital of Sikkim (1642 AD)]. *Yuksam* is the largest village and major activity centre for eco-tourism [Plate 2.4]. In *Prek chu* watershed traditional livestock grazing by *Gurung* shepherds was being extensively practiced and later the *Bhutia* yak herders used the alpine meadows for intensive yak grazing. Although, supervised livestock grazing is officially banned based on the initiatives taken by Forest department, local NGOs and local people, free ranging pack animals still graze in the alpine meadows during monsoon. Agriculture and tourism related business such as hotel, home stay, restaurant, trekking gear supply, tour guide, porter, etc., are the main livelihood options for the local communities in this watershed, particularly in *Yuksam* [Plate 2.4].

## 2.7 *Lhonak chu* catchment area

The *Lhonak chu* originates as *Goma chu* from the North and South *Lhonak* glaciers and flows in the west-east direction before joining *Zemu chu* at *Talem*. The total area of this watershed is 243 km² with *Goma chu*, *Putung chu*, *Lungma chu*, *Khora chu* and *Naku chu* as its main tributaries. It is the only watershed which is protected by ridges of 5,000 m on all sides. The two alpine passes namely the *Theu la* (5,235 m) in the south-east and the *Luna la* (5,090 m) towards the east connect it to the *Zema chu* and *Lachen chu* valley respectively (Tambe 2007). It is the highest watershed of KNP with elevation varying from 3,100 m to the top of *Jongsang* peak at 7,459 m and has a mean altitude of 5,250 m. The total relief is 4,359 m, basin length 44 km and the relief ratio 0.10. It is also the driest watershed with the annual rainfall varying from 750 mm to 1,575 mm (mean = 1,334 mm). The *Lhonak chu* also demarcates the northern boundary of KNP and its right bank forms the only trans-Himalayan zone within the park with typical faunal species [Plate 2.5]. The flats along the middle and upper reaches of this river valley serve as important summer pastures for the yaks of semi-nomadic *Dokpa* and *Bhutia* pastoralists based at *Munguthang*, *Thangu* and *Lachen* villages (Tambe 2007). The Indo-Tibetan Border Police is also stationed here manning the international border with Tibet.

**Plate 2.3 Major habitat types in *Prek chu* catchment: (A) subtropical forest, (B) temperate forest, (C) transition between temperate and subalpine, (D) subalpine forest, (E) *krummholdtz* forest, alpine habitats as (F) Juniper scrub, (G) meadow,(H) marsh meadow, non forest habitats as (I) rocky area and (J) perpetual snow cover**

**Plate 2.4 Bird's eye view of the *Yuksam* village, the centre of all socio-cultural activities in *Prek chu* catchment, depicting (A) monastery (B) sacred lake (C) main village market and road (D) Forest department office (E) agricultural fields (F) private forests (G) village households (H) hotels; different livelihood options: (I) livestock grazing (J) pack animal rearing (K) tourism related livelihoods; some sacred places and culturally important monuments and places in *Yuksam*: (1) the coronation throne (2) *Karthok* monastery (3) *Dubdi* monastery (4) deities inside *Dubdi* monastery (5) *Karthok* lake (6) *Chorten***

**Plate 2.5 Different attributes of *Lhonak chu* catchment: (A) Thangu village (B) Potato cultivation at Thangu semi-nomadic *Dokpa* and *Bhutia* pastoralists (C) trans-Himalayan landscape (D) Survey team trekking Luna la (E) Yak grazing near Munguthang (F) Himalayan marmot, (G) Horned lark (H) Grandala.**

# Chapter 3

# GENERAL METHODS

## 3.1 INTRODUCTION

All field activities were carried out in the form of field expeditions - camping in different areas of the intensive study area. One field survey was usually of 8-7 days and all the sampling units were replicated after every 7-10 days. Different conventional field methods such as trail monitoring, sign surveys, camera trapping, scat analysis and vegetation sampling were used to achieve the objectives. For regular and systematic monitoring, base camp was located at *Yuksam* (1,900 m) from where sampling in temperate forests was carried out. Sampling in the sub-alpine and *Krummholtz* region was operated from the first and second advance camps located at *Tshoka* (3,000 m) and *Dzongri* (3,950 m) respectively, while *Thansing* (4,080 m) and *Lampokhri* (4,200 m) represented the third and fourth advance base camps respectively from where sampling in the alpine region was carried out.

## 3.2 RECONNAISSANCE SURVEYS

Exploratory surveys were carried our during the initial months of the study period covering five watersheds (*Churong, Prek, Zema, Lachen* and *Lhonak*) of KBR in order to develop fair knowledge and understanding of the area, and a baseline for further intensive sampling. *Prek chu* catchment was selected as an intensive study area (discussed in Chapter 2). This was followed by application of some conventional sampling methods such as trail monitoring, sign surveys, camera trapping and local interviews. Because of the intricate topography and remoteness of the area all field activities were carried out in the form of expeditions, i.e., camping in different areas of the Biosphere Reserve.

## 3.3 TRAIL MONITORING AND SIGN SURVEYS

Trails were identified and marked in different habitats with slight modification to conventional line-transects (Burnham et al. 1981) as curvilinear transects/trails that have been used to overcome the constraints of sampling in areas having steep, rugged and inaccessible terrain (Sathyakumar 1994; Vinod and Sathyakumar 1999). Ridge walking and sign surveys along trails, ridges and *nullahs* (Chundawat 1992; Sathyakumar 1994) were also carried out. Trail were monitored repeatedly (n=24; 1.5-7 km; 784 walks) within the intensive study area (Table 3.1, Figure 3.1) and sign surveys were carried out once in a month (32 surveys). Trail monitoring and sign surveys were also conducted once in *Churong, Lachen* and *Zema* watersheds and thrice in *Lhonak* watershed. Length of each of transect/trail was calculated using ArcGIS version 9.3. In case of animal sightings and indirect evidences (scats, tracks, pugmarks, scrapes, scratch marks), information on geographic location, elevation and habitat characteristics (major vegetation type, aspect, and slope) were also recorded.

## 3.4 CAMERA TRAPPING

Camera trapping in the *Prek chu* watershed was done following a grid based design. The study area was divided into 4 km$^2$ grids/blocks using Geographic Information System (ARC GIS 9.3). In order to achieve a systematic sampling design the area was categorized into three different survey zones according to the habitats, *viz.*, temperate (1,200–3,000 m), sub-alpine (3,000–4,000 m) and alpine (above 4,000 m) and the camera traps were deployed corresponding to the area covered by each survey zone and the accessibility to the sampling grid in each zone (13 grids in temperate, 12 grids in subalpine and 16 grids in alpine). Within each survey zone, cameras were placed at likely animal-use sites based on the frequency of signs encountered. Since the study species were rare and the area being vast, the strategy was to survey more sampling sites less intensively rather than less sampling sites more intensively (Mackenzie and Royle 2005). Hence, 27 infrared-triggered camera units were deployed at 71 different sites in 41 sampling grids (Figure 3.2).

Camera traps were set with one minute delay between successive activations generally in four-burst mode, placed at 15-30 cm above ground, and attached to a rock or tree trunk at 3-5 m from a trail or point where animal movement might be expected. Cameras were set for 24 hours monitoring and checked at 10-15 day intervals which included changing batteries and memory cards. Head-on, oblique and side-view camera configurations were used to obtain photographs at varying body orientations (Jackson et al. 2006). Camera trapping was done continuously in all the seasons (winter: January-March; spring: April-May; summer: June-September; autumn: October-December). Due to difficult field condition in summer (monsoon), no camera trapping was carried out in the lowermost part of the study area, though this portion was covered intensively during autumn when the intensity of cameras in the alpine region was kept low. Among the 71 camera locations, camera units at 25 locations stopped working within 5 days due to malfunctioning or human interference. A total of 6,910 effective camera days effort was obtained from the rest 46 locations from 2009-2011 including 1,862 in temperate zone, 3,087 in sub-alpine zone and 1,961 in alpine zone, respectively (Table 3.2). For estimating absolute abundance, species specific camera trap designs were adopted operating in different sampling sessions as subsets of the primary design discussed above, which is discussed in detail in Chapter 4. In *Lhonak chu* catchment area, camera trapping was carried out during 2012 for a short period of one month following the same 4 km² grid design. Ten camera units were deployed in 10 sampling grids (Table 3.3, Figure 3.3).

## 3.5 LOCAL INTERVIEWS

In order to assess the awareness of local people regarding the carnivore diversity of the area and to compare it with camera trapping results (Can and Togan 2009) interviews and informal discussions (Mishra et al. 2006) were conducted in 15 villages adjacent to the Biosphere Reserve. Six villages bordering *Prek* and *Churong* catchments and one each in core and buffer zones, six villages bordering *Zema* and *Lachen* catchments, and one village in the *Lhonak* catchment (focusing *Dokpa* Yak herders and Indo-Tibet Border Police personnel) were sampled. In total, interviews and informal discussions were conducted with the family heads of 72 households which included farmers, livestock herders, former hunters and trekking guides during which they were also asked to identify photographs of carnivore species from field guides (Prater 1971; Menon 2003), if ever sighted in the area.

**Table 3.1 Characteristics and repeats of trails used during sampling in *Prek chu* catchment from April 2008 to August 2011 (l = length, E = elevation, S = slope, RW = replicated walks, total length walked = L)**

| Habitat | Trail | l (km) | E (m) | S (°) | RW | L (km) |
|---|---|---|---|---|---|---|
| **Temperate** | Yuksom–Intake | 1.8 | 1800-1700 | 30-45 | 23 | 41.26 |
| | Yuksom–Bakhting | 2.1 | 2000-2300 | 15-30 | 40 | 84 |
| | Yuksom–Sachen | 4.5 | 2000-2500 | 15-30 | 83 | 372.67 |
| | Sachen up-down | 1.7 | 2300-2350 | 30-45 | 38 | 64.6 |
| | Bakhim–Tsokha | 1.8 | 2300-3000 | 30-45 | 68 | 122.4 |
| | Labdang–Kasturi | 6.7 | 2300-3000 | 15-30 | 25 | 166.5 |
| | **Sub-total** | **18.54** | | | **277** | **851.43** |
| **Subalpine** | Tsokha–Ghunsa | 1.9 | 3000-3200 | 15-30 | 39 | 75.66 |
| | Phedang–Kockchurong | 7.2 | 3700-3800 | 15-30 | 25 | 181 |
| | Dzongri–Jamling | 1.7 | 3700-3900 | 30-45 | 31 | 52.7 |
| | Dzongri–Kockchurong | 2.8 | 3700-3900 | 15-30 | 26 | 72.8 |
| | Kasturi–Phedi | 4.1 | 3000-3900 | 30-45 | 19 | 77.71 |
| | Tsokha–Deorali | 3.9 | 3000-3900 | 15-30 | 54 | 212.22 |
| | **Sub-total** | **21.70** | | | **194** | **672.09** |
| ***Krummholz*** | Dzongri–Deorali | 2.0 | 3900-4000 | 0-15 | 48 | 94.08 |
| | Thansing–Pokhrital | 1.8 | 4000-4100 | 30-45 | 24 | 43.2 |
| | Dzongri–Laxmipokhri | 2.0 | 4200-4500 | 15-30 | 28 | 54.88 |
| | Dzongri–Sukhapokhri | 3.7 | 4000-4100 | 0-15 | 31 | 115.63 |
| | Lampokhri–Jaireni | 1.9 | 3900-4200 | 15-30 | 22 | 41.8 |
| | Thansing–Lampokhri | 3.1 | 4000-4200 | 15-30 | 31 | 94.86 |
| | **Sub-total** | **14.41** | | | **184** | **444.45** |
| **Alpine** | Thansing–Neerpokhri | 1.6 | 4000-4200 | 15-30 | 27 | 43.2 |
| | Chamre-Phedhi | 4.6 | 4500-4100 | 15-30 | 12 | 55.2 |
| | Lampokhri–Chamre | 1.8 | 4200-4300 | 15-30 | 27 | 47.79 |
| | Thansing–Somiti | 3.7 | 4000-4200 | 0-15 | 28 | 102.48 |
| | Dzongri–Kiniya | 2.0 | 3900-4500 | 0-15 | 31 | 61.69 |
| | Jemathang–Goechela | 5.1 | 4300-5000 | 0-30 | 4 | 20.4 |
| | **Sub-total** | **18.72** | | | **129** | **330.76** |
| | **Total** | **73.38** | | | **784** | **2298.73** |

**Figure 3.1 Array of monitoring trails in *Prek chu* catchment**

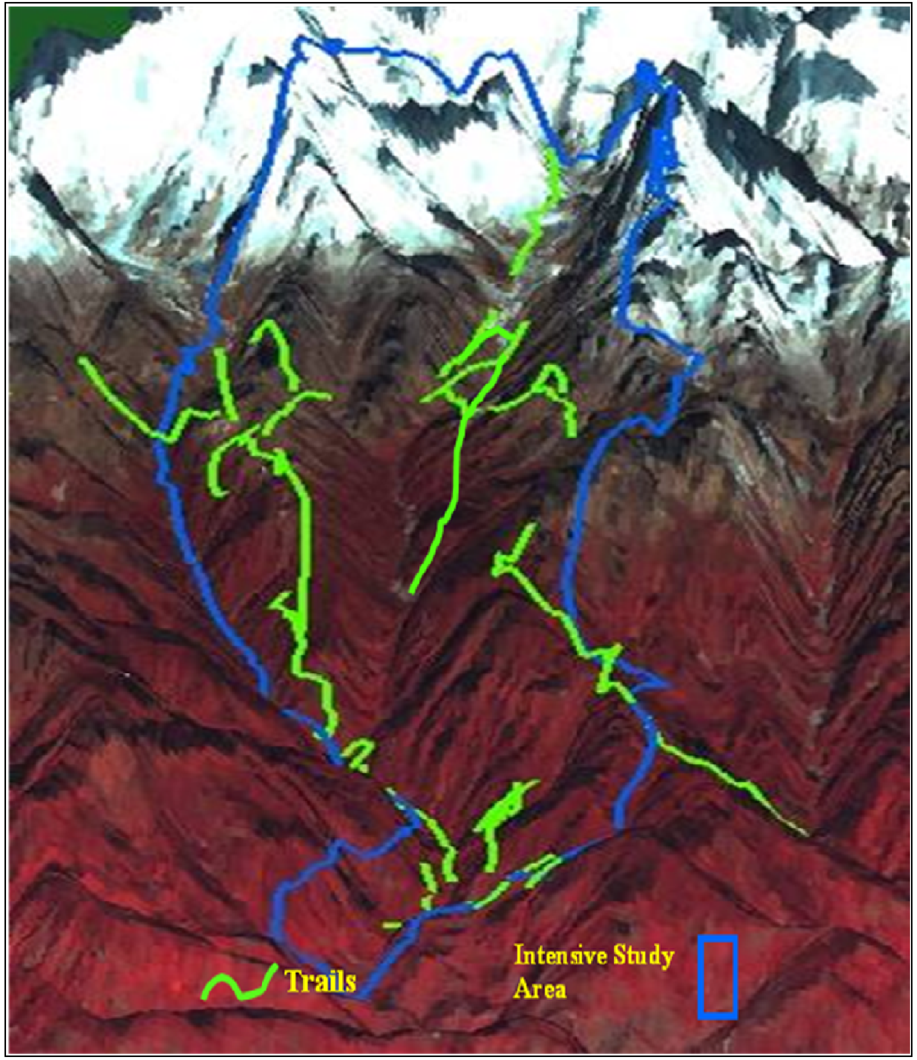


**Figure 3.2 Camera trap locations in *Prek chu* in 2×2 km grids**

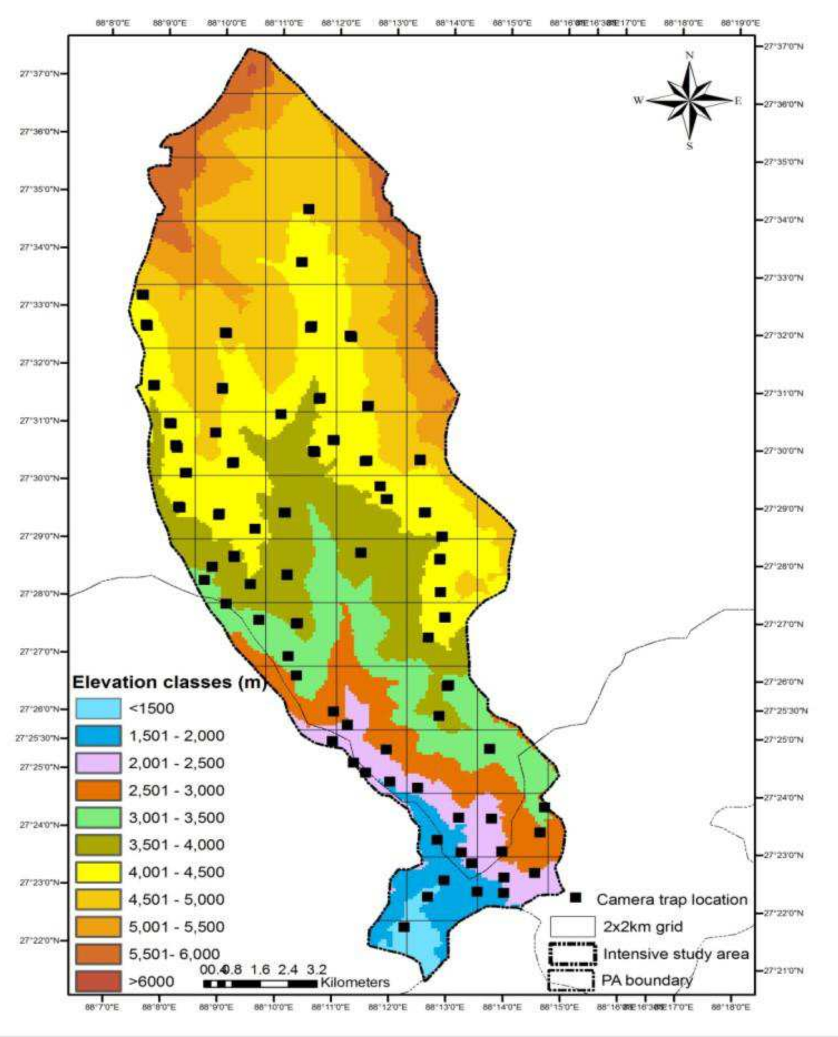


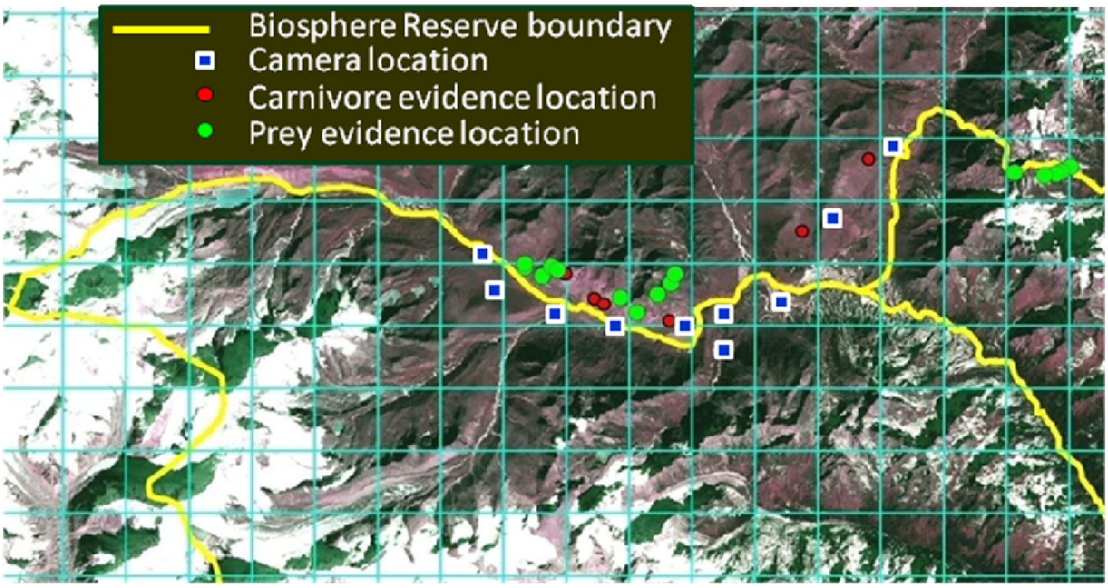
**Table 3.2 Camera locations and habitat characteristics for sampling *Prek chu* from 2009-2011 (CD, E, S, A = camera days, elevation, slope and aspect**

| S. No | Habitat | Location | CD | E (m) | S (°) | A |
|---|---|---|---|---|---|---|
| 1 | Alpine | Laxhmipokhri | 15 | 4400 | 20 | E |
| 2 | | Thansing top | 196 | 4300 | 30 | NW |
| 3 | | Lampokhri | 348 | 4350 | 15 | SW |
| 4 | | Dallepokhri | 62 | 4250 | 20 | S |
| 5 | | Weaselpoint | 259 | 4500 | 30 | W |
| 6 | | Somiti | 156 | 4300 | 20 | NE |
| 7 | | Base Camp | 139 | 4300 | 20 | S |
| 8 | | Paireni | 17 | 4250 | 30 | SW |
| 9 | | Chongrigang | 228 | 4210 | 30 | SW |
| 10 | | Jaireni | 45 | 4200 | 30 | NE |
| 11 | | Deorali | 57 | 4000 | 30 | NE |
| 12 | | Deotaghar | 77 | 4100 | 15 | NE |
| 13 | | Gochella | 40 | 4500 | 35 | NW |
| 14 | | Ghinghe | 74 | 4450 | 35 | SW |
| 15 | | Aralungchok 1 | 63 | 4300 | 35 | SW |
| 16 | | Aralungchok 2 | 58 | 4400 | 30 | W |
| 17 | | Chamre | 83 | 4500 | 20 | SW |
| 18 | | Neer Pokhri | 17 | 4250 | 20 | SW |
| 19 | | Tenzing Base | 27 | 4300 | 30 | W |
| | | **Sub-total** | **1961** | | | |
| 20 | Subalpine | Jamling | 321 | 3700 | 45 | W |
| 21 | | Ghunsa | 352 | 3100 | 30 | SW |
| 22 | | Kasturi Ridge | 376 | 3000 | 40 | SE |
| 23 | | Phedang | 382 | 3700 | 35 | W |
| 24 | | Japuley | 64 | 3800 | 20 | E |
| 25 | | Sukhapokhri | 86 | 3900 | 20 | SW |
| 26 | | Phedhi | 373 | 3850 | 35 | E |
| 27 | | Chilime Tar | 70 | 3800 | 35 | E |
| 28 | | Pokhri Tal | 66 | 3950 | 40 | E |
| 29 | | Kockchurong | 234 | 3700 | 30 | NE |
| 30 | | Kokchurong 2 | 56 | 3700 | 35 | SE |
| 31 | | Kasturi Top | 375 | 3400 | 40 | E |
| 32 | | Phedi Top | 332 | 3990 | 40 | E |
| | | **Sub-total** | **3087** | | | |
| 33 | Temperate | Jamathang | 43 | 2000 | 45 | W |
| 34 | | Malingo | 189 | 2300 | 40 | SW |
| 35 | | Sachen | 280 | 2250 | 40 | SW |
| 36 | | Gaikhurey | 54 | 2700 | 45 | S |
| 37 | | Tingsu | 152 | 2750 | 45 | SW |
| 38 | | Tsuse | 17 | 2000 | 35 | W |
| 39 | | Tirche Bato | 38 | 2000 | 40 | E |
| 40 | | Dubdi | 201 | 2100 | 30 | W |
| 41 | | Intake | 187 | 1700 | 40 | SW |
| 42 | | Tingkongkhola | 41 | 1750 | 40 | W |
| 43 | | Pakhola | 276 | 1950 | 30 | SW |
| 44 | | Quapani | 156 | 1950 | 40 | W |
| 45 | | Khengtey | 31 | 2000 | 30 | SW |
| 46 | | Lamphey | 197 | 2100 | 45 | SW |
| | | **Sub-total** | **1862** | | | |
| | | **Total** | **6910** | | | |

**Table 3.3 Camera locations and habitat characteristics used for sampling *Lhonak* c*hu* catchment during 2012 (CD, E, S, A = camera days, elevation, slope and aspect**

| S. No | Habitat | Location | CD | E (m) | S (°) | A |
|---|---|---|---|---|---|---|
| 1 | Alpine | Munguthang 1 | 23 | 4600 | 15 | SW |
| 2 | | Munguthang 2 | 22 | 4520 | 20 | SE |
| 3 | | Munguthang 3 | 23 | 4580 | 10 | NE |
| 4 | | 20 R flat | 21 | 4650 | 30 | N |
| 5 | | 20 R ridge | 22 | 4760 | 40 | E |
| 6 | | 20 R top | 12 | 4810 | 20 | SW |
| 7 | | Janak 1 | 21 | 4740 | 25 | NW |
| 8 | | Janak 2 | 20 | 4750 | 20 | N |
| 9 | | Janak 3 | 21 | 4740 | 15 | N |
| 10 | | Janak 4 | 20 | 4790 | 25 | NE |
| | | **Total** | **205** | | | |

**Figure 3.3 Camera trap locations in *Lhonak* catchment in a 2×2 km grid design**



## 3.6 SCAT COLLECTION AND IDENTIFICATION

Scats encountered during trail monitoring and sign survey were opportunistically collected, identified and categorized into different species based on their shape, size, diameter (Chame 2003; Bang and Dahlstrom 2006) and presence of ancillary evidences such as pugmarks, scrapes and rake marks. Measurements of scat size were done using a digital vernier calliper (Model = AEROSPACE). Comparison among scat morphometrics of carnivore species are given in Table 3.4.

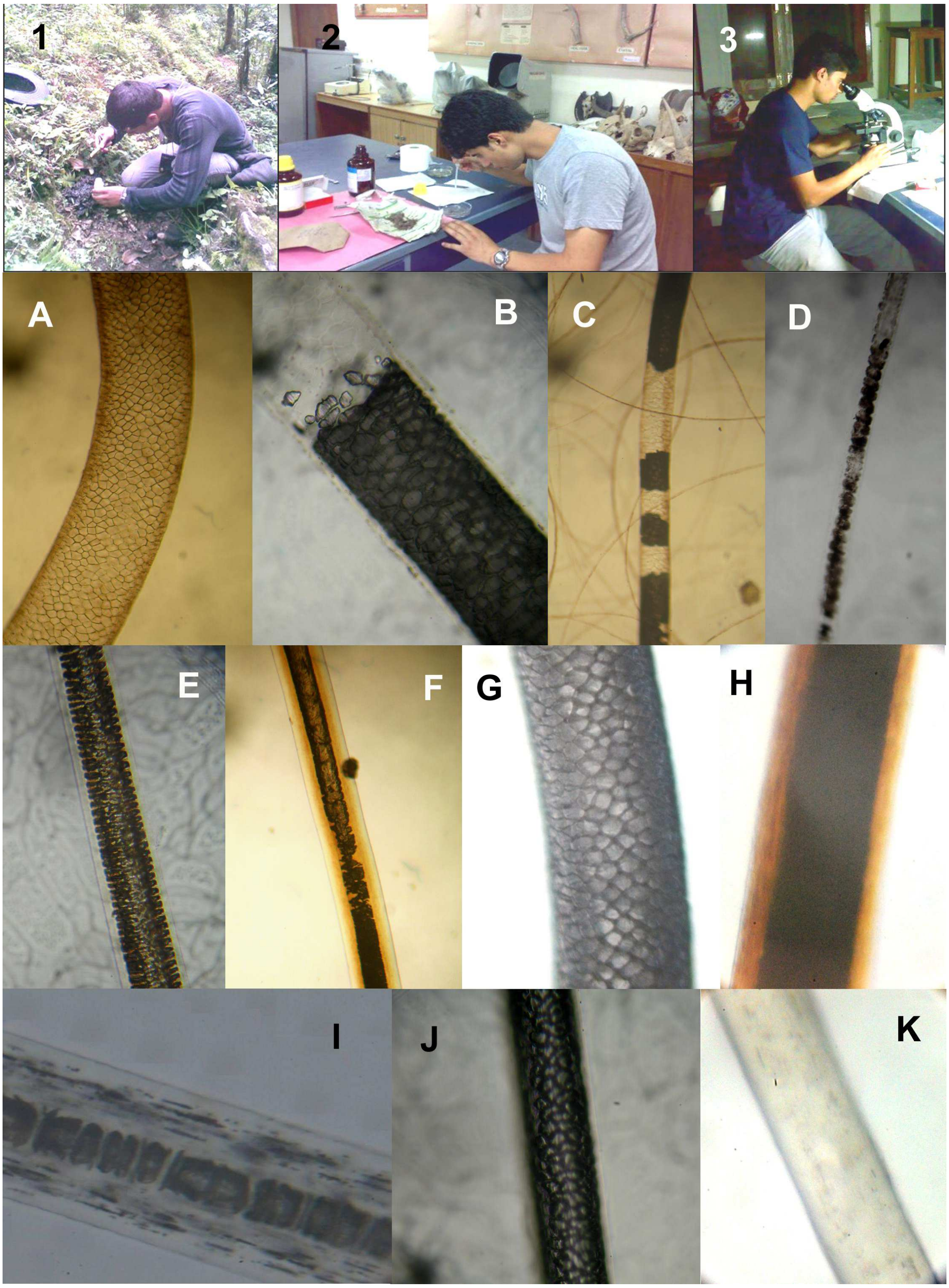
**Table 3.4 Morphological characteristics and diameters of scats of different carnivore species for identification, N = number of scats of each species.**

| **Species** | **N** | **Scat characteristics** | **Literature** | **Scat diameters (cm)** |
|---|---|---|---|---|
| Snow leopard | 114 | compact, cylindrical, sausage-shaped, sub-divided and tapered at one of the extremities | Jackson et al. 2005; Bagchi & Mishra 2006 | 2.48-3.82 |
| Red fox | 367 | cylindrical, smaller in size than snow leopard, sub-divided (generally in two parts), with blunt ends | Lanszki et al. 2007; Aryal et al. 2010 | 1.68-2.04 |
| Wild dog | 41 | clustered, cylindrical, sub-divided with blunt ends, defecated in the middle of trail | Johnsingh 1983; Acharya 2007 | 2.38-3.27 |
| Yellow-throated marten | 105 | narrower, flattened, generally single units without sub-divisions | Tatara & Doi 1994; Zhou et al. 2011 | 0.64-1.32 |
| Leopard cat | 37 | cylindrical, sausage-shaped, sub-divided and tapered at one of the extremities | Rabinowitz 1990; Tatara & Doi 1994; Grassman 1997 | 1.02-1.54 |
| Jackal | 24 | cylindrical, sub-divided with blunt ends | Vanak & Mukherjee 2008; Nadeem et al. 2012 | 1.63-2.18 |
| Black bear | 7 | hard and cylindrical when feeding on acorns | Koji Yamazaki *pres. Comm*; Robert Steinmetz *pers. comm* | 4-5 |

## 3.7 LABORATORY METHODS

Food habits of carnivores were assessed through scat analysis (Putman 1984). All collected scat samples were sun-dried in the field and later oven dried at 60°C in the laboratory (Sankar and Johnsingh 2002) for 48 h and preserved in tagged paper bags for further analysis. Prior to any further analysis, a reference key was developed for the identification of prey species on the basis of hair structure/morphology in the research laboratory of the Wildlife Institute of India, Dehradun. Each scat was carefully broken, soaked and washed with running water using fine-mesh (1.0 and 0.5 mm) filters (Juarez and Filho 2002), to separate the undigested prey remains, such as hairs, bones, hooves, mandibles, feathers, etc. Washed samples of hairs from the scats were oven dried and later a portion of these were kept in 70% alcohol over night (for dehydration) followed by a xylene treatment (to induce transparency) for 30 minutes (Koppikar and Sabnis 1976). The hair samples were then mounted on glass slide using a DPX mountant (mixture of distyrene, a plasticizer, and xylene [DPX] used as a synthetic resin mounting media) and examined under microscope with 10X and 40X resolution. At least 20 hairs were examined from each scat (Mukherjee et al. 1994) and prey species were identified by comparing with the reference key using features such as colour, length, thickness, characteristic medullar configurations (Koppikar and Sabnis 1976; Mukherjee et al. 1994; Reynolds and Aebischer 1991) and cortex to medulla ratio (Plate 3.1). Each scat sample was analysed in this manner and the information on undigested prey items in the scats and hence the corresponding prey species consumed were recorded.

**Plate 3.1 (1) Showing Scat collection, (2) Chemical treatment and slide preparation, and (3) Observations of hair characteristics of (A) Himalayan tahr (B) Goral (C) Blue sheep (D) Rodent (E) Pika (F) Serow (G) Musk deer (H) Dzo (I) Macaque (J) Barking deer and (K) Langur under microscope at 40× resolution.**



## Chapter 4

# DISTRIBUTION AND ABUNDANCE

### 4.1 INTRODUCTION

From the last decade of the previous century, the need to document and comprehend patterns of biological diversity has gained importance as the prevailing and potential threats to biological diversity have become more extensively recognized (Soulé 1986; Wilson 1988; Quammen 1996). Contrary to this, less attention has been given to variation in species richness in non-fragmented habitats especially along the elevational gradients in high mountains such as Himalaya, the reason being the remoteness of the area and difficult terrains to work upon (Schaller 1977). Since, carnivores belong to the most vulnerable group of mammals (Karanth and Chellam 2009), it becomes crucial to understand and monitor their distribution pattern and status from one of the world’s most diverse and fragile ecosystems such as the Himalayas.

Monitoring of populations may be performed at various levels of resolution. The primary question to be asked is where a particular species occurs (species distribution), and secondly the interest is in knowing how many animals are there in an area i.e., species abundance (Gese 2001). Since, viability of a species can be understood only in the context of ecological interactions with other species at multiple spatial and temporal scales (Chen et al. 2005), monitoring multiple species and their interactions at multiple scales is critical for any useful habitat conservation plans and adaptive ecosystem management programs. A multi-species monitoring approach thus provides a better understanding of the overall status of the wildlife or the target group of species in an area required for developing comprehensive long-term management and habitat conservation plans.

The Khangchendzonga BR being positioned at the convergence of two biogeographic realms, *viz.*, Palaearctic and Oriental (Mani 1974; Corbet 1978; Corbet and Hill 1992), the area has a high potential of sustaining many primitive as well as newly evolved species of carnivores. Spear-headed by big cats, this group may include members from canids, ursids, mustelids and civets. Among felids the endangered snow leopard is distributed in all five Himalayan states of India including Sikkim (Prater 1971; Anonymous 2008) up to altitudes of 5,400 m in Ladakh (Mallon 1984). Presence of species such as, Eurasian lynx and pallas's cat which are known to occur in the trans-Himalayan habitat of Ladakh have also been reported from the *Tso Lhamo* Plateau in north Sikkim (Chanchani 2007). Other medium to small cat species including clouded leopard, Asiatic golden cat, leopard cat and marbled cat are well distributed in the adjacent north-eastern states (Mishra et al. 2006; Choudhury 2007; Datta et al. 2008a), hence have a high possibility of occurrence in KBR as well. Common leopard being one of the most widely distributed felid across India occurs in lower elevations of Sikkim. Presence of five canid species are expected from KBR including the most common red fox, and the golden jackal which is common in lower elevations of the Sikkim Himalaya. Among the trans-Himalayan species presence of Tibetan sand fox and Tibetan wolf has been reported from the *Tso Lhamo* plateau of north Sikkim (Chanchani 2007), although the information on Tibetan wolf is merely based on interviews with local herders, hence needs further confirmation. In addition, presence of two species of endangered dhole has been mentioned in one of the oldest documentations of Sikkim biodiversity- the Sikhim Gazetteer (Gammie 1894). This species has been reported to occur even from the high altitudes of Ladakh, adjacent north-eastern states and Bhutan (Fox and Chundawat 1995; Johnsingh et al. 2007; Durbin et al. 2008), besides it wide distribution throughout much of India– south of the river Ganges. The Asiatic black bear is distributed from Kashmir to Arunachal Pradesh across the Himalaya including Sikkim (Sathyakumar and Choudhury 2007). Red panda being the State animal of Sikkim is known to have an overall disjunct distributed from Nepal in the west to China in the east. In eastern Himalaya it is known to occur in Darjeeling, Sikkim and Arunachal Pradesh (Pradhan et al. 2001). Mustellids and viverid diversity of Sikkim is known to include seven species including yellow-throated marten, Siberian weasel, European otter, large Indian civet, masked palm civet, spotted linsang and binturong (Anonymous 1989), but all these records are from areas outside KBR; hence, the information on their status in KBR needs to be updated.

Khangchendzonga landscape being very remote, steep and densely vegetated, the conventional sampling designs for estimating abundance of carnivore populations may be challenged. A potential applicability of camera trap technique being non-intrusive and applicable over large areas with relatively moderate effort (Silveira et al. 2003) has reduced sampling discrepancies between habitats and observers (Cutler and Swann 1999; Silveira et al. 2003; Swann et al. 2004) and considerably advanced the ability to study elusive animals (Kays and Slauson 2008). With the background of almost no reliable documentation, I specifically aimed at answering below mentioned questions, thus attempting to establish information on the diversity, distribution and abundance of carnivores in KBR.

1. Which carnivore species occur in Khangchendzonga BR?
2. What are the spatial patterns of carnivore species distribution with respect to different habitats and elevation classes in the *Prek chu* catchment?
3. What is the population status of carnivore species in the *Prek chu* catchment?

## 4.2 METHODS AND DATA ANALYSIS

### 4.2.1 Carnivore species occurrence in KBR

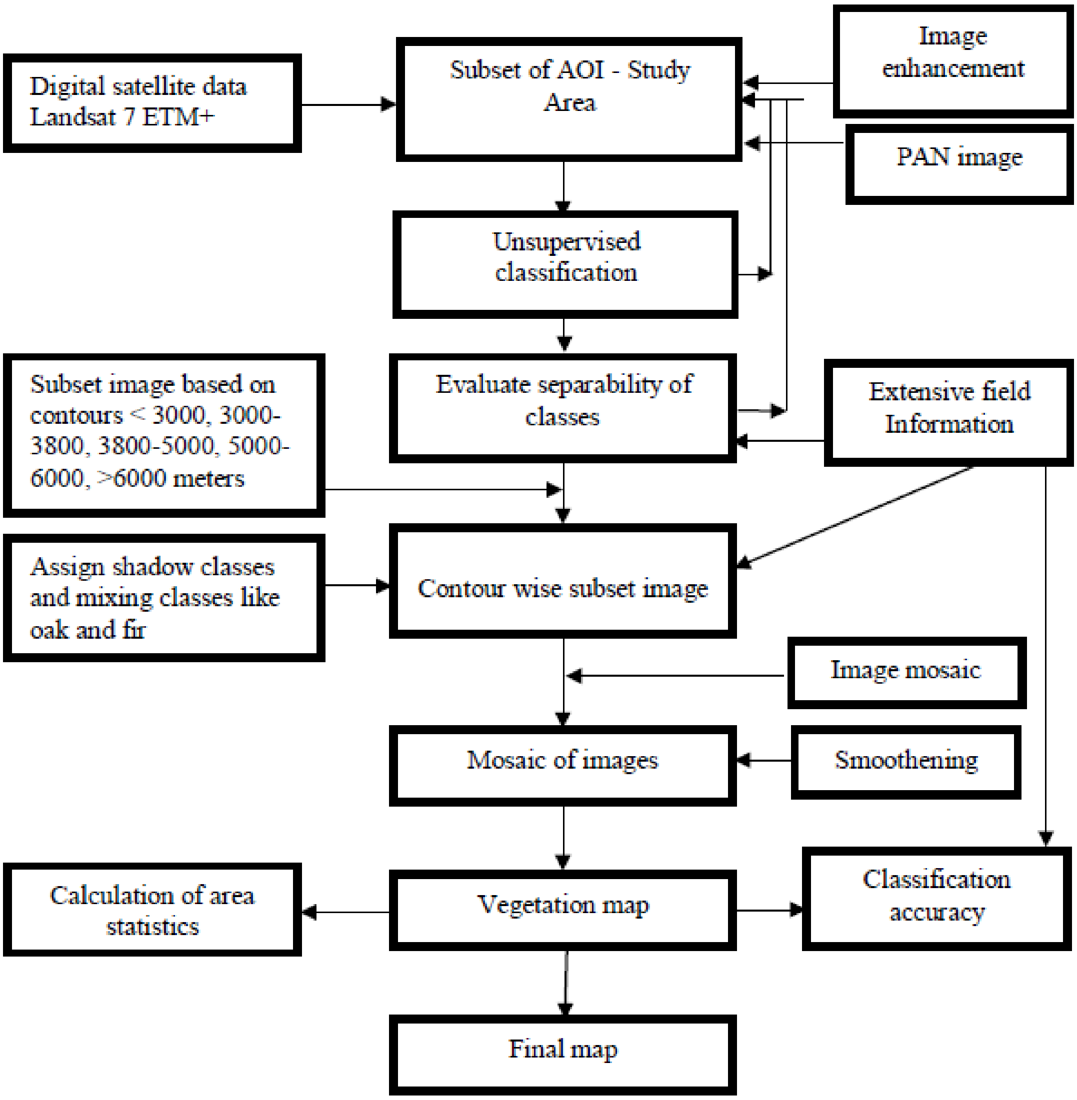
Indirect evidence of carnivores (scats, pugmarks and tracks) encountered during trail monitoring and sign surveys were primarily categorized into different groups (e.g., felids, canids and mustellids) based on their general characteristics (size, shape, and substrate) and habitats of occurrence (in literature). Moreover, collected scat samples were further identified through consultations with species expert (Sathyakumar *pers comm*; Goyal *pers comm*) and by comparing with different field guides (e.g., Bang and Dahlstorm 2006). All camera trap photographs of carnivores were carefully examined and species identification was done by comparing with species descriptions in different field guides (Prater 1971; Macdonald 2001; Menon 2003). In case of any ambiguity, consultations were also done with experts on Himalayan fauna (Sathyakumar *pers comm;* Goyal *pers comm;* Habib *pers comm*).

### 4.2.2 Distribution pattern of carnivores in *Prek chu* catchment

The boundary of KBR was digitized form the geo-referenced map procured from the DFEWM, Government of Sikkim in ERDAS IMAGINE 8.5 digital image processing software. Various features such as glaciers, lakes, rivers and watersheds were digitized from the Survey of India topographic map (1:25000 scale). The boundary of the intensive study area was also digitized. The Digital Elevation Map (DEM) was downloaded from the CGIAR-CSI GeoPortal (Jarvis et al. 2006). From this DEM, using the spatial analyst extension of Arc GIS 9.2, elevation, aspect and slope class maps for KBR and then for intensive study area (through extraction by masking) were prepared. Vegetation classification map for the entire KBR was prepared through the hybrid classification approach following Tambe (2007), [Figure 4.1] and for intensive study area using extraction tool of Arc GIS 9.2 from the classified map of KBR. Major habitat classes were determined by dissolving the minor vegetation classes with the broad habitat categories. Vegetation maps represented four major vegetation classes (temperate, sub-alpine, *krummholdtz* and alpine) and three non-forest classes (rocky area, snow and water bodies).

The vegetation class map of the intensive study area was divided into 1×1 $km^2$ grids, and all locations of carnivore evidences (sightings, signs, and camera photos) were plotted on this grid map. Relative frequencies of locations in each grid for each species were calculated following the simple equation i.e., $F = (n/A) \times 100$, where F is the relative frequency in percentage, $n$ is the number of locations in a particular grid, and $A$ is the total number of locations plotted for the species. The relative frequency of locations for each species were then joined with the grid map using the joining option of Arc GIS 9.2 and the grid map was classified as low (1-30%), medium (31-60%) and high (61-100%) and projected over the vegetation classification map to show the spatial distribution pattern of each carnivore species with respect to the corresponding vegetation classes. Important place locations (camping areas, villages, lakes) were also shown in these maps for easy referencing. Villages such as *Yuksam* (just outside KNP) and *Tsoka* (inside KNP), tourist huts such as *Sachen* and *Kasturi*, lakes such as *Somiti* and *Lampokhri* and tourist and mountaineers destinations such as *Basecamp* and *Dzongri* were also marked on these maps.

**Figure 4.1 Image processing and vegetation classification of the satellite data (after Tambe 2007)**



### 4.2.3 Population status of carnivores in *Prek* c*hu* catchment

Animal abundance can be assessed in two ways *viz.*, relative and absolute abundance. Relative abundance uses indices of animal abundance (e.g., tracking rates, faeces deposition, capture rates, or visual observations) that can be compared over time or between areas, but of itself does not estimate animal numbers. In contrast, absolute abundance involves using methods to actually count animals and then estimate the number or density of animals in the population (Gese 2001). Moreover, the incorporation of an estimate of detection probability designates the robustness of population estimate over a population index (O'Brien 2011).

#### *4.2.3.1 Abundance estimation through conventional methods*

The sign encounter rate (#/100 km walk) was estimated using the formula ER= (n/L) × 100; where 'n' is the total number of signs belonging to a species encountered during each trail monitoring and 'L' is the length (km) of the trail. Average sign encounter rate for each species on a trail was calculated by pooling the number of signs encountered from all the repeats of the trail divided by the total length walked in all the repeats of the trail. Mean sign encounter rate of each species for each habitat and overall sign encounter rate was also calculated. Kruskal-Wallis $\chi^2$ tests and Mann-Whitney U tests were applied to test the differences among the estimates in different habitats.

#### *4.2.3.2 Abundance estimations through camera trapping*

##### *4.2.3.2.1 Photographic and Photo-capture rates*

Photographic rate is defined as the number of camera days (24 h) per study species photograph summed across all camera traps in the study (Carbone et al. 2001). Based on photo captures, an index of relative abundance (RAI 1) as the number of days required for obtaining a photo capture of a species (Carbone et al. 2001) was calculated. Photo capture rate (RAI 2) was also calculated as the number of photographs of a species divided by the number of trap days per site and was expressed per 100 trap days (Kawanishi et al. 1999; O'Brien et al. 2003). Trap-days were computed as the number of 24-h periods from deployment of camera until the film/memory card was used up, or battery lasted or the camera was retrieved. Instances where the same species were captured by the same camera more than once within 1 h were excluded from trap rate calculation (Bowkett et al. 2007). This was a compromise between scoring the same individual multiple times and missing individuals (Rovero et al. 2005) and is more conservative than other published studies (e.g., Kinnaird et al. 2003). Kruskal-Wallis $\chi^2$ tests and Mann-Whitney U tests were applied according to the data distributions to test any significant differences among the estimates of photo-capture rates of species in different habitats.

##### *4.2.3.2.2 Mark recapture based abundance estimation*

Karanth (1995) demonstrated the use of capture-recapture technique for estimating population size of tigers based on camera trap photographs. Potential applicability of this method for estimating abundance and density of other secretive animals with distinct and individually identifiable spot/stripe patterns (natural markings), was also suggested (Karanth 1995; Karanth and Nichols 1998). Among the carnivore species recorded in camera traps during present study; only four species belonged to the suggested group of animals having natural markings as coat patterns on the body (i.e., common leopard, snow leopard, leopard cat and large Indian civet). Limited captures of common leopard declined the possibility of its abundance estimation. Estimation of snow leopard abundance based on photographic captures and recaptures has been successfully demonstrated earlier (Jackson et al. 2006), but no such study has been conducted on leopard cat and large India civet, though Cheyne and Macdonald (2011) had indicated its scope for leopard cats. Hence, prior to any further analysis it was necessary to evaluate the reliability of identifying individuals of both these species from their spot/stripe patterns.

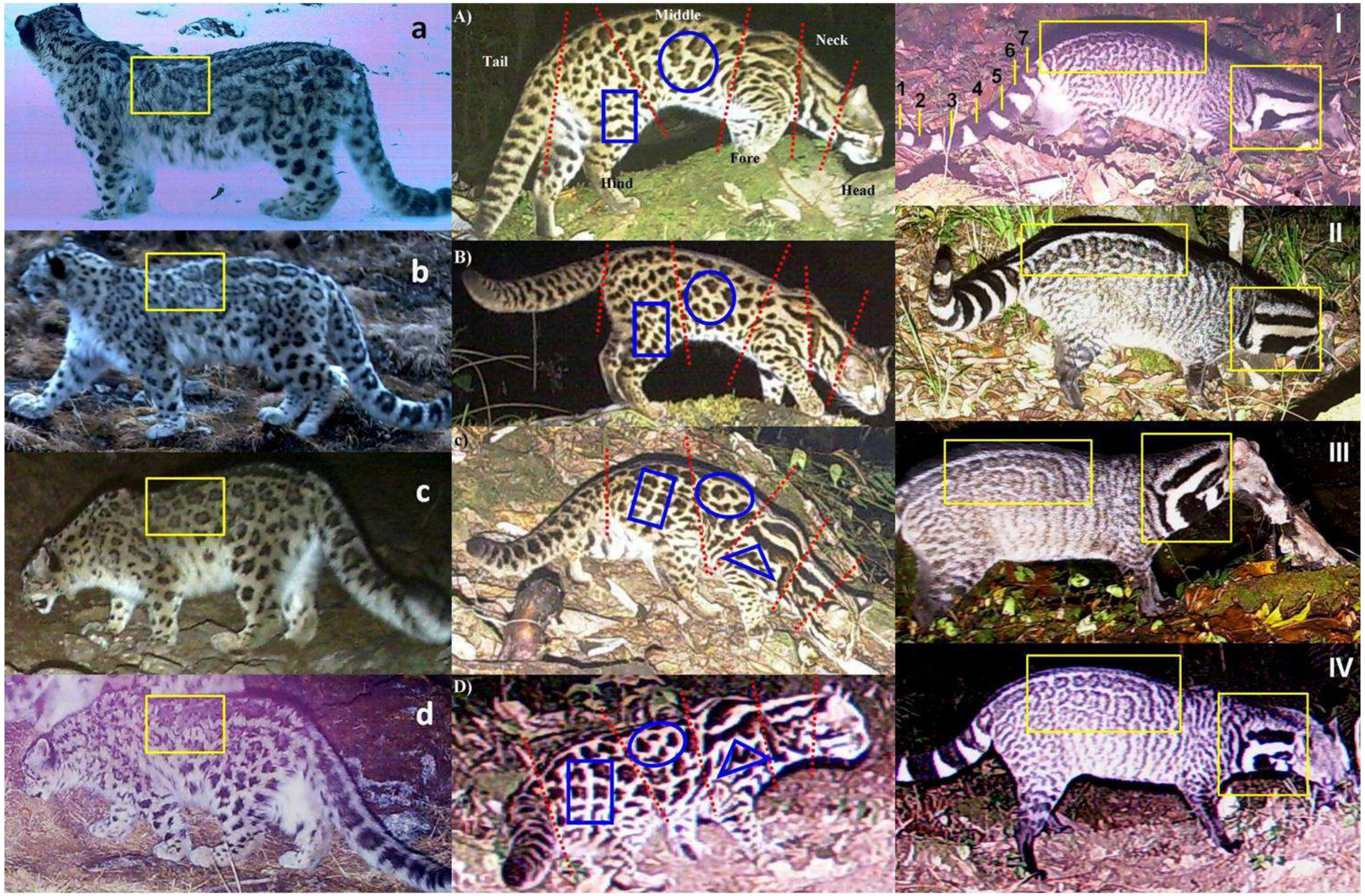
Double-blind observer identification protocol designed on pumas (*Puma concolor* - Kelly et al. 2008) and later applied on striped hyenas (*Hyaena hyaena* - Harihar et al. 2010) was thus used for this purpose. All photographs of both species were numbered with ID’s and made into three sets and distributed among three investigators (research scholars working in the project). Each investigator independently identified individuals from features such as dots, stripes, tail rings and other marks using different body parts (head, neck/throat, fore, mid, and hind-quarters and tail) (Plate 4.1). Investigators sorted photographs into positive identifications, tentative identifications, and not possible to identify, based on separate visibility of body parts in photographs. Proportion of exact matches across investigators which resulted from using different body parts (usability) based on the quality of photographs was also assessed. Kruskal-Wallis test was also done to evaluate any differences in the use of various body parts by the investigators’ to assign individual identities to the photographs.

Estimation of abundance and density from camera trap based capture-recapture data of species with distinct pelage patterns, can be done following two different analytical methods, i.e., *ad hoc* (non-spatial) MMDM based method (Karanth 1995; Karanth and Nichols 1998) and the spatially-explicit capture recapture method (Efford 2004; Royle and Young 2008). Recent probabilistic capture-recapture estimators have advanced to model biologically important factors such as capture probabilities being heterogeneous among individual animals in a population as a result of social structure. Current capture-recapture models also permit incorporation of other biologically relevant factors, such as behavioural response to trapping or temporal variation in capture probabilities.

### A) Non-spatial estimation- theory

This method involves development of capture histories for each individual of a species identified from the photographic captures. The capture history for animal assumes a value of either ‘‘0,’’ if the animal was not photographed on a particular occasion, or ‘‘1,’’ if the animal was photographed on that occasion. Depending on the ecology and longevity of the study species demographic closure is assumed for a certain period designated as a sampling session. Furthermore, the subset of X matrix deduced for the sampling session period is analysed for population closure using Close-Test (Stanley and Burnham 1999) and subsequently estimation of abundance (N) and capture probability (p) with associated standard errors done using full closed capture-recapture model in program MARK ver. 6.1 (White and Burnham 2000) using the best selected model based on minimum AIC value. Density of the species is calculated by dividing the population size (N) by the effective sampled area, which is estimated by creating a minimum convex polygon (MCP) over the trapping stations and adding a buffer width equal to half the mean maximum distance moved (1/2 MMDM) by recaptured individual to the camera trap polygon (Karanth and Nichols 1998). Capture–recapture sampling is a widely used method of estimating population size, but the spatial nature of capture-recapture data is usually neglected in analysis, which is well taken care in spatially-explicit method.

**Plate 4.1 Individual identification based on coat patterns of snow leopards (a, b, c & d- represent different individuals), leopard cat (A, B, C- represent different individuals while C & D represent same individual) and large Indian civet (I, II, III- represent different individuals while III & IV represent the same individual).**



### B) Spatially explicit CR estimation- theory

A natural framework for developing spatial models of abundance is based on point process models, and point processes have been considered as the basis for spatial capture-recapture models in a number of recent efforts (Efford 2004; Borchers and Efford 2008; Royle and Young 2008). To develop this notion, it was presumed that each individual in the population has a fixed point associated with it, its centre of activity, *Si* = (*S1i, S2i*), a two-dimensional coordinate representing a point in space about which the movements of individual *i* are concentrated. It was supposed that these point locations represent the realization of a binomial point process. That is, we suppose there exists a population of N independent centers *Si*, *i* = 1, 2, . . . , N, distributed uniformly over some region, say S, the state-space of the point process. In practice, S can be prescribed (e.g., by specifying coordinates of some polygon that contains a trapping array). The basic inference problem is to obtain an estimate of density, the number of activity centers per unit area of S, which is equivalent to an estimate of N under the point process model.

Spatially explicit capture-recapture (SECR) models hence make use of the spatial location of captures in order to first determine an individual's activity centre and then to estimate the density of activity centres across a precisely defined polygon containing the trap array (Gardner et al. 2009; Royle et al. 2009 a) and hence avoid the problem of estimating the effective area sampled. SECR methods address the uncertain edge effects and spatially heterogeneous detection probability caused by movement in conventional animal trapping (Efford 2004; Borchers and Efford 2008). Meanwhile, two different calculation techniques (Maximum likelihood and Bayesian) to estimate SECR density are available.

The *maximum likelihood-based method* using program DENSITY 4.4.1.2 (Efford 2007) directly estimates density by fitting spatial detection functions to capture–recapture data from arrays of passive detectors such as camera traps (Efford 2004) where the probability density functions for detections of animals based on distance from activity centres are modeled using hazard rate, half-normal or exponential detection functions (Efford et al. 2008). These models can be viewed as mixture models in which the mixture is over the distribution of animal locations. Estimators are based on the marginal distribution obtained by integrating the joint likelihood over the distribution of the unobserved locations taking explicit account of the spatial nature of the sampling process (Borchers and Efford 2008). Maximum likelihood-density (MLDens) can hence be estimated using the estimator that best explained the individual capture probability in MARK and based on minimum model AIC value.

The *Bayesian approach* based on data augmentation (Royle et al. 2007) is known to provide valid inference even with small sample sizes (Sollmann et al. 2011). Here it is presumed that trap- and individual specific capture probabilities are a function of distance between individual home range centers and trap locations. Using data augmentation, the spatial capture-recapture models are formulated as zero-inflated Poisson or binomial regression models with random effects (Royle et al. 2009 b). This method first involves fixing of a subset area (based on the distribution and ecology of the species) in which the activity centers will be assumed to be distributed. To implement data augmentation, the *n* observed encounter histories are physically supplement the with some large number of ‘‘all-zero’’ histories, say *M-n* such histories (Royle et al. 2007). It was assumed that this *M* includes the actual N individuals in the population as a subset and hence *M* is chosen sufficiently large (larger than the largest possible population size in the area) so that the posterior of N is not truncated. This reformulation of the model based on data augmentation is a zero-inflated binomial mixture and the number of activity centres N in S is then estimated as a fraction of M. For model analysis, we implemented a Markov chain Monte Carlo (MCMC) simulation method which estimates the joint posterior distribution of the unknown quantities in a statistical model in software package SPACECAP 1.0.3 beta (Gopalaswamy et al. 2011) assessed through program R (version 2.14.1). MCMC chains are started at arbitrary parameter values and successive iterations depend on the outcome of the previous iteration, the beginning values are hence reflected in a number of initial iterations which need to be discarded (the burn-in process).

### C) Abundance estimation of snow leopard

For estimating the population size of snow leopard in the *Prek Chu* catchment, intensive camera trapping was done for a sampling session of about four months (17th January-22nd May, 2010; 126 days) to ensure geographical and demographic closure. In total, 27 camera traps were deployed covering all accessible grids (2×2 km) of the alpine and rocky zone (4,000 to 5,000 m elevation; Figure 4.2). Accessibility of the grids being a limiting factor, camera traps were deployed to warrant minimum possible inter-trap distance based on the minimum home range size recorded for the species, which is a prerequisite for non-spatial capture re-capture models (Karanth and Nichols 1998) but not for SECR models. The entire sampling session was divided into 18 occasions of 7 days each and tested for closure assumption using Close-Test. Non-spatial estimation of abundance (N) and capture probability (p) was done using full closed capture-recapture model in program MARK ver. 6.1. Density estimation was done by dividing the abundance estimate (N) by the effective sampled area. For calculating the effective sampled area a minimum convex polygon (MCP = 70.44 km²) was created over the trapping locations and a buffer width equal to half mean maximum distance moved (½ MMDM) by a recaptured individual was added over the trap polygon. The maximum likelihood-based density estimation was done in program DENSITY 4.4.1.2 using hazard rate, half-normal or exponential detection functions and the best model was selected based on minimum ML AICc value. Density estimation based on data augmentation in a Bayesian framework was done by creating 15 km buffer around the MCP of the trap array and the area that was judged as non-habitat (below 4,000 m and above 6,000 m) was excluded using the spatial analyst tool of Arc GIS 9.3. This buffered region was described by a grid of 5201 equally spaced points, each representing an area of 0.25 km² within the buffer area of 1300.25 km². Of these, 2219 points (554.75 km²) were judged to represent suitable habitat. The activity centres were therefore assumed to be uniformly distributed over this discrete space of 2219 points, an area of approximately 554.75 km². For model analysis, a number of MCMC chains were run in software package SPACECAP, and finally the combination of values with 50,000 iterations, a burn-in of 5000 and a thinning rate of 100 and data augmentation value 10 times the individuals with known capture histories in a Bernoulli encounter model ensured the adequacy to characterize the posterior distributions. Predicted posterior pixel densities estimated for each activity centre were also plotted using the joining tool of Arc GIS 9.3.

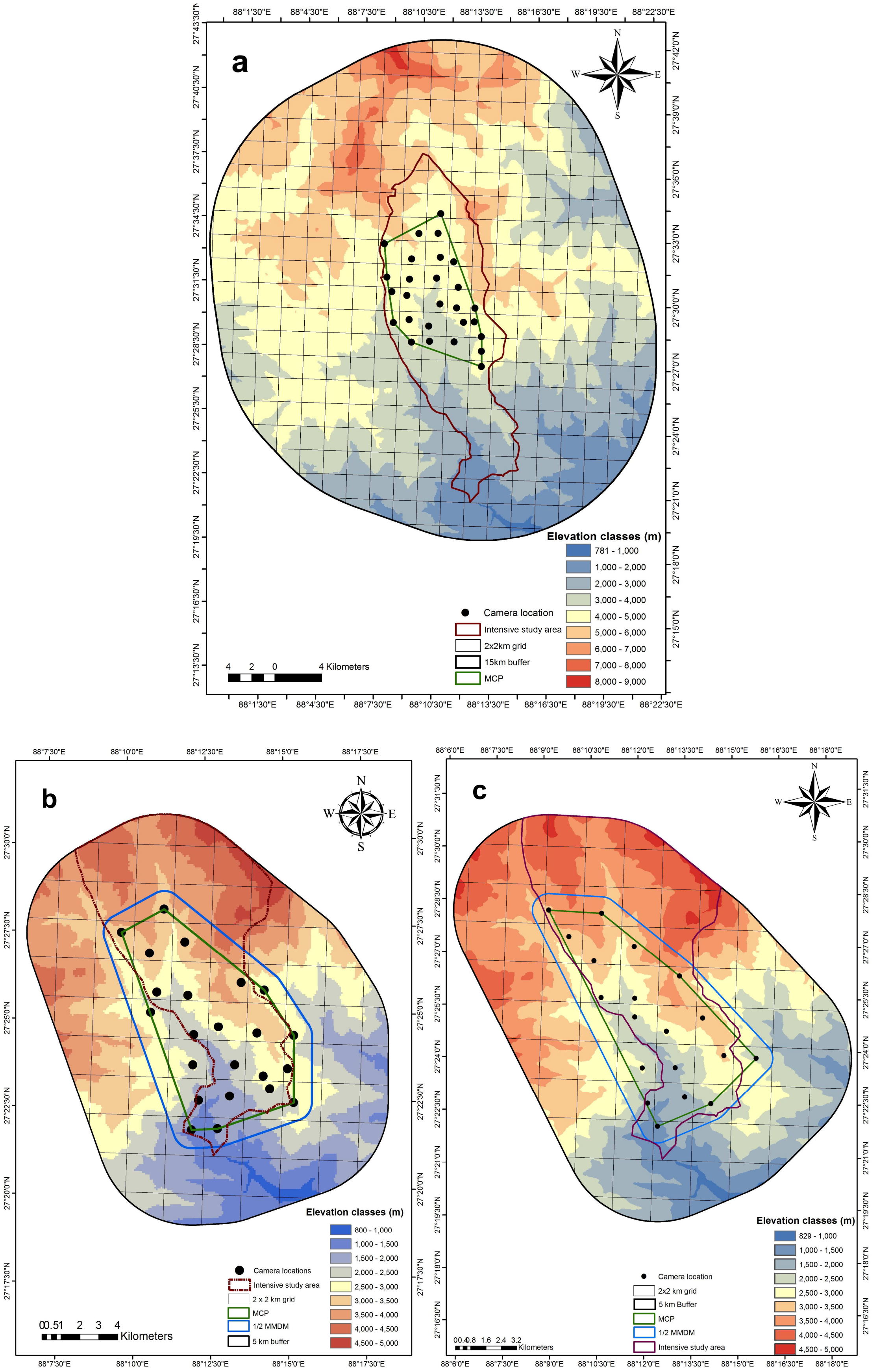
### D & E) Abundance estimation of leopard cat and large Indian civet

Similar, analytical procedures were used to estimate the non-spatial and spatially explicit densities of leopard cat and large Indian civet with appropriate modifications in the design, sampling period and analytical specifications as discussed below in Table 4.1.

**Table 4.1 Details of sampling design and analytical specifications used for estimation abundance and density of leopard cat and large Indian civet**

| **Analytical specifications** | **Leopard cat** | **Large Indian civet** |
|---|---|---|
| Sampling grid size | 2×2 km | 2×2 km |
| Number of camera units | 23 | 19 |
| Elevation range sampled | up to 3,500 | up to 3,500 |
| Sampling period | 1st Oct-30th Nov, 2009 | 17th Nov, 2009-14th Jan, 2010 |
| Sampling session | 60 days | 90 days |
| Each sampling occasion | 3 days | 3 days |
| MCP | 62.28 km² | 53.85 km² |
| Buffer outside MCP | 5 km | 5 km |
| Total buffer area | 302.94 km² | 296.64 km² |
| Suitable buffer area | 182.655 km² | 173.54 km² |
| Pixel size | 0.2025 km² | 0.2025 km² |
| MCMC iterations | 50,000 | 50,000 |
| Burn-in | 5,000 | 5,000 |
| Thinning rate | 100 | 100 |
| Data augmentation value | 10 times | 5 times |

## Figure 4.2 Camera trapping design in 2×2 km grids for estimating abundance of (a) snow leopard, (b) leopard cat, and (c) large Indian civet



### *4.2.3.2.3 Royle's Biometrics based abundance estimation*

Binomial mixture models have been projected for estimating abundance and detection probability from count data (Royle 2004; Kéry et al. 2005). These models do not essentially rely on individual identification for the estimation of detectability, making them potentially useful for species lacking distinguishing characteristics and appearing in camera traps (O'Brien 2011). These models hence estimate population size from temporally repeated counts of organisms without regard to unique identification of individuals. The key prerequisite is that count surveys (using camera traps or other methods) are replicated at a number of sampling locations over a period for which populations are closed (Royle and Nichols 2003; O' Brien 2011). If camera traps are deployed at a large number of very small, spatially distinct sampling units, then the photographic record of a single camera may be treated as a point count survey of local abundance replicated over *d* days.

Royle's repeated count model follows a sampling design wherein temporally replicated counts are collected at a number of locations in space. Under this model, site-specific abundance is viewed as a random effect, and the marginal likelihood of the counts is obtained by integrating the binomial likelihood for the observed counts over possible values of abundance for each site (Royle 2004). This model estimates the population size by assuming that the population sampled is demographically closed over the course of the surveys, the spatial distribution of the animals across the survey sites follows some kind of prior distribution, such as the Poisson distribution, and the probability of detecting *n* animals at a site represents a binomial trial (Bernoulli trial) of how many animals are actually at that site. In a nutshell, the Royle Count Model is a mixture of the Poisson and Binomial distributions, and the goal is to find the parameters that shape the Poisson and Binomial distributions in such as way that the results of the model will yield data that "match" the observed field data.

The Royle-count model assumes that each of the R sites in the survey design is home to some number of animals that can be modeled by a specified prior distribution like the Poisson represented as:

$$f_x = \frac{e^{-\lambda}\lambda^x}{x!}$$

The Poisson distribution has a single parameter, λ (lambda), the mean. In this case, lambda is the mean or average abundance across the R sites. But, our goal is to estimate the number of animals at each site (say N). Given N number of animals occur at a site, the Royle count model calculates the probability of observing lowercase *n* animals at that site with a Binomial probability.

The study design considered here is that in which animals are counted during $t = 1, 2, .., T$ sampling occasions and $i = 1, 2, ..., R$ locations (sites). Let $n_{it}$ be the number of distinct individuals counted at location *i*, in time *t.* It is assumed that population being sampled is closed so that the counts may be viewed as binomial random variables,

$$n_{it} \sim \text{Binomial}\,(Ni, p),$$

Where *Ni* is the number of individuals available for sampling (i.e., the population size at location *i*) and *p* is the detection (or capture) probability. Replicate samples at R locations yield R site-specific likelihoods conditional on $(N1, N2, ...., N_R)$ and *p*. The joint likelihood is represented as:

$$L(\{N_i\}, p \mid \{n_{it}\}) = \prod_{i=1}^{R} \left\{ \prod_{t=1}^{T} \binom{N_i}{n_{it}} p^{n_{it}} (1-p)^{N_i - n_{it}} \right\}.$$

This likelihood contains $R + 1$ unknowns, the *R* abundance parameters and a single detection probability parameter, *p*. Now, to deal with the instability problem (Carroll and Lombard 1985) we need to integrate the conditional likelihood for *p* and *N* against a beta prior distribution for *p* which is assumed to be poison as:

$$f(N; \lambda) = \frac{e^{-\lambda} \lambda^N}{N!}.$$

The estimation of abundance in Royle's repeated count model hence focuses on the integrated likelihood:

$$L(p, \theta \mid \{n_{it}\}) \\ = \prod_{i=1}^{R} \left\{ \sum_{N_i = \max_t n_{it}}^{\infty} \left( \prod_{t=1}^{T} \text{Bin}(n_{it}; N_i, p) \right) f(N_i; \theta) \right\}$$

Since, this method works well for species with non-distinct coat patterns, an attempt was made to estimate the abundance of few study species for which estimation could not be possible through mark capture-recapture. But, low sample size (few photo captures) and maintenance of population closure being the main constraint, model assumption could only be met for four study species *viz.,* red fox, yellow-throated marten, stoat and masked palm civet. Sampling was done in 2×2 km grids (sites) using camera traps (one in each grid). Based on the distribution of these species in the area separate survey designs in different periods of the year were adopted for each species (Table 4.2). Cumulative area of the sampled grids (separately for each species) was used to calculate the density of each species. Each survey lasted for a period of 60 days (to maintain population closure) consisting of 20 occasions of 3 days each.

**Table 4.2 Survey design and sampling period for estimating species abundance through Royle's repeated count method.**

| **Species** | **Areas sampled** | **Sites** | **Sampling period** |
|---|---|---|---|
| Red fox | Sub-alpine and Alpine | 19 | 1st June-30th July, 2009 |
| Yellow-throated marten | Temperate and Sub-alpine | 21 | 1st Nov-30th Dec, 2009 |
| Stoat | Sub-alpine and Alpine | 23 | 1st March-30th April, 2010 |
| Mask palm civet | Temperate | 19 | 1st Nov-30th Dec, 2009 |

Photographs of the study species captured during this specific grid based design were used to develop capture histories for each species. Following Royle (2004), capture history matrix for a species was constructed from its count (number of distinct individuals detected) at all sites across repeated sampling occasions. If two individuals of a species were detected at second site at say sampling occasion 3 then it was recorded as "2" and if not detected say at occasion 4 in site 5 it was recorded as "0" (Table 4.3).

**Table 4.3 Example of capture matrix used for Royle's repeated count model**

| Sites | Occasion 1 | Occasion 2 | Occasion 3 | Occasion 4 | Occasion 5 |
|---|---|---|---|---|---|
| **1** | 1 | 0 | 0 | 4 | 0 |
| **2** | 0 | 0 | 2 | 1 | 2 |
| **3** | 2 | 0 | 1 | 0 | 1 |
| **4** | 0 | 1 | 1 | 0 | 1 |
| **5** | 1 | 2 | 0 | 0 | 1 |
| **-** | 1 | 0 | 0 | 2 | 1 |
| **23** | 0 | 1 | 2 | 0 | 1 |

#### *4.2.3.3 Comparison among indices and estimates*

Since, one of the goals of this study was to assess the reliability of abundance indices as surrogates for true abundances in the eastern Himalaya. We tested the relationships among different monitoring methods by comparing the estimates for different species. Correlations between absolute abundance estimates (density) of species with their respective sign encounter rates and photo-capture rates were tested through linear regression and represented as scatter plots. Indices of species sign encounter rates and photo capture rates were also compared.

## 4.3 RESULTS

### 4.3.1 Carnivore species occurrence in KBR

Presence of 19 species of carnivores were confirmed from the Khangchendzonga BR during this study through combined efforts of trail monitoring, sign surveys, camera trapping and local interviews, including 11 species of global conservation significance (3 endangered, 4 vulnerable and 4 near threatened). Out of these, 9 species belonged to typical Himalayan fauna and were overall categorized to different groups of carnivores such as, felids (6 species), canids (4 species), ursids (1 species), ailurids (1 species), mustelids (4 species), and viverrids (3 species). Overall, 11 species were recorded from temperate, nine from subalpine and seven from alpine/trans-Himalayan habitats. Although, most of the species were recorded in camera traps from *Prek chu* catchment, presence of Tibetan wolf was confirmed from the trans-Himalayan part of north Sikkim (*Lhonak chu*) while Siberian weasel was confirmed only through sighting. Due to the elusive nature of study species very few sightings were recorded during the study period including snow leopard (1), leopard cat (2), yellow-throated marten (15), red fox (4), red panda (1), jackal (3), masked palm civet (1), stoat (1), Siberian weasel (1), and pale weasel (3). However, through signs (mostly scats), presence of only seven species of carnivores could be confirmed based on their deposition pattern, size, shape and habitat of occurrence. Local interviews within the *Prek chu* catchment validated the presence of all carnivore species except the Asiatic golden cat, large Indian civet and stoat, however presence of binturong was confirmed only through local interviews. Local interviews and trail monitorings in the other watersheds of KBR yielded evidences of presence of only few species of carnivores. Information on the presence of carnivore species in different catchments of KBR and their global conservation status is presented in Table 4.4.

In spite the rigorous camera trapping effort in *Prek chu* and *Lhonak chu* catchments, the study failed to record the presence of species such as spotted linsang, marbled cat and Chinese pangolin from the area, although the later was recorded in the forest adjacent to *Yuksam* village (KCC *pers comm.*). Moreover, no evidence of European otter was recorded from any stream of the study area. Although, the camera trapping was aimed at terrestrial carnivores, presence of two specifically arboreal species including clouded leopard and red panda was also recorded. Leopard known to be a common species was less common in the area and probably occurred in areas below the lowest altitudinal limit of the study area. In addition, the camera trapping interestingly reported the presence of only the melanistic form of golden cat in the area with no evidences of the common golden morph. Trail monitoring, sign surveys and camera trapping in *Lhonak chu* catchment revealed the presence of red fox, snow leopard and Himalayan marmot in addition to Tibetan wolf, but no evidence of other typical trans-Himalayan carnivores such as Eurasian lynx, pallas’s cat and sand fox, although local Dokpa herders and Indo-Tibet Border Police personnel reported the presence of wild dogs in this area.

**Table 4.4 Presence of different carnivore species in different watersheds of Khangchendzonga BR along with their IUCN status; + denotes presence, - denotes not encountered or not mentioned in local interview and * denotes typical Himalayan species.**

| **Species** | **IUCN status** | ***Churong chu*** | ***Prek chu*** | ***Lachen chu*** | ***Zema chu*** | ***Lhonak chu*** |
|---|---|---|---|---|---|---|
| *Snow leopard | EN | + | + | + | + | + |
| Clouded leopard | VU | + | + | | | |
| Common leopard | NT | + | + | + | | |
| Asiatic golden cat | NT | | + | | | |
| Jungle cat | LC | + | + | + | | |
| Leopard cat | LC | + | + | + | | |
| *Red fox | LC | + | + | + | + | + |
| *Tibetan wolf | EN | | | | | + |
| Wild dog | EN | | + | | | |
| Jackal | LC | + | + | | | |
| *Asiatic black bear | VU | + | + | + | | |
| *Red panda | VU | + | + | + | | |
| *Yellow-throated marten | LC | + | + | + | + | |
| *Stoat | LC | + | + | | | |
| *Pale weasel | NT | | + | | | |
| Siberian weasel | LC | | + | | | |
| *Mask palm civet | LC | + | + | | | |
| Large Indian civet | NT | + | + | | | |
| Binturong | VU | + | + | | | |

## 4.3.2 Distribution patterns of carnivores in the *Prek chu* catchment

### *4.3.2.1 Spatial distribution pattern*

Carnivores were distributed throughout the landscape of the *Prek chu* catchment with varied levels of intensity in different vegetation types. Grids with high, medium and low frequency of presence locations are depicted for each species. Evidences of snow leopard presence indicated its distribution to be restricted only to the alpine and rocky areas. Most of the grids depicted low evidences of presence however; grids with high and medium frequency were situated in the vicinity of high altitude lakes such as *Lampokhri*. Evidences of snow leopard were also found near tourist destinations such as *Dzongri*, *Dzongri la* and *Basecamp*, respectively (Figure 4.3). Evidences of clouded leopard were recorded from temperate and subalpine areas, specifically around the *Lamphey* area (Figure 4.4). Leopard cat was the most frequently recorded felid species in camera traps restricted only to temperate areas. Grids representing high and medium frequency of leopard cat occurrence were located near *Yuksam* village (*Khengte, Malingo, Pakholla, Dubdi*) and *Sachen* (Figure 4.5). Jungle cat was recorded from temperate, subalpine and *krummholtz* habitats of the area, but was generally rare (Figure 4.6). Evidences of melanistic Asiatic golden cat were recorded from temperate and subalpine habitats, especially in areas such as *Jamling*, *Kasturi* and *Malingo* (Figure 4.7). Based on single photo-capture of common leopard, its distribution pattern could not be assessed. It is presumed to be uncommon in the study area and may be distributed in the buffer zone of *Prek chu* catchment and its lower areas.

Among canids and even other carnivore species, signs of red fox were encountered most frequently. The species was distributed in the alpine, rocky and *krummholtz* habitats. Grids with high and medium frequency of red fox occurrence were near *Dzongri*, *Lampokhri, Paireni* and *Thangsing,* respectively (Figure 4.8). Evidences of wild dog were distributed in upper temperate, subalpine and alpine areas (Figure 4.9). Jackal on the contrary was photo captured only once at the lower most altitude of *Prek chu* catchment depicting its distribution to be restricted in the sub-tropical habitats of the area, buffer zone and the adjacent villages of the catchment (Figure 4.10).

Asiatic black bear was the largest carnivore recorded from the area. Evidences of its presence were recorded from temperate, subalpine and even alpine areas. The grids with high and medium bear frequency were located near *Jamling, Dubdi,* and *Malingo,* although a photographic evidence of its presence was also recorded near *Thangsing* during winter season (Figure 4.11). Red panda being specifically arboreal appeared in few camera traps located in the subalpine habitats indicating is distribution restricted to subalpine habitats particularly the *Kasturi* area (Figure 4.12).

Among mustellids and even other carnivores, yellow-throated marten was the most frequently occurring species in camera traps. Evidences of its presence were recorded from temperate, subalpine and *krummholtz* habitats, although grids with high frequency of occurrence were located in temperate (near *Yuksam*, *Dubdi*, *Malingo* areas) and with medium frequency were located in upper temperate (*Lamphey*, *Sachen*) and subalpine (*Ghunsa*, *Jamling*, *Kasturi*) areas (Figure 4.13). Evidences of stoat on the other hand were recorded mainly from the subalpine areas (Figure 4.14).

Evidences of presence of both mask palm civet (MP-civet) and large Indian civet (LI-civet) were restricted to temperate areas and high frequency grids were also located near *Yuksam* village (*Dubdi*, *Pakholla*, *Malingo*), but it was observed that mask palm civet showed comparatively more tendency to occupy the upper temperate areas as well (Figure 4.15 & 4.16).

**Fig 4.3 Relative frequency of snow leopard evidences in 1×1 km grid**

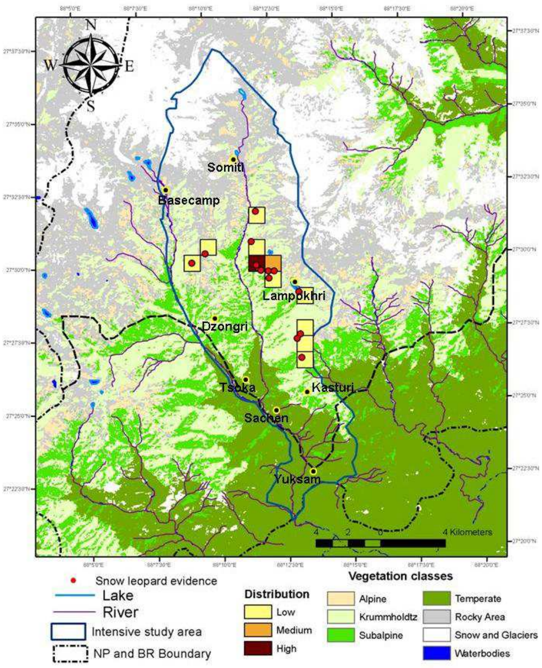
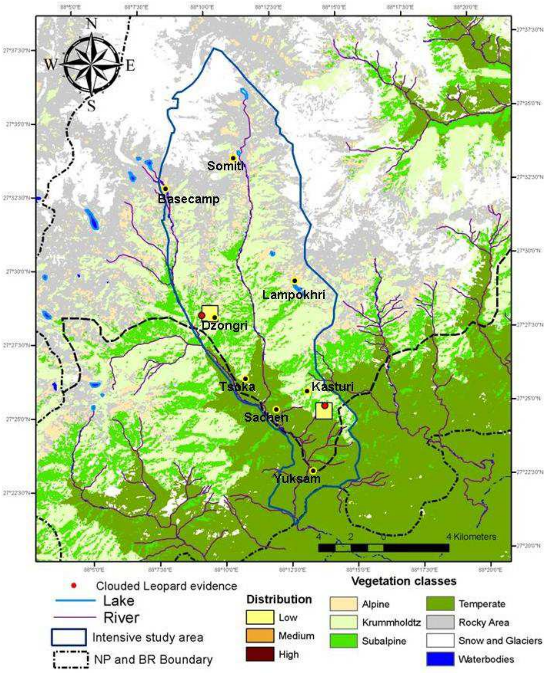
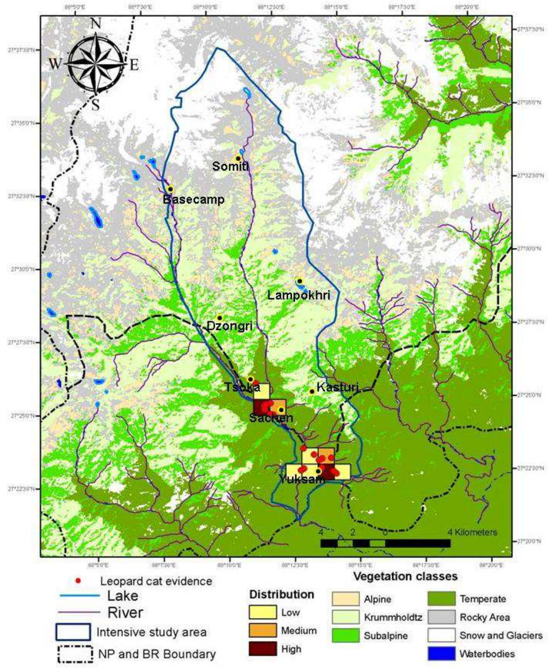


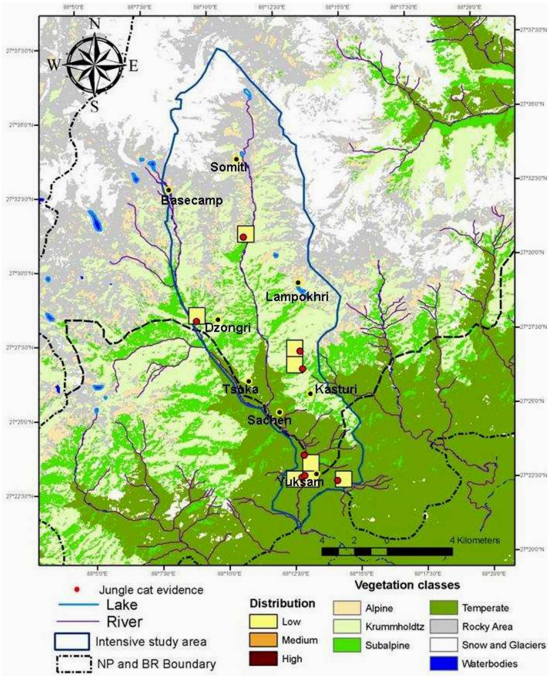
**Fig 4.4 Relative frequency of clouded leopard evidences in 1×1 km grids**



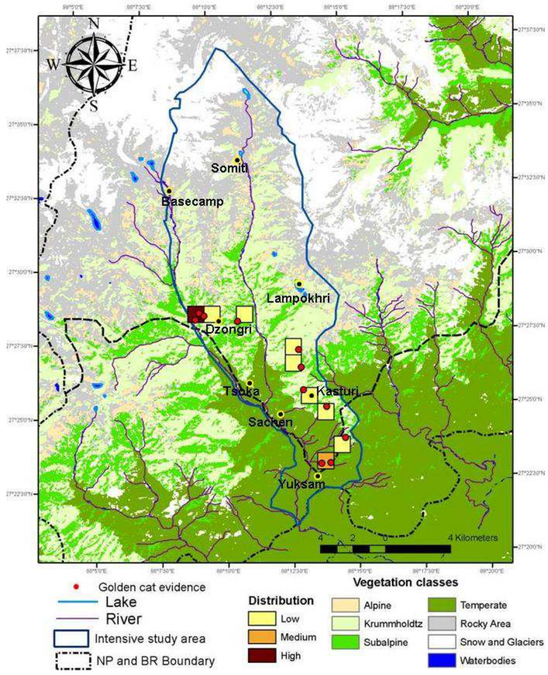
## Fig 4.5 Relative frequency of leopard cat evidences in 1×1 km grids



## Fig 4.6 Relative frequency of jungle cat evidences in 1×1 km grids



## Fig 4.7 Relative frequency of golden cat evidences in 1×1 km grids



## Fig 4.8 Relative frequency of red fox evidences in 1×1 km grids

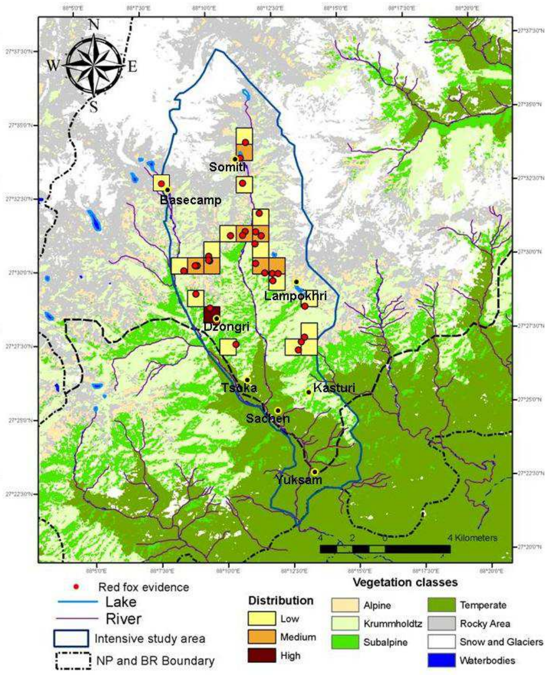


Fig 4.9 Relative frequency of wild dog evidences in 1×1 km grids

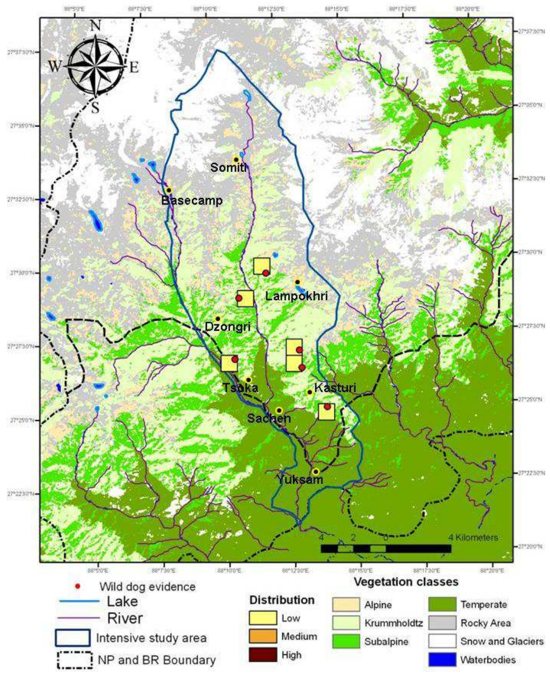


Fig 4.10 Relative frequency of golden jackal evidences in 1×1 km grids

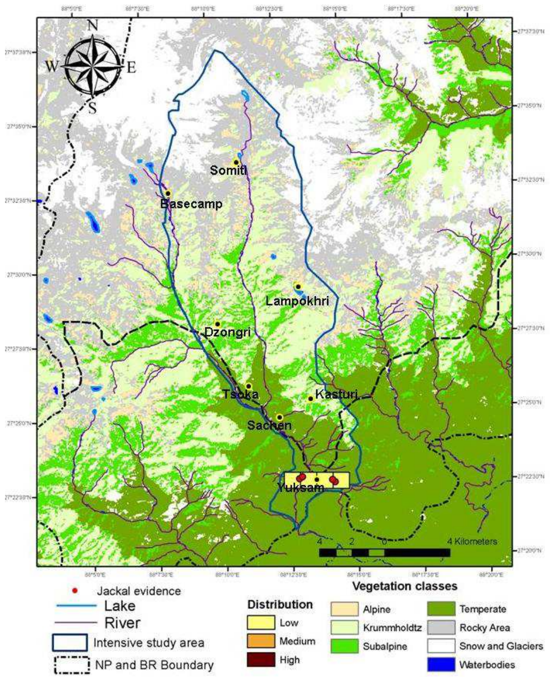


Fig 4.11 Relative frequency of black bear evidences in 1×1 km grids

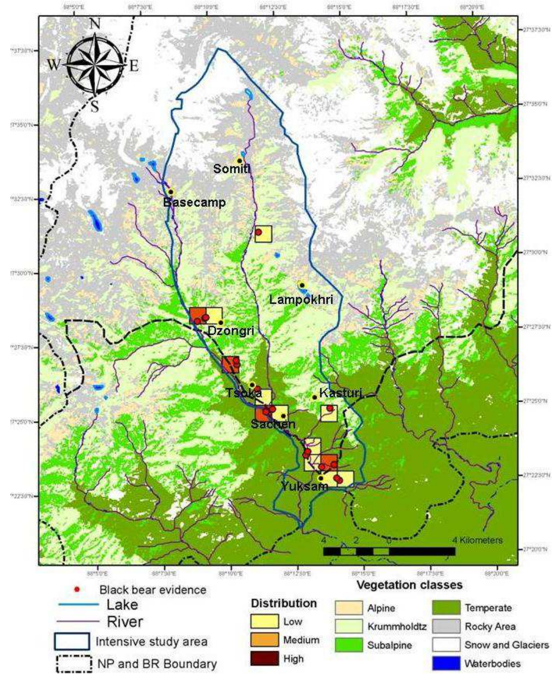
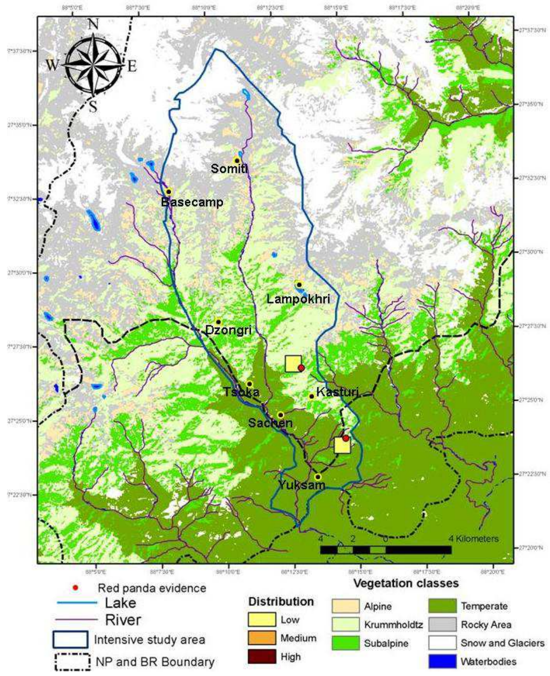
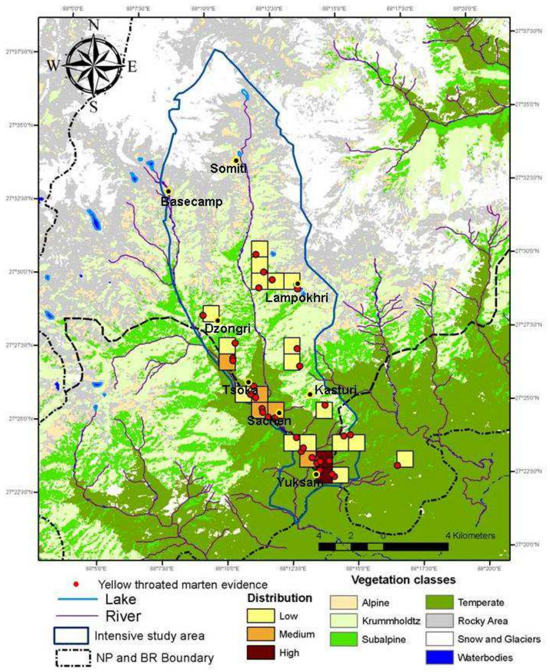


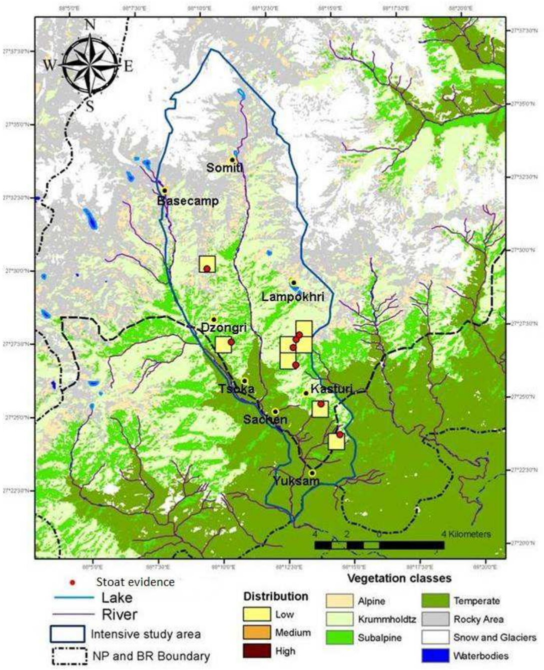
Fig 4.12 Relative frequency of red panda evidences in 1×1 km grids



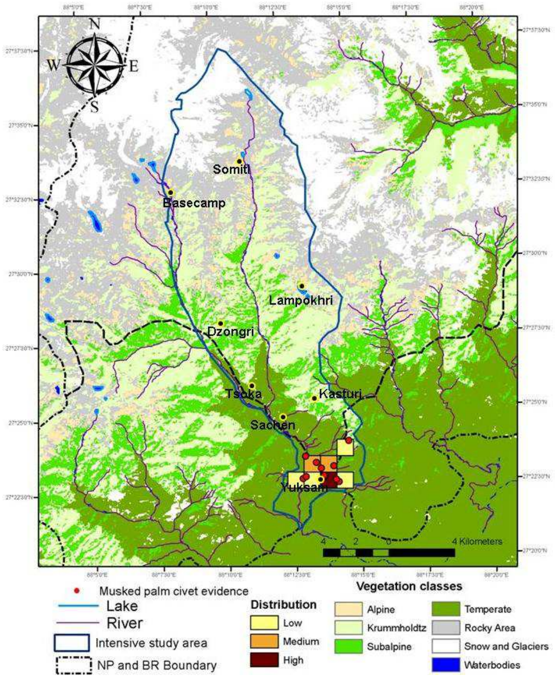
### Fig 4.13 Relative frequency of YT-marten evidences in 1×1 km grids



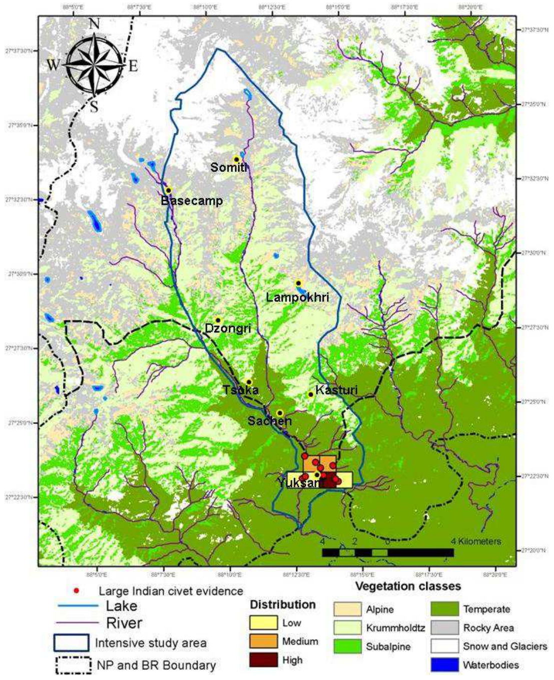
### Fig 4.14 Relative frequency of stoat evidences in 1×1 km grids



### Fig 4.15 Relative frequency of masked palm civet evidences in 1×1 km grids



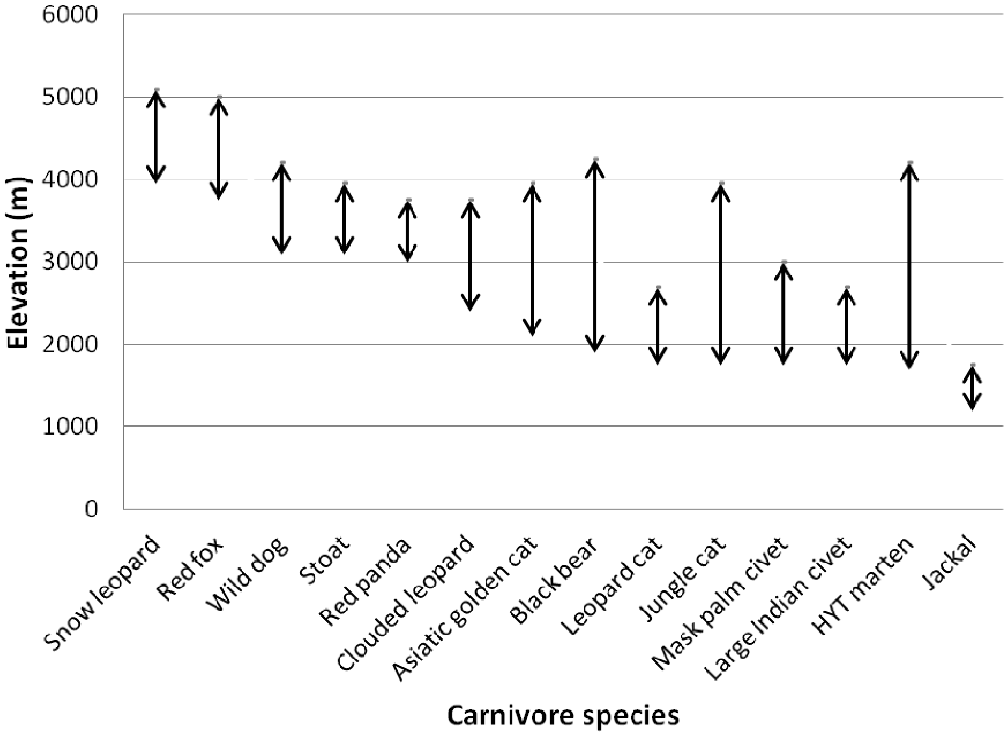
### Fig 4.16 Relative frequency of large Indian civet evidences in 1×1 km grids



#### *4.3.2.2 Altitudinal distribution pattern*

Carnivores were distributed along different elevation classes, some wide ranging while others had a restricted range. Snow leopard was distributed above 4,000 m up to 5,100 m. Asiatic golden cat ranged from 2,100 m up to 3,950 m which is an altitudinal record for the species. Clouded leopard ranged from 2,400 m to 3,750 m, but leopard cat was restricted only up to 2,700 m. Jungle cat believed to be a low land species was recorded even at 3,950 m which is also an altitudinal record for the species. Red fox ranged from 3,750 m to 5,000 m, and even wild dogs were recorded up to 4,200 m in the alpine-*krummholtz* habitats which is generally rare. Black bear occurred from 1,900 m up to 4,250 m, while red panda was restricted only up to 3,750 m. Yellow-throated marten was distributed from temperate habitats at 1,700 m to alpine areas at 4,200 m. Both the civets were distributed in the temperate habitats with an altitudinal range varying from 1,750 m to maximum 3,000 m. Among all the carnivore species recorded, yellow-throated marten had the most wide altitudinal range of more than 2,500 m while red panda was the most restricted species with an altitudinal range of less than 750 m. Details of the altitudinal ranges of species are given in Figure 4.17.

**Figure 4.17 Altitudinal range of different carnivore species recorded in the *Prek chu* catchment**



### 4.3.3 Population status of carnivores in *Prek chu* catchment

#### *4.3.3.1 Sign (scat) encounter rates*

During trail monitorings and sign surveys in different habitats of the *Prek chu* catchment, a total of 827 scat samples were encountered and collected. But, only 695 samples could be assigned species identities and the remaining (132 samples) were left unidentified. The identified scats belonged to seven species included snow leopard, red fox, yellow-throated marten, wild dog, leopard cat, Asiatic black bear and golden jackal. Among these species highest scat encounter rate (#/100 km walk) was recorded for red fox (34.8 ± 8.27) followed by snow leopard (17.82 ± 3.62) and leopard cat (6.7 ± 2.3), while lowest encounter rate was recorded for Asiatic black bear (1.46 ± 0.54). While, comparing the scat encounter rate for species in different habitats of their occurrence, scat encounter rate of red fox was significantly different in subalpine, *krummholtz* and alpine habitats (Kruskal-Wallis $\chi^2 = 10.71$, df = 2, $p = 0.005$), but no such significant difference was observed for other species among their habitats of occurrence (Table 4.5). In temperate habitat highest scat encounter rate was recorded for yellow-throated marten (8.8 ± 4.1), where as in subalpine, *krummholtz* and alpine habitats highest scat encounter rate was recorded for red fox. The overall encounter rate of unidentified scats was (6.9 ± 1.04) and their encounter rate did not vary significantly among different habitats.

#### *4.3.3.2 Photo-capture rates*

A sampling effort of 6,910 camera-days across 46 sample sites in the *Prek chu* catchment resulted in 5,100 photo captures [3,190 wild animals (1,906 mammals, 1,279 birds, 1 snake and 6 unidentified) 948 domestic animals and 962 human], out of which 708 photo captures were of wild carnivore species. Camera traps detected the presence of 17 out of total 19 species of carnivores confirmed from the study area, the other two being Siberian weasel (confirmed through sighting) and binturong (confirmed trough local interviews). Highest overall photo-capture rate (# captures/100 camera days) was observed for yellow-throated marten (5.76 ± 1.23) followed by red fox (4.33 ± 1.15) and Tibetan wolf (4.1 ± 2.055), while lowest photo-capture rate was recorded for common leopard (0.036 ± 0.036) and jackal (0.038 ± 0.038). Of the total carnivores, 9 species were photographed in single habitat zone and others in up to three different zones (Table 4.6). Photo capture rate of carnivores in different habitats were not significantly different except for red fox (Kruskal-Wallis $\chi^2$

= 8.25, df = 2, *p* = 0.016) and yellow-throated marten (Mann-Whitney U = 67, df = 1, *p* = 0.039). In temperate and subalpine zones highest photo-capture rates were observed for yellow-throated marten (8.51 ± 2.38 and 3.5 ± 0.8) respectively, while in alpine zone photo-capture rate was highest for red fox (8.16 ± 2.15). In addition to this, short camera trapping session of 205 camera days in the trans-Himalayan area of *Lhonak chu* yielded only 16 photo captures of carnivores belonging to red fox and Tibetan wolf with photo-capture rates of 3.2 ± 2.71 and 4.1 ± 2.055, respectively.

**Table 4.5 Mean encounter rate (with standard errors) of carnivore scats (#/100 km walk) in different habitats of the *Prek chu* catchment with Kruskal-Wallis $\chi^2$ statistics (* is Mann-Whitney U statistics). T = Temperate, SA = Subalpine, *K* = Krummholtz, A = Alpine, O = Overall**

| **Species** | **N** | **T** | **SA** | ***K*** | **A** | **O** | **KW $\chi^2$** | **df** | ***p*** |
|---|---|---|---|---|---|---|---|---|---|
| Snow leopard | 114 | - | - | 14.2<br>(2.6) | 21.5<br>(6.7) | 17.82<br>(3.62) | 14* | 1 | 0.522 |
| Red fox | 367 | - | 5.5<br>(1.67) | 49.7<br>(18.9) | 49.1<br>(9.2) | 34.8<br>(8.27) | 10.71 | 2 | 0.005 |
| HYT marten | 105 | 8.8<br>(4.1) | 3.9<br>(1.2) | 6.3<br>(2.75) | 1.77<br>(0.5) | 5.19<br>(1.3) | 6.49 | 3 | 0.09 |
| Wild dog | 41 | 2.97<br>(1.1) | 2<br>(0.54) | - | - | 2.5<br>(0.6) | 15.5* | 1 | 0.688 |
| Leopard cat | 37 | 6.7<br>(2.3) | - | - | - | 6.7<br>(2.3) | - | - | - |
| Black bear | 7 | 1.46<br>(0.54) | - | - | - | 1.46<br>(0.54) | - | - | - |
| Jackal | 24 | 4.42<br>(2.2) | - | - | - | 4.42<br>(2.2) | - | - | - |
| Unidentified | 132 | 7.3<br>(2.83) | 8.87<br>(1.9) | 6.9<br>(2.14) | 4.48<br>(1.35) | 6.9<br>(1.04) | 2.78 | 3 | 0.427 |

**Table 4.6 Photo-capture rate (with standard errors) of carnivore scats (# captures/100 camera days) in different habitats of the *Prek chu* and trans-Himalayan region of *Lhonak chu* with Kruskal-Wallis $\chi^2$ statistics (* is Mann-Whitney U statistics).**

| **Species** | **N** | **Temperate** | **Subalpine** | **Alpine** | **trans-Himalaya** | **Overall** | **KW $\chi^2$** | **df** | ***p*** |
|---|---|---|---|---|---|---|---|---|---|
| Snow leopard | 24 | - | - | 0.85 (0.35) | - | 0.85 (0.35) | - | - | - |
| Clouded leopard | 6 | 0.145 (0.145) | 0.037 (0.026) | - | - | 0.086 (0.06) | 114.5* | 1 | 0.73 |
| Leopard | 1 | 0.036 (0.036) | - | - | - | 0.036 (0.036) | - | - | - |
| Golden cat | 25 | 0.4 (0.24) | 0.43 (0.13) | - | - | 0.41 (0.13) | 95.5* | 1 | 0.28 |
| Jungle cat | 7 | 0.13 (0.09) | 0.11 (0.07) | - | - | 0.098 (0.047) | 116* | 1 | 0.85 |
| Leopard cat | 62 | 2.86 (0.87) | - | - | - | 2.86 (0.87) | - | - | - |
| Asiatic black bear | 25 | 0.52 (0.21) | 0.13 (0.07) | 0.05 (0.05) | - | 0.23 (0.08) | 5.6 | 2 | 0.06 |
| Red panda | 5 | - | 0.071 (0.05) | - | - | 0.071 (0.05) | - | - | - |
| Wild dog | 9 | - | 0.174 (0.09) | 0.09 (0.09) | - | 0.138 (0.06) | 88* | 1 | 0.18 |
| Red fox | 170 | - | 2.07 (1.19) | 8.16 (2.15) | 3.2 (2.71) | 4.33 (1.15) | 8.25 | 2 | 0.016 |
| Tibetan wolf | 9 | - | - | - | 4.1 (2.055) | 4.1 (2.055) | - | - | - |
| Mask palm civet | 40 | 1.9 (0.78) | - | - | - | 1.9 (0.78) | - | - | - |
| Jackal | 1 | 0.038 (0.038) | - | - | - | 0.038 (0.038) | - | - | - |
| Large Indian civet | 30 | 1.47 (0.66) | - | - | - | 1.47 (0.66) | - | - | - |
| HYT-marten | 281 | 8.51 (2.38) | 3.5 (0.8) | - | - | 5.76 (1.23) | 67* | 1 | 0.039 |
| Stoat | 34 | - | 0.46 (0.19) | 2.11 (1.49) | - | 1.17 (0.66) | 89* | 1 | 0.27 |
| Pale weasel | 3 | - | - | 0.056 (0.04) | - | 0.056 (0.04) | - | - | - |

### *4.3.3.3 Absolute abundance (Mark recapture)*

#### *4.3.3.3.1 Abundance of Snow leopard*

Intensive camera trapping for 126 days using 27 cameras yielded 15 photo-captures of snow leopard out of which 13 were usable and assigned individual identities. Four individual snow leopards were identified from the photo captures. The capture histories of these individuals did not violate the population closure assumption which was evident from the result the close test ($\chi^2$ = 9.51, df = 7, $p$ = 0.218). Mo (null model) was selected as the best model for abundance estimation followed by Mh (heterogeneity) and Mb (behavioral) based on minimum AICc value (Table 4.7 a). Non-spatial *ad hoc* method estimated snow leopard abundance as 4.0 ± 0.4 individuals with a capture probability of 0.18. The mean maximum distance moved (MMDM) by a recaptured snow leopard was 1.736 ± 0.271 km. The half MMDM effective trapping area calculated was 102.98 km$^2$ and thus the non-spatial density was estimated at 3.88 ± 0.4 individuals/100 km$^2$. Spatially-explicit maximum likelihood based method estimated snow leopard density at 4.25 ± 2.55 individuals/100 km$^2$ under the half normal detection function based on minimum ML AICc value (Table 4.7 b), while the Bayesian method estimated density at 4.77 ± 1.81 individuals/100 km$^2$ (Table 4.7 c).

**Table 4.7 (a) Model selection, (b) Abundance and density (per 100 km$^2$) of snow leopard using spatially-explicit maximum-likelihood and (c) Bayesian framework.**

**(a) Model selection**

| Model | AICc | Δ AICc | AICc wt | Model LL | # Par | Deviance |
|---|---|---|---|---|---|---|
| $M_o$ | 62.64 | 0.00 | 0.368 | 1 | 1 | 56.5 |
| $M_h$ | 63.7 | 1.06 | 0.216 | 0.588 | 2 | 59.68 |
| $M_b$ | 64.82 | 2.18 | 0.123 | 0.336 | 2 | 56.5 |
| $M_{bh}$ | 65.81 | 3.17 | 0.075 | 0.204 | 3 | 59.68 |
| $M_t$ | 83.03 | 20.39 | 0.00001 | 0 | 18 | 32.17 |
| $M_{th}$ | 158.2 | 95.6 | 0.00001 | 0 | 36 | 8.14 |

**(b) Spatially explicit density estimation (Max Likelihood)**

| Detection function | # par | ML AICc | MLDens (SE) | LL | UL | MLg0 (SE) | MLσ (SE) |
|---|---|---|---|---|---|---|---|
| Half normal | 5 | 116.76 | 4.25 (2.55) | 1.43 | 12.62 | 0.06 (0.036) | 1047.4 (268.1) |
| -ve exponential | 5 | 118.42 | 4.05 (2.48) | 1.34 | 12.23 | 0.13 (0.36) | 707.3 (231.1) |
| Hazard rate | 6 | 119.64 | 4.95 (2.87) | 1.73 | 14.2 | 0.033 (0.03) | 1699.6 (250.2) |

**(c) Spatially explicit density estimation (Bayesian with MCMC simulation)**

| Detection function | Psi (S.E) | $\sigma$ (S.E) | $\lambda_0$ (S.E) | N super (S.E) | D (S.E) | 95% CI | |
|---|---|---|---|---|---|---|---|
| | | | | | | LL | UL |
| Half normal | 0.6<br>(0.23) | 0.135<br>(0.122) | 0.052<br>(0.025) | 26.49<br>(10.08) | 4.77<br>(1.81) | 1.8 | 7.93 |
| -ve exponential | 0.56<br>(0.21) | 0.154<br>(0.054) | 0.143<br>(0.063) | 25.02<br>(9.3) | 4.5<br>(1.68) | 1.44 | 7.57 |

*4.3.3.3.2 Double-blind observer identification*

Individual identification of leopard cat and large Indian civet individuals through double-blind observer identification method proved convincing for both the species. In case of leopard cat, on comparing the different body parts, hind-quarter was utilized more successfully in classifying 83.9% of photographs with 77.8% exact matches among investigators, followed by mid-quarter (66.7%) and tail (64.2%). The investigators' opinions (exact matches) to assign identity to individuals based on separate body parts did not vary significantly for all parts (Table 4.8 a). For large Indian civet, on comparing the different body parts, tail was utilized more successfully in classifying 65.1% of photographs with 70.8% exact matches among investigators, followed by neck (59.8%) and hind-quarter (38.2%). The investigators' opinions (exact matches) to assign identity to individuals based on body parts did not vary significantly for all parts (Table 4.8 b).

**Table 4.8 Percentage of exact matches among the three investigators based on photo-captures with respective usability for each body part and significance of difference in usability categorization of each part across investigators for leopard cat (a) and large Indian civet (b)**

**(a) Leopard cat**

| Part | Exact matches % | Usability % | Kruskal-Wallis Test | |
|---|---|---|---|---|
| | | | $\chi^2$ | p-Value |
| Head | 33.3 | 48.1 | 0.879 | 0.644 |
| Neck | 29.6 | 51.8 | 3.81 | 0.149 |
| Fore-quarter | 48.1 | 54.3 | 0.688 | 0.709 |
| Mid-quarter | 59.2 | 66.7 | 1.317 | 0.518 |
| Hind-quarter | 77.8 | 83.9 | 2.35 | 0.308 |
| Tail | 40.7 | 64.2 | 5.19 | 0.074 |
| Overall match between investigators in categorizing individuals | | | | 84.15% |

**(b) Large Indian civet**

| Part | Exact matches % | Usability % | Kruskal-Wallis Test | |
|---|---|---|---|---|
| | | | $\chi^2$ | p-Value |
| Head | 0 | 4.35 | 6.74 | 0.034 |
| Neck | 58.5 | 59.8 | 11.83 | 0.003 |
| Fore-quarter | 8.33 | 7.07 | 4.32 | 0.115 |
| Mid-quarter | 23.3 | 24.9 | 4.63 | 0.099 |
| Hind-quarter | 28.8 | 38.2 | 8.89 | 0.012 |
| Tail | 70.8 | 65.1 | 14.95 | 0.001 |
| Overall match between investigators in categorizing individuals | | | | 88.9% |

### A) Abundance of leopard cat

During the intensive camera trapping session of 60 days using 23 cameras a total of 45 photo-captures (31 right and 14 left flanks) were recorded, out of which 27 and 10 were categorized as usable by all three investigators. The number of leopard cat individuals identified by investigators varied from 10 to 13 using right flank and 5 using left flank photo-captures, hence, right flank photo-captures was used for further capture-recapture analysis. Close test on capture histories of these individuals indicated the population to be closed ($\chi^2$ = 15.25, df = 13, $p$ = 0.291). Mo (null model) was selected as the best model for abundance estimation based on minimum AICc value (Table 4.9 a). Non-spatial *ad hoc* method estimated leopard cat abundance as 15.0 ± 2 individuals with a capture probability of 0.094. The mean maximum distance moved (MMDM) by a recaptured leopard cat was 1.917 ± 0.558 km. The half MMDM effective trapping area calculated was 96.35 km$^2$ and thus the non-spatial density of leopard cat was estimated at 15.57 ± 2.8 individuals/100 km$^2$. Spatially-explicit maximum likelihood based method estimated leopard cat density at 17 ± 5.33 individuals/100 km$^2$ with the hazard rate detection function based on minimum ML AICc value (Table 4.9 b), while though Bayesian method the density was estimated at 17.52 ± 5.52 individuals/100 km$^2$ using a half normal detection function (Table 4.9 c).

**Table 4.9 (a) Model selection, (b) Abundance and density (per 100 km$^2$) of leopard cat using spatially-explicit maximum-likelihood and (c) Bayesian framework.**

**(a) Model selection**

| **Model** | **AICc** | **Δ AICc** | **AICc wt** | **Model LL** | **# Par** | **Deviance** |
|---|---|---|---|---|---|---|
| $M_o$ | 131.136 | 0.00 | 0.504 | 1.00 | 2 | 108.27 |
| $M_h$ | 132.7 | 1.56 | 0.231 | 0.458 | 3 | 107.8 |
| $M_b$ | 133.18 | 2.04 | 0.181 | 0.36 | 3 | 108.27 |
| $M_{bh}$ | 134.75 | 3.62 | 0.083 | 0.163 | 4 | 107.9 |
| $M_t$ | 150.35 | 19.21 | 0.00003 | 0.0001 | 21 | 85.65 |
| $M_{th}$ | 152.74 | 21.6 | 0.00001 | 0 | 22 | 85.65 |

**(b) Spatially explicit density estimation (Max Likelihood)**

| **Detection function** | **# par** | **ML AICc** | **MLDens (SE)** | **LL** | **UL** | **MLg0 (SE)** | **ML σ (SE)** |
|---|---|---|---|---|---|---|---|
| Hazard rate | 4 | 278.99 | 17<br>(5.33) | 9.3 | 30.9 | 0.018<br>(0.005) | 2597.55<br>(74.14) |
| Half normal | 3 | 281.44 | 15.94<br>(5.3) | 8.4 | 30.1 | 0.029<br>(0.01) | 1501<br>(292.74) |
| -ve exponential | 3 | 282.54 | 15.96<br>(5.4) | 8.3 | 30.6 | 0.069<br>(0.029) | 967.49<br>(243.75) |

**(c) Spatially explicit density estimation (Bayesian with MCMC simulation)**

| **Detection function** | **Psi (S.E)** | **σ (S.E)** | **λ0 (S.E)** | **N super (S.E)** | **D (S.E)** | **95% CI** | |
|---|---|---|---|---|---|---|---|
| | | | | | | **LL** | **UL** |
| Half normal | 0.227<br>(0.074) | 0.2063<br>(0.093) | 0.0267<br>(0.007) | 32.02<br>(10.08) | 17.52<br>(5.52) | 8.76 | 26.82 |
| -ve exponential | 0.223<br>(0.082) | 0.275<br>(0.079) | 0.035<br>(0.013) | 31.16<br>(10.3) | 17.06<br>(5.64) | 9.3 | 27.4 |

### B) Abundance of large Indian civet

A total of 23 photo-captures (20 right and 15 left flanks) were recorded during the intensive camera trapping session of 90 days using 19 cameras out of which 17 and 8 photo-captures were categorized as usable by all three investigators. Five to six individuals of large Indian civet were identified by investigators using right flank photo-captures and used for further capture-recapture analysis. Close test on capture histories of these individuals indicated the population to be closed ($\chi^2$ = 7.84, df = 10, *p* = 0.644). Mo (null model) was selected as the best model for abundance estimation based on minimum AICc value (Table 4.10 a). Non-spatial *ad hoc* method estimated the abundance of large Indian civet as 5.0 ± 0.4 individuals with a capture probability of 0.107. The mean maximum distance moved (MMDM) by a recaptured large Indian civet was 1.236 ± 0.492 km. The half MMDM effective trapping area calculated was 75.53 km$^2$ and the non-spatial density of large Indian civet was estimated at 6.62 ± 0.99 individuals/100 km$^2$. Spatially-explicit maximum likelihood based method estimated the density of large Indian civet at 9.75 ± 5.8 individuals/100 km$^2$ with negative exponential detection function based on minimum ML AICc value (Table 4.10 b), while though Bayesian method the density was estimated at 10.98 ± 3.38 individuals/100 km$^2$ using a half normal detection function (Table 4.10 c).

**Table 4.10 (a) Model selection, (b) Abundance and density (per 100 km$^2$) of large Indian civet using spatially-explicit maximum-likelihood and (c) Bayesian framework.**

**(a) Model selection**

| **Model** | **AICc** | **Δ AICc** | **AICc wt** | **Model LL** | **# Par** | **Deviance** |
|---|---|---|---|---|---|---|
| $M_o$ | 94.30 | 0.00 | 0.47 | 1 | 1 | 85.75 |
| $M_b$ | 95.39 | 1.08 | 0.27 | 0.58 | 2 | 84.78 |
| $M_h$ | 96.35 | 2.05 | 0.167 | 0.358 | 2 | 85.75 |
| $M_{bh}$ | 97.47 | 3.17 | 0.096 | 0.205 | 3 | 84.78 |
| $M_t$ | 133.07 | 38.7 | 0 | 0 | 30 | 50.85 |
| $M_{th}$ | 136.2 | 41.9 | 0 | 0 | 31 | 50.85 |

**(b) Spatially explicit density estimation (Max Likelihood)**

| **Model** | **# par** | **ML AICc** | **MLDens (SE)** | **LL** | **UL** | **MLg0 (SE)** | **MLSigma (SE)** |
|---|---|---|---|---|---|---|---|
| -ve exponential | 5 | 123.64 | 9.75 (5.8) | 3.3 | 28.7 | 0.110 (0.09) | 494.2 (148.24) |
| Half normal | 5 | 123.97 | 9.13 (5.26) | 3.2 | 26.1 | 0.044 (0.013) | 809.66 (196.83) |
| Hazard rate | 6 | 144.17 | 9.23 (5.8) | 3 | 28.5 | 0.067 (0.029) | 603.14 (288.42) |

**(c) Spatially explicit density estimation (Bayesian with MCMC simulation)**

| **Detection function** | **Psi (S.E)** | **σ (S.E)** | **λ0 (S.E)** | **N super (S.E)** | **D (S.E)** | **95% CI** | |
|---|---|---|---|---|---|---|---|
| | | | | | | **LL** | **UL** |
| -ve exponential | 0.629 (0.196) | 0.108 (0.045) | 0.164 (0.123) | 19.06 (5.86) | 10.98 (3.38) | 5.8 | 17.3 |
| Half normal | 0.615 (0.22) | 0.087 (0.075) | 0.038 (0.0186) | 18.52 (6.45) | 10.67 (3.71) | 4.6 | 17.3 |

#### *4.3.3.4 Absolute abundance (Royle's Biometrics)*

Application of Royle's repeated count method on photo capture histories of red fox, yellow-throated marten, stoat and mask palm civet yielded important information on the detection probability, occupancy and abundance of these species. An attempt to calculate their densities was also made based on the abundance estimates and the total area of the sampled grids (effective sampled area [ESA]). The estimates are given below (Table 4.11).

**Table 4.11 Abundance of red fox, yellow-throated marten, stoat and masked palm civet estimated from Royle's repeated count method in the *Prek chu* catchment.**

| **Parameters** | **Red fox** | **YT-marten** | **Stoat** | **MP-civet** |
|---|---|---|---|---|
| Naïve occupancy | 46% | 52% | 34% | 31% |
| Detection probability (r) | 0.22±0.04 | 0.12±0.02 | 0.07±0.028 | 0.12±0.04 |
| Occupancy ($\psi$) | 0.53±0.11 | 0.72±0.09 | 0.43±0.146 | 0.34±0.12 |
| Average abundance/sample unit ($\lambda$) | 0.73±0.24 | 1.34±0.31 | 0.41±0.18 | 0.56±0.26 |
| Total abundance (N) | 13.84±4.56 | 28.16±5.56 | 9.44±4.16 | 10.64±4.9 |
| Effective sampled area (km$^2$) | 76 | 84 | 92 | 76 |
| Density (#/100 km$^2$) | 18.21±6 | 33.52±7.8 | 10.26±4.52 | 14.03±6.5 |

#### *4.3.3.5 Comparison among abundance estimates*

While comparing the abundance indices (sign encounter rates and photo-capture rates) of species with the estimate of absolute abundance (density) in order to assess the reliability of indices as surrogates for true abundances in the context of intricate eastern Himalaya terrain of *Prek chu* catchment, the relationship between sign encounter rate and density estimate was not found to be linear (Figure 4.18) indicated by low $R^2$ value. However, high degree of positive association and monotonic relationship was observed between photo-capture rates and estimates of absolute abundance ($R^2$ = 0.896, Figure 4.19). In addition, while comparing among two indices of abundance, the relationship between sign encounter rate and photo-capture was although positive but not strong ($R^2$ = 0.466, Figure 4.20).

**Figure 4.18 Relationship between sign encounter rates (# scats/100 km walk) and density estimates (#/100 km²) of carnivore species.**

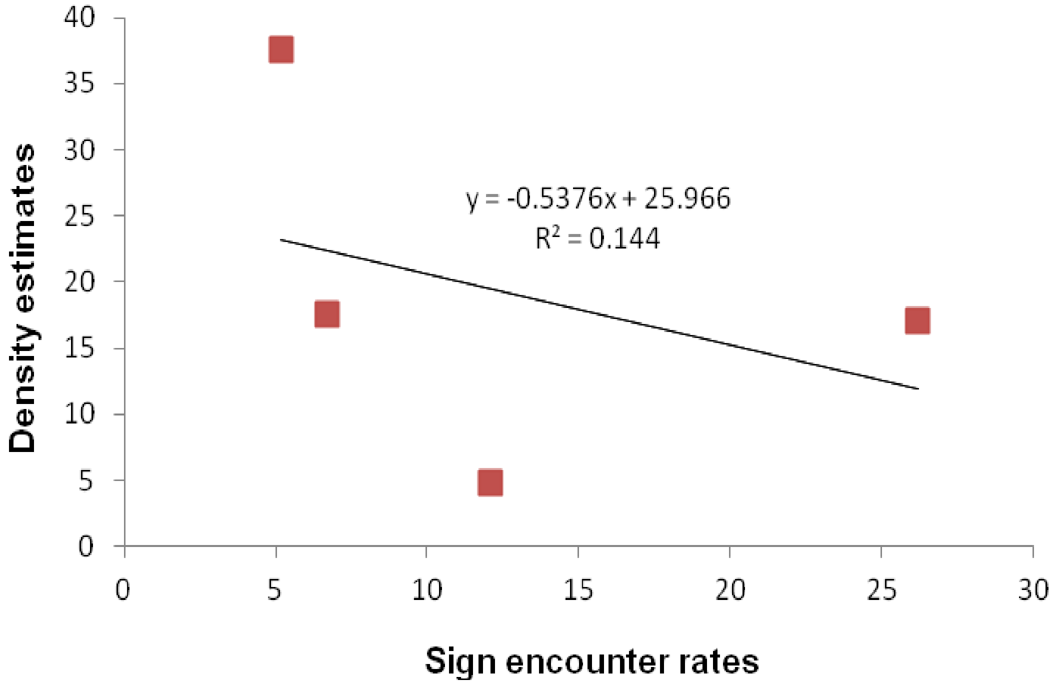


**Figure 4.19 Relationship between photo-capture rates (# photo-captures/100 trap days) and density estimates (#/100 km²) of carnivore species.**

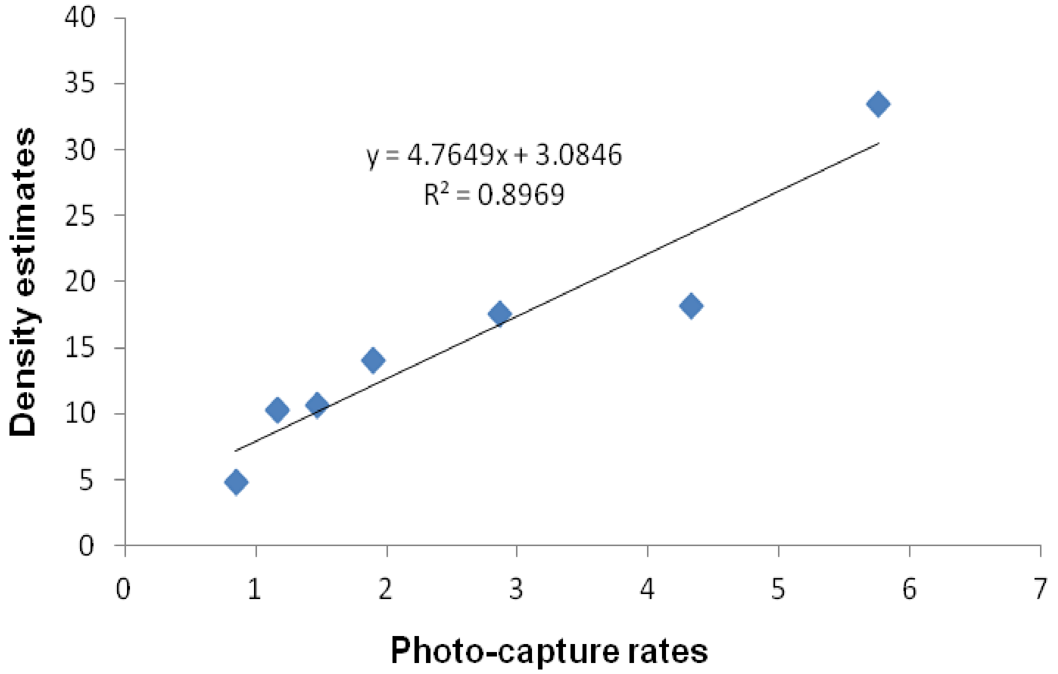
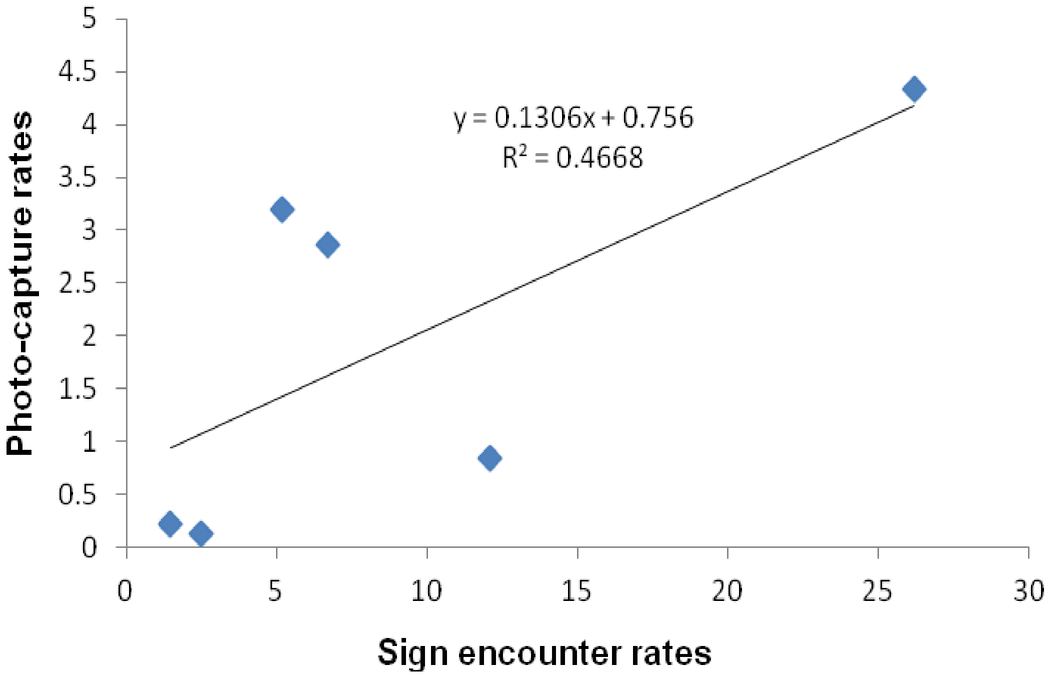


**Figure 4.20 Relationship between sign encounter rates (# signs/100 km walk) and photo-capture rates (# captures/100 trap days) of carnivore species.**



## 4.4 DISCUSSION

### 4.4.1 Carnivore assemblage

The occurrence of at least 19 species of carnivores indicates that the area has rich carnivore diversity ranging from small to large sized species. Presence of 11 carnivore species of global conservation significance including 9 typical Himalayan species validates the importance of the area as a high altitude biodiversity hotspot and warrants its effective conservation and management. Although, the carnivore diversity decreased with increase in altitude from temperate to alpine, it was found to be higher than the Himalayan landscape of Dhorpatan Hunting Reserve of Nepal (8 species- Aryal et al. 2010), northern Barandabhar Forest Corridor, Nepal (15 species- Aryal et al. 2012), a proposed high altitude National Park in Arunachal Pradesh (12 species- Mishra et al. 2006), and Khao Yai National Park, Thailand reported to home 12 species of carnivores (Jenks et al. 2011). This high altitude carnivore assemblage included elements from both the Palearctic and Indo-Malayan realms (Corbet and Hill 1992) with about 60% species of Palaearctic origin including snow leopard, leopard, dhole, red fox, Tibetan wolf, golden jackal, Asiatic black bear, yellow-throated marten, stoat, pale weasel and Siberian weasel and remaining 40% of Indo-Malayan origin, including Asiatic golden cat, jungle cat, leopard cat, clouded leopard, red panda, mask palm civet, large Indian civet and binturong (Chattopadhayay et al. 2006). Such high diversity of small carnivore is not only due to the wide altitudinal range resulting in a high diversity of habitat types from lowland forests to alpine areas, but also due to the contiguity of evergreen forest areas and their proximity to forests of South-east Asia rich in small carnivores.

Among the species mentioned above, this study provides the first photographic confirmation for many carnivores even from the entire state of Sikkim. Besides, reporting the presence of overall 42 species of mammals from the Biosphere Reserve (Sathyakumar et al. 2011a), some interesting observations on certain species of carnivores was also recorded. Snow leopard in spite of its cryptic nature was detected frequently compared to other common felid species such as leopard and jungle cat. Presence of only the melanistic form of Asiatic golden cat (reported as occasional elsewhere) did not even resemble any of the described three subspecies *viz.*, *Catopuma. t. temminckii*, *C. t. dominicanorum*, and *C. t. tristis* (Grubb et al. 2005). This occasional melanistic form has although been photographed only once from Makalu-Barun National Park, Nepal (Ghimirey and Pal 2009) which is located further west of KBR. The species being also reported from the northeast states of India (Choudhury 2007; Datta et al. 2008a; Lyngdoh et al. 2011), it is hypothesized that either all the golden cats of KBR are melanistic or they could be a different subspecies (Bashir et al. 2011). In addition, the reddish-brown long coat and long white or reddish brown hairs on paws indicate that the dhole population in KBR could be the subspecies *Cuon alpinus primaevus* (Durbin et al. 2004; Bashir et al. 2013a) which is believed to be one of the rarest subspecies of dholes in India (Johnsingh 1985) and has also been reported from the lower and middle altitudes of Bhutan (Johnsingh et al. 2007). Confirmed presence of Tibetan wolf through photo-captures from the trans-Himalayan part of north Sikkim validates the earlier reports of its presence by local herders (Chanchani 2007).

Inspite of the occurrence of arboreal species (clouded leopard and red panda) in camera traps; non-detection of binturong may have been due to its distribution in lower elevations where the camera trapping effort was low. Similar reason may be attributed for low-detection of common species such as leopard and jackal, although leopards have been reported to be recently (in 2008) eradicated from most of the lower areas in the buffer zone due to heavy persecution by locals as retaliation to their livestock losses (Lachungpa *pers comm*). However, present study could not detect the presence of species such as, spotted linsang, marbled cat, Pallas's cat, Eurasian lynx and European otter which were otherwise reported to be present in Sikkim and parts of KBR (Anon 1989; Awasthe and Jha 1999; Chattopadhayay et al. 2006; Chanchani 2007). Due to logistic constraint, rigorous and repeated monitoring could not be done in all the catchments of the KBR as main focus was limited to *Prek chu* catchment. The study was however successful in generating information that can act as a baseline for future monitoring of KBR as a whole.

### 4.4.2 Spatial and altitudinal distribution pattern

Distribution of a carnivore in an area is primarily determined by the spatial array of its prey (Gittleman 1996; Krebs 2001). But, in the Himalayan context habitat and elevation are also known to play a vital role in shaping its space. In the *Prek chu* catchment of the KBR, carnivores were distributed according to different habitats determined by sharp elevation gradients with a certain degree of overlap among their ranges. Although, some species were restricted to a particular habitat or elevation class, majority were wide ranging.

#### *4.4.2.1 Felids*

Snow leopard distribution was strictly confined to rocky areas and alpine habitats at altitudes above 4,000 m, thought the species has been reported to range between 3,000-5,400 m (Mallon 1984). Its concentration near the *Lampokhri* Lake may be due to the peculiar location of the lake which besides being a water source is surrounded by alpine meadows, lies in immediate connectivity to subalpine and *krummholtz* areas of *Kasturi* and *Rangit chu* catchment from one side and alpine and rocky areas of *Somiti-Gochella* and *Dzongri-Basecamp* from the other sides. This brings in diverse prey base from different habitat (such as Himalayan tahr and musk deer from subalpine and *krummholtz,* and blue sheep from alpine), and hence the snow leopard. In addition, the meadows surrounding the lake are also used by herders to graze their livestock, hence luring the snow leopards for easy prey. Preference for such areas has already been reported in other parts of its distribution range indicating its affinity for steep and rugged terrain with cliffs, alpine habitats, areas rich in ungulate prey and edges (Schaller 1977; Jackson and Ahlborn 1984; Mallon 1984; Chundawat 1992; McCarthy et al. 2005; Ale 2007). Moreover, the occurrence of snow leopard in the trans-Himalayan part of north Sikkim indicates that this area (in the eastern Himalayan context) also represents a potential snow leopard habitat similar to that of Ladakh, Lahul and Spiti in the western Himalayas (Mallon 1984; Mishra 1997) which are believed to hold the best snow leopard populations (Fox et al. 1991). Clouded leopards on the other hand are believed to primarily inhabit dense evergreen forests (Grassman et al. 2005a; Austin et al. 2007) and primary rainforest habitats (Santiapillai 1989). Their distributed in the upper temperate and subalpine habitats of the study area validates the affinity for dense and deeper forested areas due to its arboreal nature (Santiapillai 1989). Since, it has been recorded up to 2,500 m in the Himalayan foothills (Nowell and Jackson 1996), its occurrence up to 3,750 m (altitudinal record) indicates a broad altitudinal range.

The Asiatic golden cat although known to primarily inhabit tropical and sub-tropical evergreen to mixed and dry deciduous forests (Nowell and Jackson 1996) and less frequent in open habitats such as shrub and grasslands (Choudhury 2007), it is less common in montane forests (Holden 2001; Mishra et al. 2006). Recorded even at 3,738 m in Bhutan (Wang 2007), its occurrence up to 3,960 m in subalpine and *krummholtz* habitats near tree line besides frequent presence in temperate habitat depicts its adaptability to montane forests dominated by *Abies* and *Rhododendron*, and to tree line with dwarf *Rhododendrons*. Similarly, the occurrence of jungle cat up to 3,960 m near tree line is an interesting record contrary to its general affinity for low land habitats with water and dense vegetative cover, especially the reed swamps, marsh, and littoral and riparian environments, grassland, shrubby woodland and deciduous dipterocarp forest (Nowell and Jackson 1996; Sunquist and Sunquist 2002; Duckworth et al. 2005).

Spatial distribution of leopard cat was confined only to sub-tropical and temperate habitats and never occurred above 2,750 m elevation. The species is otherwise adapted to inhabit a broad variety of habitats, from tropical rain forest to temperate broadleaf and marginally coniferous forest, as well as shrub forest, successional grasslands, human modified agricultural landscapes, logged forests and rubber and oil palm plantations (Nowell and Jackson 1996; Grassman et al. 2005b; Rajaratnam et al.

2007). Although, it has been reported to extend into rhododendron-oak-maple forests up to 3,254 m (Ghimirey and Ghimire 2010) in Himalayas, the probable reason for its restricted distribution pattern in the study area may be to avoid competition with other lesser carnivores by occupying areas (sub-tropical and temperate) in close connectivity to human habitations in the buffer zone. Compared to other felid species, the leopard known to be the most habitat tolerant ranging from rainforests to deserts and up to 5,200 m in Himalayas (Nowell and Jackson 1996), the reason for its least detection could be either its restricted distribution to lower sub-tropical areas as reported by the locals or a strategy to avoid competition with other carnivores.

### *4.4.2.2 Canids*

The distribution of red fox being confined mainly to alpine and rocky areas and only till the tree line (*krummholtz* zone) at a minimum elevation of 3,750 m is justified based on its record in moorland and mountain areas (even above the treeline and alpine passes) up to 4,500 m (Macdonald and Reynolds 2008). However, the present study also indicates its orientation towards higher reaches compared to its distribution in western Himalayas where it has been reported at elevations as low as 505 m (Anwar et al. 2012). On the other hand, dholes being nomadic in behavior have been reported to inhabit a wide range of habitats except desert regions. Their sparse distribution patterns in the study area from upper temperate to alpine habitats is in accordance to their distribution in rest of the Himalayas, north-west and north-east India where they are reported to be more fragmented and recorded even from high altitudes of Ladakh at 4,500 m (Fox and Chundawat 1995; Durbin et al. 2008). Occurrence of Tibetan wolf on the contrary only in the trans-Himalayan habitats of north Sikkim is characteristic to the species and its distribution range elsewhere (Fox and Chundawat 1995; Mishra 1997). The golden jackal being the most common and widely occurring canid in India, inhabiting a variety of habitats including degraded open areas around human habitation and up to elevations of 2,000 m around hill stations (Prater 1971; Jhala and Moehlman 2008), its distribution in the sub-tropical habitats of the study area brings in interesting information on its space use. Moreover, the presence of jackal only up to 1,750 m and other canids not below 2,600 m (including indirect evidences), there appears to be an altitudinal gap of 850 to 1,350 m (no canid zone) in the distribution of wild canids in the area. This hypothesis needs to be tested through further rigorous monitoring and can help in better understanding of the altitudinal distribution of carnivores in the eastern Himalayas.

### *4.4.2.3 Ursids*

Asiatic black bear being a typical species to Himalayas, its distribution pattern in the study area inhabiting mainly the temperate and subalpine habitats, and occasionally the alpine areas is in accordance to its association with broad-leaved and coniferous forests with infrequent use of alpine meadows observed elsewhere (Garshelis and Steinmetz 2008). Its occurrence at 4,250 m matches with its previous record at 4,300 m in northeast India (Sathyakumar and Choudhury 2007). Such rare movements to different habitats and elevations seasonally are generally meant to track changes in food abundance (Izumiyama and Shiraishi 2004). The distribution of red panda on the contrary being restricted to an altitudinal range of just 750 m of subalpine habitats validates its habitat specificity and preference for mixed deciduous and coniferous forests with a bamboo-thicket understory (Roberts and Gittleman 1984). Supplementary to this, in eastern Himalaya it has been reported relatively abundant between 2,800 m to 3,600 m in other parts of Sikkim, Darjeeling, Arunachal Pradesh and Nepal (Pradhan et al. 2001; Sharma and Belant 2009), although its occurrence has also been reported at 2,350 m elevation in Neora Valley NP, Darjeeling (Mallick 2010), but never found above 3,730 m (Sharma and Belant 2009) which is similar to its present record at 3,750 m.

### *4.4.2.4 Mustellids*

The widest altitudinal range up to 4,200 m (altitudinal record) and occurrence in a diverse array of habitats depicts yellow-throated marten to be densely distributed throughout the *Prek chu* catchment, though more concentrated in the temperate areas. Since, it is generally known to be a voracious predator of live animals capable of feeding on a range of prey items (Prater 1971), can scavenge and also climb trees (Parr and Duckworth 2007), these characteristics enhances its adaptability to a wide range of habitats and elevation classes reflected by its occurrence from 30-2,680 m altitude (Than Zaw et al. 2008). On the contrary, stoats known to prefer coniferous and mixed woodlands (Pulliainen 1999), though they occupy a wide range of habitats and often found in forest-edge habitats, scrubs, alpine meadows and marshes (King 1983; Martinoli et al. 2001); their space use in the study area was restricted only to subalpine and *krummholtz* edge. The distribution pattern of pale weasel only in the alpine zone of the area ascertains its preference for alpine meadows, rocky slopes, sparse forest vegetation and predominately open landscapes (Hornskov and Foggin 2007). Moreover, it has been found to occur in the Himalayas up to 5,000 m and in Bhutan from 1,500-5,200 m (Thinley et al. 2004). Scanty information on the Siberian weasel although confirms its occurrence in the subalpine habitat, but continued monitoring is required to illustrate on this.

#### *4.4.2.5 Viverids*

Spatial distribution of both masked palm civet and large Indian civet confined mainly to temperate habitats indicates that both are restricted to the lower areas which are generally adjacent to human habitation. This pattern of space use is in accordance with their known habitat preferences for evergreen and deciduous forest, and disturbed habitat (Rabinowitz 1991; Heydon and Bullon 1996; Grassman 1998a; Duckworth 1997; Azlan 2003; Datta et al. 2008b). The occurrence of masked palm civet up to 3,000 m and large Indian civet up to 2,700 m also indicates the tendency of the former towards higher elevations as it has been previously recorded up to 2,500 m (Thinley et al. 1994) contrary to large Indian civet recorded up to 1,600 m (Than Zaw et al. 2008). Space use by binturong based on local interviews can be speculated to be confined mainly to the lower temperate and sub-tropical areas below 1,700 m which is in accordance with its reported occurrence in evergreen forests, primary and secondary lowland forest including grassland/forest mosaics elsewhere (Duckworth et al. 1999; Datta 1999; Grassman et al. 2005c).

### 4.4.3 Abundance indices and estimates

Estimating abundance of cryptic carnivore is always challenging, particularly in areas with limited accessibility, rugged terrain, dense vegetation and harsh weather conditions. Due to these constraints very few studies have been conducted on carnivores in Himalayas and particularly in eastern Himalaya (Mishra et al. 2006; Datta et al. 2008b; Gopi et al. 2012), and hence abundance estimates or indices for species are hardly available from these areas. In the present study, estimates of abundance and indices (photo-capture rates) indicated yellow-throated marten as the most abundant carnivore followed by red fox and leopard cat. Consequently, mustellids and viverrids were the most frequently occurring groups in camera traps followed by canids and felids.

### *4.4.3.1 Felids*

Snow leopards have been described as extremely cryptic and notoriously difficult to enumerate primarily due to their extensive range, inhabiting terrain and occurrence in very low densities (Schaller 1977; Jackson and Hunter 1996). As a result, majority of the studies relied largely on signs to generate information on their status (Schaller et al. 1988; Jackson and Hunter 1996; Hussain 2003). In the present study, sign (scat) encounter rate (0.178/km) of snow leopard was found to be much less compared to that of Sagarmatha NP, Nepal (4.18/km- Ale et al. 2007; 4.5/km- Lovari et al. 2009), north-east Pakistan (1.16/km- Hussain 2003), Zewar Gol, Ujnu Gol and Sha Junali areas of Torkhow Valley, Pakistan (12.8, 5.8, 2.8/km- Din and Nawaz 2011) and northwest India (0.8/km- Fox et al. 1991). While, comparing the photo-capture rates (#/100 trap days- RAI 2), the relative abundance of snow leopard (0.85) was comparable to the estimate at Jangart (0.93) and higher than at SaryChat (0.09) and Kyrgyzstan (McCarthy et al. 2008), but less than the estimate at Hemis NP, Ladakh (8.9 and 5.62- Jackson et al. 2006) and Tomur, China (2.37- McCarthy et al. 2008). Estimates of absolute abundance calculated through *ad hoc* capture-recapture models (3.88±0.4 individuals/100 km$^2$) were found to be higher compared to the estimates at Kyrgyzstan and China (0.15, 0.87 and 0.74- McCarthy et al. 2008), but lower than at Hemis NP, Ladakh (8.49 and 4.45- Jackson et al. 2006). These results indicate that snow leopards are not rare in KBR, though less common. Spatially-explicit capture recapture density (4.25±2.55- maximum likelihood and 4.77±1.81- Bayesian) estimated for snow leopard was comparatively higher than the *ad hoc* estimate and with higher standard errors. The reasons being that SECR models address the movement patterns more explicitly and hence are believed to be more comprehensive and robust. Moreover, the predicted posterior pixel densities by the Bayesian method indicated alpine and *krummholtz* habitats adjacent to *Lampokhri* Lake as high density areas (Figure 4.21 a) which is in accordance with the earlier prediction from photo-capture rates as its relative index of spatial use.

Robust estimates of abundance and density of leopard cats were not available from any part of the species range except recently from three commercial forest reserves in

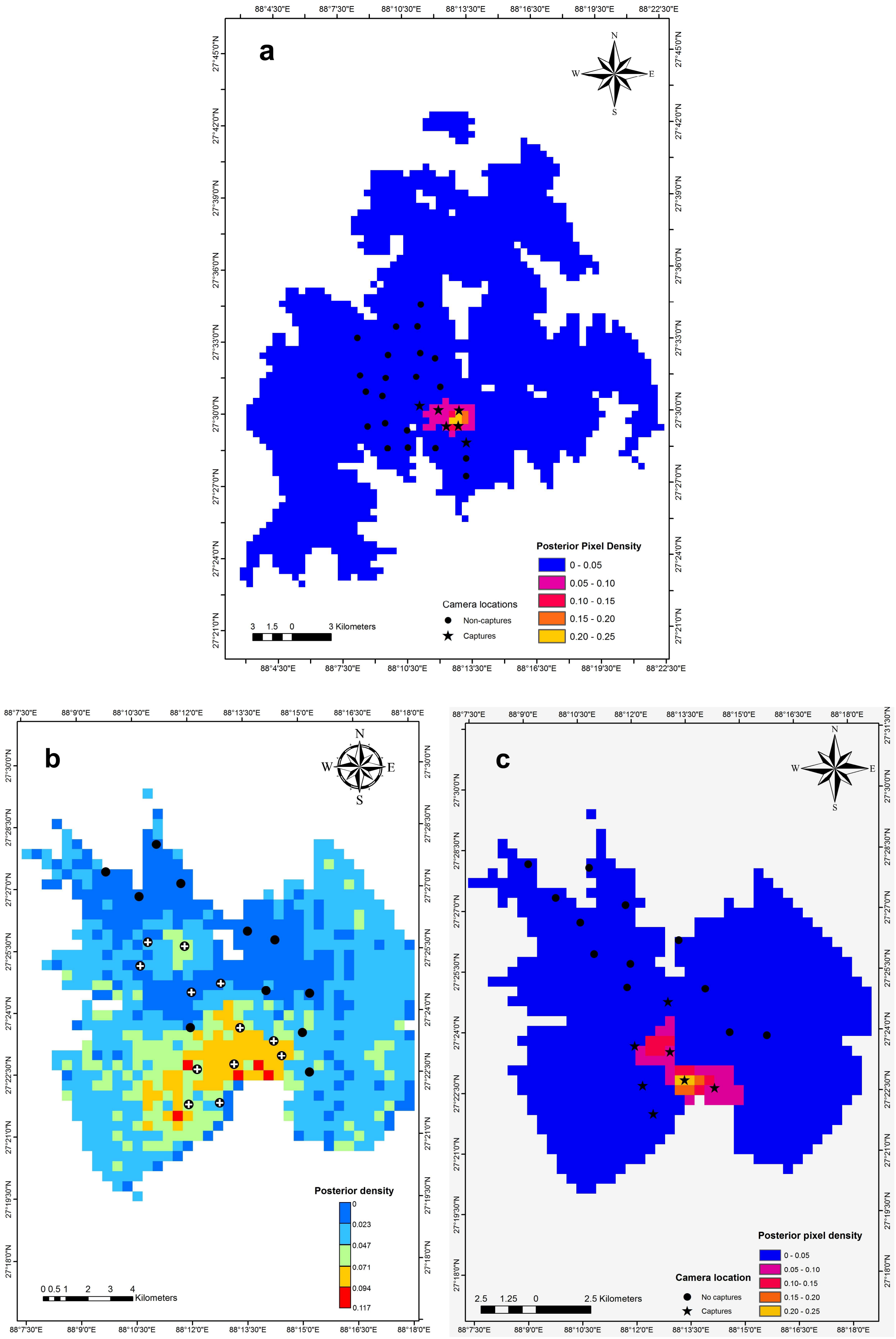
Sabah, Malaysian Borneo (Mohamed et al. 2013), and previous studies had relied only on sign encounter rates and photo-capture indices. Comparisons of RAI 2 (2.86) with different studies indicated that leopard cats are more abundant in KBR compared to secondary forests of Peninsular Malaysia (1.44- Azlan and Sharma 2006) and Sabangau peat-swamp forest of Indonesian Borneo (2.45- Cheyne and Macdonald 2011), but less abundant than in Deramakot Forest Reserve in Sabah, Malaysia (9.55- Azlan et al. 2009) and the Annapurna Conservation Area, Nepal (8.65- Appel et al. 2013). Estimates of absolute abundance indicated leopard cat to be the most abundant felid of the lower elevations of the study area. While comparing the density estimate, leopard cats were found to be more abundant than observed in three commercial forest reserves in Sabah, Malaysian Borneo (12.4±1.6, 16.5±2 and 9.6±1.7 individuals/100 km$^2$; Mohamed et al. 2013). Moreover, the predicted posterior pixel densities deduced through the Bayesian method also indicated dense temperate and sub-tropical forests of low elevation as high density areas for leopard cat (Figure 4.21 b) as earlier mentioned through their photo-capture rates.

Very few studies have focussed on estimating the population status of small felids and whatever information available (mainly relative abundance) is a bi-product of studies on large carnivores. While comparing the relative abundance index of Asiatic golden cat (RAI 2 = 0.41) it was found to be higher than at Kuala Terengan (0.082) and Kuala Koh (0.246) but, lower than at Merapoh (0.484) and Jerangau Forest Reserve (0.64) in Peninsular Malaysia (Azlan and Sharma 2006; Kawanishi and Sunquist 2008). Although, it has been reported from Nepal Himalaya and north-eastern states of India, any estimates of abundance are lacking from these areas (Choudhury 2007; Datta et al. 2008a; Ghimirey and Pal 2009; Lyngdoh et al. 2011). Clouded leopards although being arboreal are rarely captured in camera traps; still have been recorded in a number of camera trap studies in Himalayas and elsewhere. While comparing their photo-capture rates (0.086), they were found to be less abundant in KBR than in Annapurna Conservation Area, Nepal (0.33- Appel et al. 2013), Sabangau peat-swamp forest of Indonesian Borneo (2.74- Cheyne and Macdonald 2011), Deramakot Forest Reserve in Sabah, Malaysia (0.52- Azlan et al. 2009) and secondary forests of Peninsular Malaysia (0.22- Azlan and Sharma 2006). Although, jungle cat is believed to be common and abundant elsewhere, its status seems to be rare in KBR. Similarly, the status of common leopard is also rare depicted by a single photo-capture in the entire camera trapping effort.

### *4.4.3.2 Canids*

Red fox being one of the common species of Himalayas was also the most abundant wild canid recorded in the KBR. Although, many studies have reported its presence through sightings, signs and even camera trap photos; very few have provided estimates of its abundance, particularly from Himalayas. The indices generated in the present study can act as baseline for future comparisons. Dholes on the contrary are believed to be less common in the Himalayas and north-eastern states. Although being reported from the high altitudes of Ladakh (Fox and Chundawat 1995), Namdapha National Park (Datta et al. 2008a) and from six different valleys of high altitudes of Arunachal Pradesh (Mishra et al. 2006), no estimates of their abundance are available from these areas. However, trail monitoring in Pakke Tiger Reserve depicted dhole sign encounter rates to be low (0.5/km- Gopi et al. 2012) but still much higher than the indices derived in the present study (0.025/km). These inferences indicate the dholes are rare in the north-eastern states and particularly the subspecies found in Sikkim (*Cuon alpinus primaevus*) is the rarest and hence needs adequate protection (Bashir et al. 2013a). Among the trans-Himalayan species, the Tibetan wolf known to be mysterious and wide ranging (Fox and Chundawat 1995) was frequently detected in camera traps. Photo-capture rates showed the wolves are even more abundant than red foxes in the trans-Himalayan parts of Sikkim and hence offer a potential scope for long term studies. Jackals were however frequently sighted in the sub-tropical areas and also reported by locals. Being common elsewhere, similar scenario is hypothesized for the lower elevations zones of the Biosphere Reserve adjacent to human settlements.

Figure 4.21 Posterior densities of (a) snow leopard, (b) leopard cat, and (c) large Indian civet predicted through spatially explicit capture-recapture model in a Bayesian framework



#### *4.4.3.3 Ursids*

Abundance indices of Asiatic black bear as sign encounter rate and photo-capture rate (1.46/100 km; 0.23/100 trap days) indicated that bears occur at very low densities in the area compared to Dachigam National Park, Kashmir (scat = 75/100 km; RAI 2 =18- Charoo 2012) and similar forests of Annapurna Conservation Area, Nepal (RAI 2 = 0.81- Appel et al. 2013), though they may be abundant outside the National Park adjacent to human habitation. Being arboreal in nature, besides a single sighting no signs of red panda (scats) were recorded during the study except the camera trap photos. Previous studies have reported the presence of red panda in the north-eastern states and parts of Nepal, but very few have calculated abundance indices (expressed mainly as crude estimates of signs/hour of survey). Comparing single sighting in four years of monitoring with 32 sightings in 3 year (Pradhan et al. 2001) and 34 sightings in 11 years (Mallick 2010), red pandas are either more elusive or occur in low densities in the study area.

#### *4.4.3.4 Mustellids*

Based on indices and estimates of absolute abundance, yellow-throated marten was found to be the most abundant carnivore of the study area. The presence of this species has been reported in a number of studies in Himalayas and regarded as common in occurrence. Its photo-capture rate (5.76/100 trap days) indicated it to be more abundant in KBR than Namdapha NP in north-east India (0.32- Datta et al. 2008b), Annapurna Conservation Area, Nepal (2.7- Appel et al. 2013) and Deramakot Forest Reserve in Sabah, Malaysia (0.052- Wilting et al. 2010). Abundance indices of stoat, Siberian weasel (single sighting) and pale weasel indicated these species to occur in low densities in the study area.

#### *4.4.3.5 Viverids*

Estimates of both photo-capture rate and absolute abundance estimates indicated masked palm civet to be more abundant in the area compared to large Indian civet. Since, no estimates of absolute abundance are available across the distributional range of the two species particularly for masked palm civet; comparisons were possible only on the basis of indices. Large Indian civets were found to be more abundant (RAI 2 = 1.47) in the study area compared to Namdapha NP (0.72- Datta et al. 2008b) and Khao Yai National Park, Thailand (1.4- Jenks et al. 2011) but less abundant compared to

Pakke WS (1.73- Datta et al. 2008b) and Annapurna Conservation Area, Nepal (2.97- Appel et al. 2013). Similarly, the abundance of masked palm civet (RAI 2 = 1.9) was higher compared to Namdapha NP (0.32- Datta et al. 2008b). The predicted posterior pixel densities deduced through the Bayesian method also indicated dense temperate forests as high density areas for large Indian civet as earlier mentioned through their photo-capture rates (Figure 4.21 c). Although, binturong did not occur in camera traps, it was reported to occupy lower areas and probably in low densities which need to be illustrated through future studies.

### 4.4.4 Monitoring methods

The study demonstrated the potential applicability of camera trap technique as a powerful monitoring tool for detecting the presence of elusive carnivore species (Silveira et al. 2003; Kays and Slauson 2008), even in the intricate habitats of eastern Himalaya (Sathyakumar et al. 2011a). Other methods such as trail monitoring, sign surveys and local interviews could not prove to be so effective, and detected only few species without much confirmation. Information generated through local interviews was however more reliable (reporting 15 carnivores) compared to trail monitoring and sign surveys reporting the presence of 11 species of carnivores. As evident from the relationship between different indices and estimates of abundance, photo-capture rates could act as reliable surrogates of absolute abundance for species in such intricate high altitude terrain, while sign surveys and trail monitorings may provide biased estimates and hence lead to erroneous inference.

In an attempt to estimate absolute abundance of carnivore species, this study also demonstrated the protocol for identifying leopard cat (Bashir et al. 2013b) and large Indian civet individuals from their coat patterns. In addition, the applicability of spatially explicit capture-recapture (SECR) technique to reliably estimate the abundance and density of snow leopard, besides leopard cat and large Indian civet was also tested. The results showed that the densities estimated under different models (*ad hoc* and SECR) were almost similar (leopard cat), slightly different (snow leopard) and over estimated by SECR models (large Indian civet). The pattern is clearly depicted from their respective mean maximum distances moved (leopard cat>snow leopard>large Indian civet). Generally, non-spatial *ad hoc* models are known to overestimate densities as these underestimate individual movements (Royle and

Young 2008). The possible explanation for a rather contrary scenario in the present case may be due to significant differences in movement patterns of individuals. Also, as evident from capture histories and posterior distributions predicted by the Bayesian model, the species (particularly large Indian civet and snow leopard) were found to be concentrated over small areas and had restricted movements which probably might have resulted in the overestimation of their densities through SECR methods. The SECR models are however known to be more comprehensive as these address the movement patterns more explicitly which was even evident from comparatively high standard errors in their density estimates in the present cases. While comparing among SECR models, although classical maximum-likelihood based inference procedures are asymptotic and establish unbiasedness for large sample sizes (Efford 2011), their relevance to small sample situations has been inferred questionable (Royle and Young 2008). Conversely, Bayesian inferences do not rely on asymptotic arguments and are valid regardless of the sample size (Royle et al. 2009 b). But, in reference to the identification of some bias and poor coverage of credible intervals for Bayesian estimates attributed in part to small samples by Marques et al. (2011) and Efford (2011), the issue of robustness of their estimates continue to be a topic of debate. In order to overcome the sample size constrains, future studies need to optimize camera placement for these species in terms of distance from the path, height of camera in addition to placing cameras following sign surveys. Considering this as a case study from Sikkim Himalaya, it is recommended that SECR technique should be adopted as a regular monitoring tool for estimating densities of these and similar species in all camera trapping studies.

Although, abundance estimation through occupancy based Royle’s repeated count method (Royle 2004) proved promising for species with non-identifiable coat patterns in the Himalayan terrain, sample size constrains limited its applicability only to few species. Still, the variability in the information regarding the movement patterns of species reported in different studies might have violated the model assumptions and hence influenced the estimates. Such constraints can be addressed in future studies through improved sampling designs.

### 4.4.5 Factors determining distribution and abundance

It was observed that carnivore species in the study area generally occurred at higher elevation and had a broader habitat range compared to their previous records elsewhere. Latitudinal position of this area near to the Tropic of Cancer and geographically proximal to the Bay of Bengal and thus to the monsoon winds makes this zone more humid and more luxuriant in primary productivity. While moving towards the higher range the vegetation changes distinctly from sub-tropical pine and broad leaved forests to temperate oak and conifers to the subalpine and tree line vegetation of Fir, Birch and Rhododendron. In more humid conditions in eastern Himalaya the upper limit of forest varies from 4,600-4,900 m compared to 3,800-4,200 m in western Himalayas. Moreover, due to the heavy rainfall, the 'treeline' approaches at higher altitude of 4,000 m and the *krummholtz* and alpine scrub zone become more wide-ranging, thus causing species to inhabit broader habitat ranges and higher elevations. Again, since the area is positioned at the convergence of two biogeographic realms it harbors a rich and abundant carnivore assemblage. In order to understand the variation in abundance within different groups and species, information on other topographic, climatic and ecological variables and their relationship with carnivore space use and abundance is necessary. Moreover, competition within groups and species could be another important factor that can play significant role in shaping the carnivore guild of the area and hence needs to be assessed as well. In the next chapter (Chapter 5), an attempt has been made to address these questions in order to understand the strategies of resource partioning that lead to their co-existence.

# Chapter 5

# FACTORS DETERMINING CO-EXISTENCE

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## 5.1 INTRODUCTION

Understanding how ecological communities are structured is a central issue in ecology and has been a source of debate among ecologists since Clements (1916) and Gleason (1926) offered their competing views on the organization of plant communities. Given the finite resources within any ecosystem, species coexist under a variety of conditions and therefore are likely to be subjected to different ecological processes at different localities (Campbell 2004). Locally, the classic mechanisms of competition, predation, mutualism, parasitism and disease play important roles, but at coarser scales habitat heterogeneity can act as a primary contributor to species diversity (Campbell 2004). This struggle for limited resources in nature even compels the carnivores which are positioned at the apex of the trophic level to share their niches thereby leading to overlap, co-existence or even competition.

Resource partitioning has been used to explain inter-and intra-guild competition and co-existence among carnivores (Sunquist and Sunquist 1989; Walter 1991; Meriggi and Lovari 1996). Partitioning by means of selection for prey species (Karanth and Sunquist 2000; Wang 2008), sizes (Gittleman 1985; Karanth and Sunquist 2000), foraging habits (Palomares et al. 1996), activity patterns (Fedriani et al. 1999; Karanth and Sunquist 2000) and use of space (Palomares et al. 1996; Durant 1998) have all been used to explain co-existence among carnivores in ecological systems (Schoener 1974). Therefore, investigations on resource use by predators and their relationship with prey are important in understanding the mechanisms that influence vertebrate community structure (Vieira and Port 2007). One way to understanding community organisations, is to measure overlap in resource use among different species in a community guild and the most common resources measured in order to calculate overlap are food and space (Lanszki et al. 2007).

Dietary analysis is a frequent first step in studying an animal's ecology because diet directly reflects resource use and can provide insight into habitat utilization and competitive interactions (Litvaitis 2000). Precise knowledge of a carnivore’s diet is essential to assess the species’ role in the ecosystem, potential competition with other carnivores and impact on prey populations (Klare et al. 2011). Information on the feeding ecology of carnivores can be acquired through a variety of techniques including faecal analysis, spoor tracking, radio tracking and direct observation of animals hunting. But, in cases where the study species is solitary and elusive the condition gets difficult because direct observation of prey capture is rarely possible, particularly in rugged mountainous areas. Thus, in such scenarios, scat analysis remains the primary tool to assess carnivore diets, especially when focusing on individual prey items (Klare et al. 2011).

Partitioning in space can be achieved either through evolved differences in species-specific habitat preferences (Nú˜nez et al. 2000), or by active avoidance of the actual presence of competitor individuals, for example by using olfactory cues (Sollmann et al. 2012). While aspects of spatial ecology and interaction among species and individuals are best studied using radio-telemetry (Millspaugh and Marzluff 2001), these methodologies are challenging to be administered in Himalayan landscapes due to various logistic constrains; and in addition being invasive their applicability for elusive species such as snow leopards are often subjected to sample size constrains. In contrast, non-invasive methods such as camera trapping can yield information on a larger part of the population under study (MacKay et al. 2008; Chen et al. 2009; Davis et al. 2011). Since camera detections can be geographically referenced, these data can be used to gain insight in aspects of spatial ecology of elusive species (Harmsen et al. 2009; Sollmann et al. 2012). But, the use of raw photographic data such as count statistics can be problematic and the analyses of space use can lead to biased results if they do not account for imperfect detection, which can further vary among species and in space (MacKenzie et al. 2006). Site occupancy is a reliable method for estimating the distribution and space use of a species in a given area by taking detection probability into account (MacKenzie and Royle 2005; MacKenzie 2006). Site occupancy corresponds to the total number of sites/sampling units that a species occupies on a spatial scale while detection probability is the chances of any species being detected in that particular sampling area (Mackenzie et al. 2002). In the present study, modelling of the essential habitat characters for occupancy of carnivores was hence carried out to understand their broad-scale resource utilization pattern. Keeping in view that imperfect detection can effectively cause bias in the results, modelling of detection probability of each carnivore species was also carried out relative to detection variables varying with time.

As mention earlier, to facilitate co-existence and avoid competition in environments with limited available resources, sympatric carnivores often partition the resources temporally. To elucidate on this requires knowledge on the activity patterns of these species. Carthew and Slater (1991) demonstrated that automated photography could be used to monitor mammal activity. Subsequently, camera trapping has also proved to be a useful tool for providing information on the activity pattern of several elusive species, based on the date and time stamps on each photograph captured (Azlan and Sharma 2006; Grassman et al. 2006; Chen et al. 2009; Cheyne and Macdonald 2011), with much improved precision over physical traps used for small mammals.

A frequent observation however in the studies on resource partioning of mammals is that primary separation occurs along the microhabitat dimension rather than the food dimension (Schoener 1974). Part of the difficulty whether such a generalization holds true for world-wide communities is the lack of studies documenting both habitat and food utilization patterns simultaneously. In case of carnivores, a number of studies have been carried out to understand the mechanism of resource partioning in different communities in varied habitat conditions (e.g., Karanth and Sunquist 1995; Juarez and Filho 2002; Andheria et al. 2007; Papakosta et al. 2010; Kumaraguru et al. 2011) but only few have been done in mountainous habitats (Paquet 1992; Fedriani et al. 1999; Chen et al. 2009; Davis et al. 2011; Selvan et al. 2013). Furthermore, Himalaya in spite of being a unique ecosystem inhabiting a rich diversity of carnivores including typical high-altitude species there has not been any attempt so far towards understanding how these species manage to coexist in a community and what strategies these animals adopt in order to partition the limited resources available in such fragile habitats. With this background, I tried to answer few research questions mentioned below through this chapter, in order to understand the complexities associated with carnivore communities inhabiting such unique high-altitude ecosystems in the Khangchendzonga landscape.

1. Which factors govern the intensity of space use by carnivores in the study area?
2. What is the dietary spectrum of major carnivores and is there any overlap in their food habits?
3. What is the activity pattern of carnivores and how does it differ among sympatric carnivores?
4. What is the pattern by which carnivores ultimately partition the resources with respect to space, food and time?

## 5.2 METHODS AND DATA ANALYSIS

### 5.2.1 Occupancy design- camera trapping

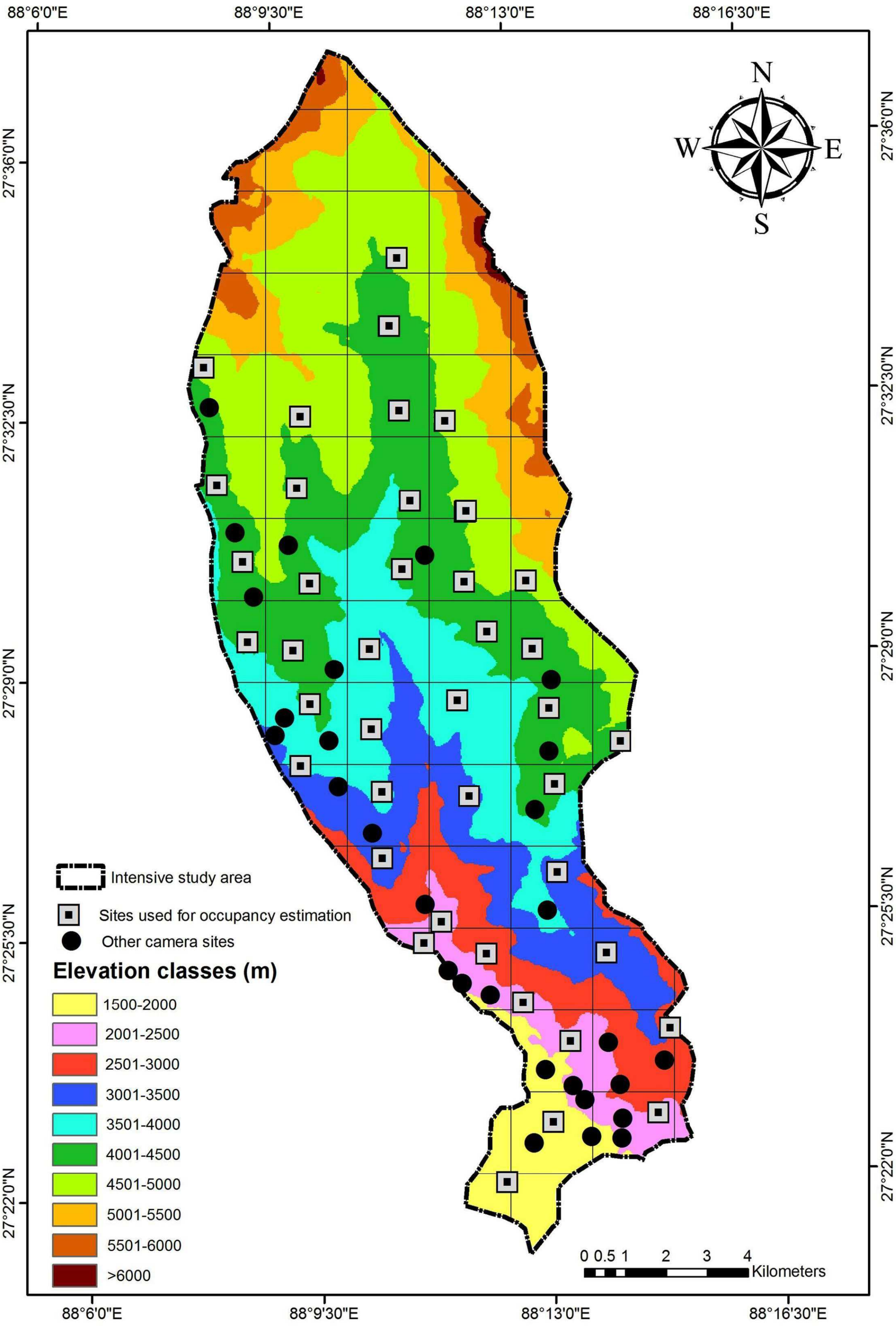
Based on the knowledge acquired through reconnaissance surveys conducted during 2008 in different catchments of Khangchendzonga BR, *Prek chu* catchment was selected for camera trap sampling (Sathyakumar et al. 2009) as mentioned in the previous chapter. Sign surveys and trail walking in the *Prek chu* catchment helped in identifying appropriate locations for camera deployment. The study area was divided into 4 km$^2$ blocks/grids/sites with the help of Geographic Information System (GIS) tools and the area was categorized into four different survey zones according to the habitats, *viz.*, temperate (1,200-3,000 m), sub-alpine (3,000-3,800 m), krummholtz (3,800-4,200 m) and alpine (above 4,200 m). Camera traps were deployed based on the area coverage of each survey zone and the accessibility of the grids [13 blocks in temperate, 8 blocks in subalpine, 8 blocks in krummholtz and 12 blocks in alpine]. Twenty seven cameras were deployed along trails as well as inside the forest at 71 locations in 41 blocks and for occupancy analysis data from only one camera was used for each block (Figure 5.1). Although, camera trapping was done continuously in the intensive study area, a maximum of three months of camera trapping data (survey period) was used for occupancy estimation of each species. This design was followed by considering the mean annual home range size of the individual species and assuming that the occupancy status of grids/sites remains constant (MacKenzie et al. 2006) during this survey period (Table 5.1). In case the occupancy status of sites change within the survey period (e.g., for large home range species snow leopard, black bear and golden cat) and these changes being random, the occupancy-based inference will still be valid provided the site occupancy is interpreted as the probability of site utilization (Bailey et al. 2007). Due to limitations of camera units (n = 27), camera traps were deployed for at least one month in each grid/site (n = 41).

The habitat characteristics at each camera location were assessed by laying a 10 m×10 m plot around the camera and information on site covariates such as, elevation (altimeter), aspect, slope (clinometer), habitat type, forest type, % tree/vegetation cover (including ground cover along with understorey and canopy cover- ocular estimation), presence of trekking trails, photo-capture rate of small (rodents, pika, squirrels, and galliformes) and medium-large (ungulates) sized wild prey expressed as RAI (photographs/100 trap days), and sampling/detection covariates such as, human presence and weather condition were recorded.

**Table 5.1 Home ranges (km²) of carnivore species mentioned in literature and survey periods for assessing their occupancy in the study area.**

| **Species** | **Mean home ranges (km²)** | **Literature** | **Occupancy Survey period** |
|---|---|---|---|
| Snow leopard | 11-451 | Chundawat 1992; Jackson 1996; Oli 1997; McCarthy et al. 2005 | Jan-March, 2010 |
| Red fox | 0.3-14.7 | Harrison et al. 1989; Cavallini & Lovari 1994 | May-July, 2009 |
| Yellow-throated marten | 0.7-10.5 | Heptner & Sludskii 2002; Grassman et al. 2005c | Sep-Nov, 2009 |
| Stoat | 0.051-2.08 | Cuthbert & Sommer 2002; Hellstedt & Henttonen 2006 | March-May, 2010 |
| Black bear | 7.2-117 | Izumiyama & Shiraishi 2004; Hwang et al. 2010; Sharma 2012 | Sep-Nov, 2009 |
| Golden cat | 32.6-47.7 | Grassman et al. 2005a | Nov, 2009-Jan, 2010 |
| Large Indian civet | 1.7-12 | Rabinowitz 1991 | Oct-Dec, 2009 |
| Masked palm civet | 1.1-5.9 | Rabinowitz 1991; Grassman 1997 | Oct-Dec, 2009 |
| Leopard cat | 1.5-14 | Grassman et al. 2005b; Rabinowitz 1990; Rajaratnam et al. 2007 | Oct-Dec, 2009 |

**Figure 5.1 Map showing locations of camera traps in 2×2 km grids for occupancy survey of carnivores in different habitats and elevation classes of *Prek chu* catchment**



#### ***5.2.1.1 Occupancy modelling***

Camera trap data being more reliable than the sign encounter data was used as an index of space use or habitat use by the carnivores based on their occupancy status in a given sampling grid. The sampling records at each site for each species were divided into six consecutive 5-day segments (occasions) based on the date stamp on the photographs as cameras were deployed at least for one month at each site within the sampling session of three months. For occupancy analysis, a detection matrix (0 = non-detection and 1 = detection) was created for each sampling site for each species. In order to avoid numerical optimization of the likelihood (Hines 2006) all numerical variables were z-standardized and categorical covariates were dummy-coded. A logit link function (Mackenzie et al. 2006) was used to model carnivore presence as dependent on habitat covariates in program PRESENCE version 4.0 (Hines 2006). Site occupancy rates and detection probability rates relative to 11 site covariates and two sampling/detection variables (Table 5.2) were estimated only for nine carnivore species (snow leopard, red fox, yellow-throated marten, stoat, black bear, golden cat, large Indian civet, masked palm civet and leopard cat), due to very low photo captures of other carnivores. Occupancy models were ranked based on minimum Akaike Information Criterion (AIC; Akaike 1973) values (AICc for small samples) and all the models with ΔAICc <2 were considered as equivalent models. Model fit was assessed through parametric bootstrap (10,000 iterations) to assess the probability of the observed chi-square statistics, and overdispersion parameter ($\hat{c}$) was used as a variance-inflation factor and to correct AIC for lack of fit. The summed model weight of each covariate in these equivalent models was used to determine the most significant variables influencing the occupancy of the species. The sign (positive or negative) and standard error of logistic coefficient ($\beta$) of each variable was used to determine the direction of influence of the variable. As broad-habitat types along the elevation gradient of the study area were selected as the major resource which governed the site occupancy of the carnivores in the *Prek chu* catchment, habitat wise site occupancy rates were also calculated and graphically represented (mean ± S.E) for each species. For species with large home-ranges (with high possibility of violating the closure assumption), site occupancy was interpreted and inferred only as the probability/intensity of site utilization.

**Table 5.2 Variables used for estimating the site occupancy rates and detection probabilities of carnivores in the occupancy model.**

| Abbreviation | Name | Description |
|---|---|---|
| **Sampling variables** | | |
| E | Elevation | Numeric (Range 1830–4520 m) |
| A | Aspect | Categorical [Warm – NE, E, SE, S (denoted as 0); Cold – N, NW, W, SW (denoted as 1)] |
| S | Slope | Categorical (steep > 30°≈ 0, gentle ≤ 30°≈ 1) |
| V | Vegetation cover (%) | Numeric (Range 0-80%) |
| Al | Alpine zone | Categorical (present, absent) |
| K | Krummholtz | Categorical (present, absent) |
| C | Conifer | Categorical (present, absent) |
| B | Broadleaved forest | Categorical (present, absent) |
| Up | Ungulate PCR | Numeric -RAI (0-0.775/100 trap days) |
| Sp | Small prey PCR | Numeric - RAI (0-0.968/100 trap days) |
| T | Trekking trail | Categorical (present, absent) |
| **Detection variables** | | |
| H | Human presence | Direct and indirect evidences, present/absent |
| R | Season | Rainy weather, present/absent |

### *5.2.1.2 Spatial overlap*

Model derived site occupancy estimate of each site (n = 41) conditional on detection history was used to calculate spatial overlap between carnivores. This site occupancy value was averaged for best selected models (ΔAICc <2) for each site for each species and the resulting averaged values were converted into zeros and ones (0 if $\psi < 0.5$ and 1 if $\psi > 0.5$) representing non-use and use of sites by the species. Spatial overlap between two species (say X and Y) was compared using the Jaccard similarity index (*S*; Magguran 2004) expressed as;

$$S = A/(A+B+C)$$

where *A* is the number of sites used by both species, *B* is the number of sites used only by species X, and *C* is number of sites used only by species Y. The value of the index ranges from 0 (no overlap) to 1 (complete overlap). The associated probability (*P*) for *S* was also calculated to determine if the value for the index differed from what would be expected at random (Real and Vargas 1996), using the following formula:

$$P(A,B,C) = \frac{N!}{A!B!C!} \times 3^{-N}$$

where $A$, $B$, and $C$ are the same as in the Jaccard similarity index and $N = A+B+C$.

### 5.2.2 Dietary analysis

Diet composition of carnivores assessed through scat analysis was expressed as frequency of occurrence (FO), defined as:

$$\mathbf{FO} = s\,\mathbf{(100)/N,}$$

where $s$ is the number of scat samples containing each prey type and N is the total number of scat samples, and as percent relative occurrence (RO), defined as:

$$\mathbf{RO} = p\,\mathbf{(100)/T,}$$

where $p$ is the number of occurrences of each prey type and T is total occurrences of all prey types in all scat samples of a carnivore, thus representing the relative importance of a food type in the diet of the carnivore (Reynolds and Aebischer 1991). Since, frequency of occurrence does not adequately represent the proportion of different body sized prey in the diet and consumption rate of small sized prey is overestimated in a carnivore's diet due to numerous hairs per unit body weight (Floyd et al. 1978; Ackerman et al. 1984; Weaver 1993). This bias was corrected by converting the frequency occurrence of different prey species in the scats to the percent relative biomass (Karanth and Sunquist 1995), as it provides the best approximation to the carnivore's true diet (Klare et al. 2011). This was done using few regression equations developed on carnivore and applying those on the study species of comparable body size and digestive physiology as below:

**$Y = 1.980 + 0.035X$** ([Ackerman et al. 1984]; used for snow leopards)

**$Y = 0.035 + 0.020X$** ([Floyd et al. 1978]; used for wild dogs)

**$Y^* = 6.76 + 0.013X$** ([Maurya et al. 2012]; used for red foxes)

**$\ln Y^* = 1.76 + 0.19\ln X$** ([Zielinski 1986]; used for yellow-throated marten) where, Y is the kilogram of prey consumed per field collectible scat (Y* is in grams) and X is the average weight of an individual of a particular prey type (Ackerman et al. 1984). Information about average body weights of prey species mentioned by Macdonald (2001) were used for this calculation. Multiplying each Y by the number of scats found to contain a particular prey species gave the relative weight of each prey type consumed. These values were used to estimate percent relative biomass (RB) contribution of different prey species to the carnivore diet (Klare et al. 2011).

Both percent relative occurrence (RO) and percent relative biomass (RB) of food categories in the diet of carnivores were used to calculate their niche breadths and dietary overlaps. To estimate the trophic niche breadths of carnivores standardized Levin's index ($B_{\text{sta}}$) (Levins 1968; Colwell and Futuyama 1971) was used. The Levins' index formula is:

$$B = \frac{1}{\sum_{i=1}^{n} p_i^2}$$

where $n$ is the number of food categories and $p$ is the proportion of records in each food category ($i$) set at 100%. The standardized Levins' index is calculated as:

$$\boldsymbol{B_{\text{sta}} = (\mathrm{B} - 1)/(B_{\text{max}} - 1)}$$

where $B$ is the Levins' index (Levins 1968; Krebs 1989) and $B_{\text{max}}$ is the total number of food categories recognized. The index values range from 0 to 1.

Dietary overlap among carnivore species was evaluated using the niche overlap index ($O$) (Pianka 1973). This index also varies between 0 (complete separation) to 1 (complete overlap). The Pianka index is calculated using formula as:

$$O_{jk} = \frac{\sum_{i=1}^{n} p_{ij} p_{ik}}{\sqrt{\sum_{i=1}^{n} p_{ij}^2 \sum_{i=1}^{n} p_{ik}^2}}$$

where $p_{ij}$ (or $p_{ik}$) is the proportion of food category $i$ recorded in the diet of the species $j$ (or $k$). To test for significance of niche overlap EcoSim 7 software (Gotelli and

Entsminger 2006) was used by comparing observed values with values obtained from 10,000 random iterations of the original matrices.

In order to depict prey selection by carnivores, Jacobs' selectivity index (Jacobs 1974) was used to measure the preference for certain species during each phase of the predatory behavioural repertoire. Although there are numerous selectivity indices (Krebs 1989), Jacobs' index is one of the better ones as it minimizes several of the problems that afflict many other indices, such as non-linearity, bias to rare food items, increasing confidence intervals with increasing heterogeneity, unbound or undefined nature, and lack of symmetry between selected and rejected values (Jacobs 1974; Strauss 1979; Norbury and Sanson 1992). In accordance with previous studies conducted on the prey preferences of large predators (Hayward et al. 2006ab; 2011; 2012), Jacobs' index standardizes the relationship between the relative proportion that each species makes up of the carnivores diet $r$ and prey relative abundance $p$ (i.e. the proportion that each species makes up of the total abundance of all censused prey species at a site). The formula for Jacobs' index is:

$$D = \frac{(r_i - p_i)}{(r_i + p_i - 2r_i p_i)}$$

where $r_i$ is the relative biomass proportion of prey species $i$ in the carnivore scats and $p_i$ is the proportion of biomass of the prey species $i$ in the available prey community. The resulting values range from +1 to -1, where +1 indicates maximum preference and -1 indicates maximum avoidance. Prey selectivity was then assessed by chi-square ratio test between the observed (biomass consumed) and expected (biomass available) contribution by each prey species. Proportion of prey availability was calculated from the camera trap data as relative abundance expressed in terms of number of individuals captured/100 trap days, considering independent captures at 1 hour duration (Bowkett et al. 2007). This index was transformed into available biomass by multiplying it with the individual body weights of prey species, i.e., 3/4 × mean adult female body mass of prey species was used in order to take account of calves and subadults eaten (Hayward and Kerley 2005).

### 5.2.3 Temporal activity

The activity pattern of carnivores was assessed based on the date and time information on the camera trap photographs assuming that the numbers of photographs taken were correlated with the carnivore activity levels (Kawanishi 2002). Capture events of species at a site were considered as independent records at minimum 1 hour duration as mentioned earlier (Bowkett et al. 2007). Due to the low sample size, all such independent records or photos of target species were pooled for the entire study period to establish the overall activity pattern for each species. All independent time records of species for every hour were converted into percent photographic captures in that particular hour for 24 hrs and mean activity period of carnivores was determined using the program Oriana 4.0 (Kovach 2011). The percentage of activity level was used to indicate whether the carnivore is nocturnal or diurnal. Nocturnal activity was categorized as that which occurred from 1800 to 0500 hrs, the approximate times of sunset and sunrise in the area during the study period. Rayleigh Test was applied to determine the uniformity in the activity pattern of each carnivore. The differences within the activity pattern of a species and with other carnivore species were also tested through Watson's $U^2$ Test (for single species and pair wise) in program Oriana 4.0 (Kovach 2011).

The percent photographic captures of species for each hour were further analysed using Pianka niche overlap index (Pianka 1973) to assess temporal overlap between their activity patterns (pair wise) in software TimeOverlap version 1.0 (Castro-Arellano et al. 2010). TimeOverlap is a simulation program that evaluates the amount of overlap among a number of distributions. The program represents a species-based approach; as such the program employs a randomization algorithm (ROSARIO) that performs random shifts of entire distributions (i.e., in contrast to other simulation approaches ROSARIO preserves much of the temporal autocorrelation inherent in activity patterns) within the time frame of analyses (Castro-Arellano et al. 2010). For each randomization, the program calculates the amount of overlap of each group of randomized distributions for each iteration; and its significance is determined by comparison of randomized overlap values to the amount of empirical overlap. The analysis was run at 10,000 iterations at 0.05 level of statistical significance.

## 5.3 RESULTS

### 5.3.1 Occupancy modelling

#### *5.3.1.1 Species-wise occupancy modelling of carnivores*

Snow leopard was detected in 9 while red fox in 12 out of total 41 sites. The estimated site occupancy/probability of site utilization in case of snow leopard (Ψ = 0.2204±0.068) was slightly higher than the naive estimates (Ψ Naïve = 0.2195 [without including detection probability]). The estimated detection probability (p) was 0.265±0.0684. The site occupancy of red fox (Ψ = 0.294±0.075) was also slightly higher than its naïve estimate (Ψ Naïve = 0.2927) with detection probability (p) equal to 0.481±0.077. Occupancy modelling showed that detection probability of both snow leopard and red fox was not influenced by human presence at camera sites (Table 5.3a). Elevation, vegetation cover and presence of alpine habitat were determined best predictors of snow leopard occupancy with vegetation cover showing negative influence (Table 5.3b). Whereas, presence of *krummholtz* habitat and trekking trails were determined as the best predictors' positively influencing red fox occupancy in the area.

**Table 5.3a Top models for predicting site occupancy (Est. Ψ) and detection probability (Est. P) of snow leopard and red fox in *Prek chu* catchment (*k* = number of parameters, ĉ = overdispersion parameter and *p* = significance for chi-square test (derived from 10,000 parametric bootstraps)**

| **Models** | **AIC** | **Δ AIC** | ***k*** | **Ψ (S.E)** | **P (S.E)** | **ĉ** | ***p*** |
|---|---|---|---|---|---|---|---|
| **Snow leopard** | | | | | | | |
| ψ(EUp),p(.) | 104.64 | 0 | 4 | 0.221 (0.072) | 0.271 (0.069) | 1.079 | 0.254 |
| ψ(EAl),p(.) | 104.86 | 0.22 | 4 | 0.22 (0.064) | 0.264 (0.068) | 1.083 | 0.268 |
| ψ(EV),p(.) | 104.86 | 0.22 | 4 | 0.22 (0.065) | 0.267 (0.068) | 1.075 | 0.274 |
| ψ(EUpAl),p(.) | 106.36 | 1.72 | 5 | 0.221 (0.073) | 0.259 (0.069) | 1.082 | 0.255 |
| ψ(EVAl),p(.) | 106.52 | 1.88 | 5 | 0.22 (0.066) | 0.264 (0.068) | 1.07 | 0.285 |
| **Red fox** | | | | | | | |
| ψ(ETK),p(R) | 125.38 | 0 | 6 | 0.294 (0.075) | 0.451 (0.094) | 0.845 | 0.825 |
| ψ(ETK),p(.) | 125.52 | 0.14 | 5 | 0.294 (0.075) | 0.510 (0.06) | 0.844 | 0.825 |
| ψ(TAlK),p(.) | 127.02 | 1.64 | 5 | 0.295 (0.069) | 0.509 (0.06) | 0.845 | 0.845 |
| ψ(ETAlK),p(R) | 127.16 | 1.78 | 7 | 0.294 (0.075) | 0.451 (0.095) | 0.842 | 0.840 |
| ψ(ETAlK),p(.) | 127.30 | 1.92 | 6 | 0.295 (0.076) | 0.510 (0.060) | 0.840 | 0.824 |
| ψ(EVTK),p(R) | 127.37 | 1.99 | 7 | 0.294 (0.083) | 0.451 (0.095) | 0.834 | 0.835 |

**Table 5.3b Summed model weight ($\sum w$), Average β value and sign of influence [positive (+), negative (-) and (*) if significant] of each sampling variable in the equivalent models for snow leopard and red fox**

| | **Snow leopard** | | **Red fox** | |
|---|---|---|---|---|
| **Sampling variables** | **$\sum w$** | **β (SE)** | **$\sum w$** | **β (SE)** |
| Elevation (E) | 0.866 | + 6.51 (2.75)* | 0.718 | + 6.16 (4.25) |
| Slope (S) | NA | NA | NA | NA |
| Tree % (V) | 0.2994 | - 19.28 (7.63)* | 0.0858 | + 0.188 (2.15) |
| Aspect (A) | NA | NA | NA | NA |
| Trekking trail (T) | NA | NA | 0.8208 | + 3.49 (1.6)* |
| Alpine (Al) | 0.4028 | + 24.05 (5.96)* | 0.2862 | + 26.7 (15.06) |
| Krummholtz (K) | NA | NA | 0.8208 | + 16.78 (6.9)* |
| Conifer (C) | NA | NA | NA | NA |
| Ungulate prey (Up) | 0.3477 | - 0.92 (1.45) | NA | NA |
| Other prey (O) | NA | NA | NA | NA |
| **Detection variables** | | | | |
| Human presence (H) | NA | NA | NA | NA |
| Season (R) | NA | NA | 0.4129 | + 1.035 (0.734) |

Yellow-throated marten was detected in 13 and stoat in 9 sites. The estimated site occupancies of both yellow-throated marten (Ψ = 0.348±0.106) and stoat (Ψ = 0.314±0.14) were higher than their respective naive estimates (Ψ Naïve = 0.3174) and (Ψ Naïve = 0.2195). The estimated detection probability (p) of yellow-throated marten was 0.368±0.0753 while that of stoat was 0.169±0.057, respectively. Occupancy modelling showed that the detection probability of both the species was not influenced by weather condition (Table 5.4a), but was positively influenced by human presence at camera sites for yellow-throated marten. Vegetation cover and presence of trekking trails were determined as best predictors for the site occupancy of yellow-throated marten with former showing positive and later negative influence (Table 5.4b). In case of stoat elevation, presence of coniferous forest and vegetation cover were determined as best predictors positively influencing on its site occupancy.

**Table 5.4a Top models for predicting site occupancy (Est. Ψ) and detection probability (Est. P) of yellow-throated marten and stoat in *Prek chu* catchment (*k* = number of parameters, ĉ = overdispersion parameter and *p* = significance for chi-square test (derived from 10,000 parametric bootstraps)**

| Models | AIC | Δ AIC | *k* | Ψ (S.E) | P (S.E) | ĉ | *p* |
|---|---|---|---|---|---|---|---|
| **Yellow-throated marten** | | | | | | | |
| ψ(VTK),p(H) | 135.47 | 0 | 6 | 0.347 (0.097) | 0.369 (0.075) | 0.992 | 0.445 |
| ψ(EVTK),p(H) | 137.11 | 1.64 | 7 | 0.350 (0.109) | 0.367 (0.075) | 1.006 | 0.428 |
| ψ(VTKB),p(H) | 137.46 | 1.99 | 7 | 0.346 (0.111) | 0.369 (0.076) | 0.986 | 0.453 |
| **Stoat** | | | | | | | |
| ψ(AC),p(.) | 94.71 | 0 | 4 | 0.301 (0.13) | 0.176 (0.058) | 1.126 | 0.278 |
| ψ(EV),p(.) | 96.38 | 1.67 | 4 | 0.340 (0.14) | 0.157 (0.054) | 1.165 | 0.212 |
| ψ(AAlC),p(.) | 96.71 | 2.00 | 5 | 0.300 (0.15) | 0.175 (0.059) | 1.177 | 0.275 |

**Table 5.4b Summed model weight (∑*w*), Average β value and sign of influence [positive (+), negative (-) and (*) if significant] of each sampling variable in the equivalent models for yellow-throated marten and stoat**

| | **Yellow-throated marten** | | **Stoat** | |
|---|---|---|---|---|
| **Sampling variables** | **∑ *w*** | **β (SE)** | **∑ *w*** | **β (SE)** |
| Elevation (E) | 0.1872 | - 0.605 (1.05) | 0.1336 | + 4.61 (2.13)* |
| Slope (S) | NA | NA | NA | NA |
| Tree % (V) | 0.7693 | + 3.97 (1.55)* | 0.1336 | + 3.95 (1.96)* |
| Aspect (A) | NA | NA | 0.4212 | + 1.88 (1.25) |
| Trekking trail (T) | 0.7693 | - 4.023 (1.89)* | NA | NA |
| Broadleaved (B) | 0.1571 | - 0.160 (1.51) | NA | NA |
| Conifer (C) | NA | NA | 0.4212 | + 3.416 (1.59)* |
| Krummholtz (K) | 0.7693 | + 7.09 (3.86) | NA | NA |
| Alpine (Al) | NA | NA | 0.1133 | - 0.09 (1.245) |
| Ungulate prey (U) | NA | NA | NA | NA |
| Other prey (O) | NA | NA | NA | NA |
| **Detection variables** | | | | |
| Human presence (H) | 0.7693 | + 1.19 (0.485)* | NA | NA |
| Season (R) | NA | NA | NA | NA |

The presence of Asiatic black bear and golden cat was detected at 7 and 8 sites, respectively. The estimated site occupancies/probabilities of site utilization (Ψ = 0.286±0.161) and (Ψ = 0.263±0.142) for both the species were higher than their respective naive estimates (Ψ Naïve = 0.1707) and (Ψ Naïve = 0.1915). The estimated detection probability (p) of black bear was 0.142±0.0712 while that of golden cat was 0.155±0.059. Occupancy modelling showed that the detection probability of both the species was not influenced either by human presence or weather condition (Table 5.5a). Presence of broadleaved and coniferous forests were determined as the best predictors positively influencing the probability of site utilization by black bear, while the probability of site utilization by golden cat was best predicted by high vegetation cover (Table 5.5b).

**Table 5.5a Top models for predicting site occupancy (Est. Ψ) and detection probability (Est. P) of black bear and golden cat in *Prek chu* catchment (*k* = number of parameters, ĉ = overdispersion parameter and *p* = significance for chi-square test (derived from 10,000 parametric bootstraps)**

| **Models** | **AIC** | **Δ AIC** | ***k*** | **Ψ (S.E)** | **P (S.E)** | **ĉ** | ***p*** |
|---|---|---|---|---|---|---|---|
| **Black bear** | | | | | | | |
| ψ(V),p(.) | 74.13 | 0 | 3 | 0.288 (0.162) | 0.140 (0.075) | 0.596 | 0.635 |
| ψ(VC),p(.) | 74.87 | 0.74 | 4 | 0.283 (0.165) | 0.144 (0.071) | 0.620 | 0.640 |
| ψ(ECB),p(.) | 75.87 | 1.74 | 5 | 0.283 (0.142) | 0.143 (0.061) | 0.557 | 0.628 |
| ψ(EV),p(.) | 76.09 | 1.96 | 4 | 0.295 (0.189) | 0.138 (0.077) | 0.653 | 0.578 |
| ψ(CB),p(.) | 76.12 | 1.99 | 4 | 0.279 (0.151) | 0.145 (0.072) | 0.648 | 0.600 |
| **Golden cat** | | | | | | | |
| ψ(VB),p(.) | 78.24 | 0 | 4 | 0.259 (0.124) | 0.157 (0.059) | 0.552 | 0.714 |
| ψ(VAB),p(.) | 78.38 | 0.14 | 5 | 0.252 (0.134) | 0.161 (0.06) | 0.562 | 0.719 |
| ψ(VAC),p(.) | 79.61 | 1.37 | 5 | 0.277 (0.16) | 0.147 (0.059) | 0.561 | 0.729 |
| ψ(EV),p(.) | 79.9 | 1.66 | 4 | 0.266 (0.13) | 0.153 (0.058) | 0.535 | 0.747 |
| ψ(VSAB),p(.) | 79.94 | 1.7 | 6 | 0.254 (0.147) | 0.159 (0.059) | 0.542 | 0.737 |
| ψ(EVA),p(.) | 80.04 | 1.8 | 5 | 0.276 (0.159) | 0.147 (0.061) | 0.530 | 0.744 |
| ψ(EVB),p(.) | 80.19 | 1.95 | 5 | 0.256 (0.14) | 0.158 (0.06) | 0.537 | 0.734 |

**Table 5.5b Summed model weight (∑*w*), Average β value and sign of influence [positive (+), negative (-) and (*) if significant] of each sampling variable in the equivalent models for black bear and golden cat**

| | **Black Bear** | | **Golden Cat** | |
|---|---|---|---|---|
| **Sampling variables** | **∑ *w*** | **β (SE)** | **∑ *w*** | **β (SE)** |
| Elevation (E) | 0.2125 | - 2.8 (2.92) | 0.2121 | + 1.76 (1.98) |
| Slope (S) | NA | NA | 0.0743 | + 0.57 (0.88) |
| Tree % (V) | 0.5527 | + 2.44 (1.69) | 0.7099 | + 2.81 (1.28)* |
| Aspect (A) | NA | NA | 0.3947 | + 1.98 (1.57) |
| Trekking trail (T) | NA | NA | NA | NA |
| Broadleaved (B) | 0.2007 | + 19.67 (3.9)* | 0.4758 | - 4.04 (2.28) |
| Conifer (C) | 0.3855 | + 24.14 (5.83)* | 0.0876 | + 2.42 (1.65) |
| Krummholtz (K) | NA | NA | NA | NA |
| Ungulate prey (U) | NA | NA | NA | NA |
| Other prey (O) | NA | NA | NA | NA |
| **Detection variables** | | | | |
| Human presence (H) | NA | NA | NA | NA |
| Season (R) | NA | NA | NA | NA |

Among the two civet species, the presence of large Indian civet was detected at 6 sites while masked palm civet was detected at 7 sites. The estimated site occupancies of both civet species, large Indian civet (Ψ = 0.157±0.0576) and masked palm civet (Ψ = 0.182±0.0747) were higher than their respective naive estimates (Ψ Naïve = 0.1463) and (Ψ Naïve = 0.1707). The estimated detection probability (p) of large Indian civet 0.495±0.130 was higher than that of masked palm civet 0.334±0.0792. The detection probability of large Indian civet was negatively influenced by human presence contrary to that of masked palm civet which was not influenced by either of the detection variables (Table 5.6a). Elevation was determined as the best predictor negatively influencing the site occupancy of both the civets (Table 5.6b), but in addition presence of broadleaved forest also showed positive influence on the site occupancy of large Indian civet.

**Table 5.6a Top models for predicting site occupancy (Est. Ψ) and detection probability (Est. P) of large Indian civet and masked palm civet in *Prek chu* catchment (*k* = number of parameters, ĉ = overdispersion parameter and *p* = significance for chi-square test (derived from 10,000 parametric bootstraps)**

| **Models** | **AIC** | **Δ AIC** | ***k*** | **Ψ (S.E)** | **P (S.E)** | **ĉ** | ***p*** |
|---|---|---|---|---|---|---|---|
| **Large Indian civet** | | | | | | | |
| ψ(SpB),p(H) | 65.33 | 0 | 5 | 0.149 (0.052) | 0.501 (0.120) | 1.217 | 0.164 |
| ψ(E),p(HR) | 65.42 | 0.09 | 5 | 0.158 (0.051) | 0.495 (0.138) | 1.47 | 0.065 |
| ψ(EV),p(HR) | 65.64 | 0.31 | 6 | 0.160 (0.052) | 0.493 (0.138) | 1.55 | 0.062 |
| ψ(EB),p(H) | 65.74 | 0.41 | 5 | 0.157 (0.051) | 0.493 (0.119) | 1.67 | 0.019 |
| ψ(VSpT),p(H) | 65.98 | 0.65 | 6 | 0.149 (0.066) | 0.501 (0.120) | 1.77 | 0.009 |
| ψ(ET),p(HR) | 66.29 | 0.96 | 6 | 0.159 (0.063) | 0.494 (0.138) | 1.56 | 0.057 |
| ψ(ES),p(HR) | 66.9 | 1.57 | 6 | 0.164 (0.068) | 0.490 (0.138) | 1.53 | 0.062 |
| **Masked palm civet** | | | | | | | |
| ψ(ESp),p(.) | 78.30 | 0 | 4 | 0.179 (0.068) | 0.340 (0.077) | 1.726 | 0.018 |
| ψ(EB),p(.) | 79.13 | 0.83 | 4 | 0.186 (0.074) | 0.327 (0.081) | 1.837 | 0.019 |
| ψ(ESpT),p(.) | 80.12 | 1.82 | 5 | 0.178 (0.079) | 0.342 (0.077) | 1.742 | 0.015 |
| ψ(ESSp),p(.) | 80.13 | 1.83 | 5 | 0.178 (0.077) | 0.341 (0.077) | 1.708 | 0.024 |
| ψ(ESB),p(.) | 80.20 | 1.90 | 5 | 0.187 (0.084) | 0.326 (0.082) | 1.812 | 0.020 |
| ψ(E),p(.) | 80.22 | 1.92 | 3 | 0.187 (0.066) | 0.325 (0.081) | 1.857 | 0.020 |

**Table 5.6b Summed model weight (∑*w*), Average β value and sign of influence [positive (+), negative (-) and (*) if significant] of each sampling variable in the equivalent models for large Indian civet and masked palm civet**

| | **Large Indian civet** | | **Masked palm civet** | |
|---|---|---|---|---|
| **Sampling variables** | **∑ *w*** | **β (SE)** | **∑ *w*** | **β (SE)** |
| Elevation (E) | 0.6795 | - 5.9 (2.77)* | 0.7113 | - 4.12 (1.97)* |
| Slope (S) | 0.0837 | + 0.78 (1.22) | 0.1732 | + 0.253 (1.06) |
| Tree % (V) | 0.2898 | + 4.81 (4.88) | NA | NA |
| Aspect (A) | NA | NA | NA | NA |
| Trekking trail (T) | 0.2462 | + 2.28 (1.85) | 0.0886 | + 0.57 (1.35) |
| Broadleaved (B) | 0.333 | + 23.05 (6.27)* | 0.2304 | + 7.12 (4.54) |
| Conifer (C) | NA | NA | NA | NA |
| Ungulate prey (U) | NA | NA | NA | NA |
| Other prey (Sp) | 0.3161 | + 3.27 (2.72) | 0.3967 | + 2.35 (1.95) |
| **Detection variables** | | | | |
| Human presence (H) | 0.995 | - 1.96 (0.842)* | NA | NA |
| Season (R) | 0.530 | + 0.57 (0.98) | NA | NA |

Leopard cat presence was detected in 12 sites. The estimated site occupancy (Ψ = 0.327±0.0577) was slightly higher than its naive estimates (Ψ Naïve = 0.2927) and the estimated detection probability (p) was 0.313±0.0698 (Table 5.7a). Occupancy modelling showed that detection probability of leopard cat was not significantly influenced by human presence and weather condition. Elevation, abundance of small mammal and galliformes prey, and presence of broadleaved forest were determined as best predictors for the site occupancy of leopard cat (Table 5.7b). Both prey abundance and broadleaved forest had positive influence, while elevation had a negative influence on the site occupancy of leopard cat.

**Table 5.7a Top models for predicting site occupancy (Est. Ψ) and detection probability (Est. P) of leopard cat in *Prek chu* catchment (*k* = number of parameters, ĉ = overdispersion parameter and *p* = significance for chi-square test (derived from 10,000 parametric bootstraps)**

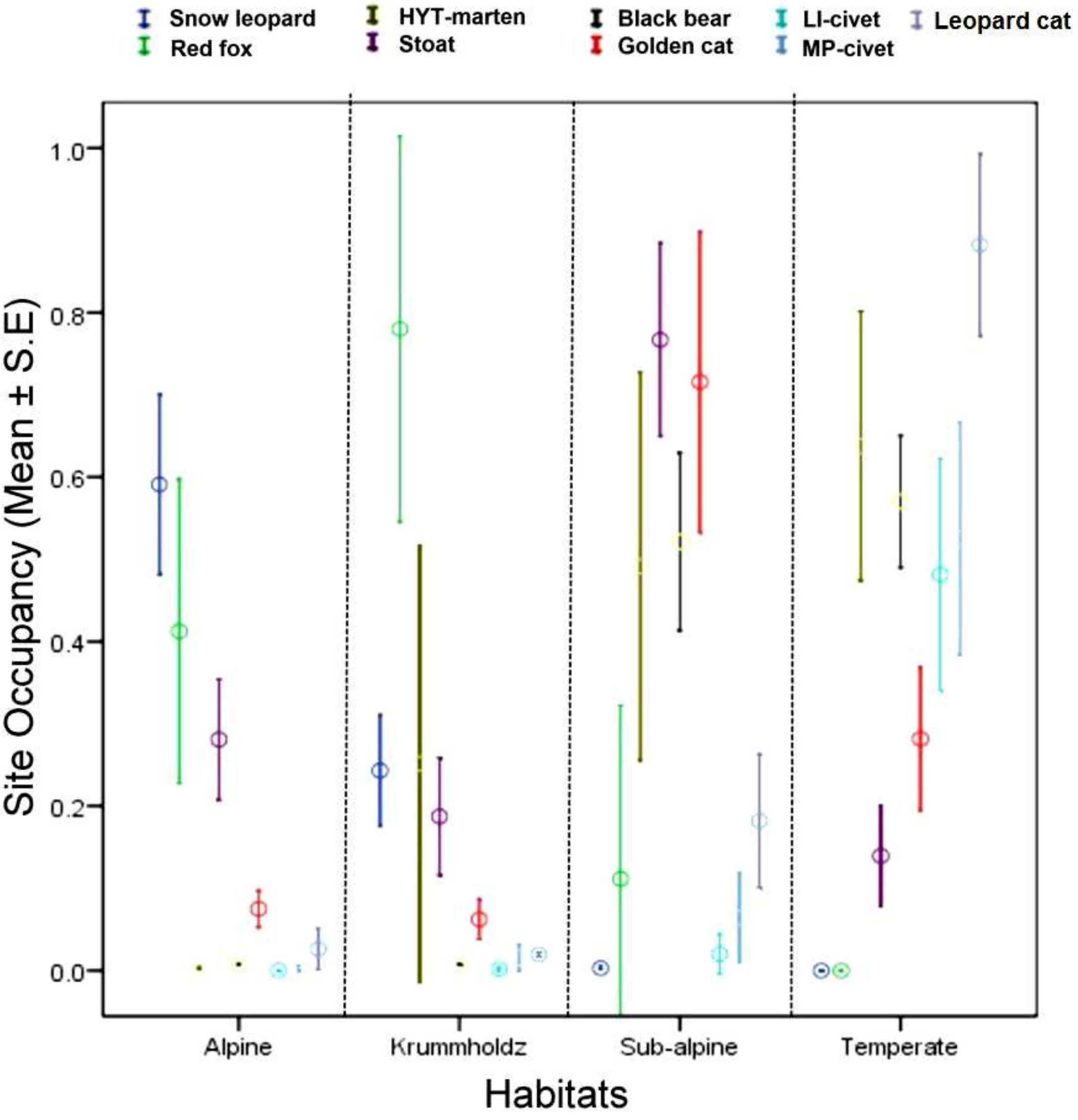
| Models | AIC | Δ AIC | *k* | Ψ (S.E) | P (S.E) | ĉ | *p* |
|---|---|---|---|---|---|---|---|
| **Leopard cat** | | | | | | | |
| ψ(EO),p(R) | 115.25 | 0 | 5 | 0.345 (0.0357) | 0.297 (0.062) | 1.137 | 0.258 |
| ψ(OB),p(R) | 115.75 | 0.5 | 5 | 0.312 (0.0736) | 0.324 (0.071) | 0.949 | 0.529 |
| ψ(EB),p(HR) | 116.13 | 0.88 | 6 | 0.325 (0.0504) | 0.317 (0.085) | 1.013 | 0.405 |
| ψ(EVO),p(R) | 116.69 | 1.44 | 6 | 0.327 (0.0612) | 0.313 (0.066) | 1.063 | 0.335 |
| ψ(EVTB),p(R) | 117.12 | 1.87 | 7 | 0.325 (0.0674) | 0.315 (0.065) | 1.033 | 0.372 |

**Table 5.7b Summed model weight (∑*w*), Average β value and sign of influence [positive (+), negative (-) and (*) if significant] of each sampling variable in the equivalent models for leopard cat**

| | **Leopard Cat** | |
|---|---|---|
| **Sampling variables** | ∑ ***w*** | **β (SE)** |
| Elevation (E) | 0.6860 | - 3.78 (1.56)* |
| Slope (S) | NA | NA |
| Tree % (V) | 0.1539 | + 1.98 (1.43) |
| Aspect (A) | NA | NA |
| Trekking trail (T) | 0.0699 | + 1.15 (1.7) |
| Broadleaved (B) | 0.3768 | + 4.86 (2.3)* |
| Conifer (C) | NA | NA |
| Ungulate prey (U) | NA | NA |
| Other prey (O) | 0.6542 | + 2.52 (1.12)* |
| **Detection variables** | | |
| Human presence (H) | 0.1344 | - 0.317 (0.506) |
| Season (R) | 0.8585 | - 1.42 (0.803) |

Moreover, habitat wise site occupancy rates represented graphically (mean ± S.E) for each species (Figure 5.2) showed that highest site occupancy/probability of site utilization in the alpine habitat was observed for snow leopard (0.591±0.0546). In *krummholtz*, sub-alpine and temperate habitats highest site occupancies were observed for red fox (0.78±0.117), stoat (0.767±0.0587) and leopard cat (0.882±0.055), respectively. Correspondingly, the overall site utilization of snow leopard was restricted mainly within the alpine habitat. Although, presence of red fox was detected in alpine, *krummholtz* and sub-alpine habitats its site occupancy was high in *krummholtz* habitat. Site occupancy of yellow-throated marten was restricted mainly to temperate and sub-alpine habitats but its value was higher for temperate habitat. In case of stoat and golden cat, their site occupancies were high and restricted mostly to sub-alpine habitat. Site occupancy of black bear was concentrated mainly in temperate and sub-alpine habitats. For both civet species including leopard cat, site occupancy was high and restricted primarily to temperate habitat.

**Figure 5.2 Graphical representation of site occupancy (mean ± standard error) of carnivores in different habitat categories of *Prek chu* catchment**



### *5.3.1.2 Spatial overlap*

Space use by carnivores in the study area seemed to be mainly influenced by altitude and habitat specificity as mentioned earlier. The Jaccard's similarity index indicated very less spatial overlap between most of the species depicting substantial segregation in their space use. However, considerable degree of overlap was also observed among few species groups (Table 5.8). Highest spatial overlap was recorded between the two civets ($S = 0.9$; $P = 0.00017$) followed by large Indian civet-leopard cat ($S = 0.692$; $P = 0.00045$) and black bear-masked palm civet ($S = 0.67$; $P = 0.00021$), respectively.

**Table 5.8 Spatial overlap between carnivores based on site occupancies conditional on detection histories.**

| **Species** | **Jaccard Similarity Index (*S*)** | **Probability (*P*)** |
|---|---|---|
| Snow leopard-Red fox | 0.3125 | 0.03348 |
| Snow leopard-HYT marten | 0.0435 | 0.00007 |
| Snow leopard-Stoat | 0.1765 | 0.01581 |
| Red fox-HYT marten | 0.08 | 0.0004 |
| Red fox-Stoat | 0.095 | 0.00185 |
| HYT marten-Stoat | 0.19045 | 0.01112 |
| Red fox-Black bear | 0 | 0.000036 |
| Red fox-Golden cat | 0 | 0.000036 |
| Red fox-Leopard cat | 0 | 0.0000096 |
| HYT marten-Black bear | 0.47 | 0.0036 |
| HYT marten-Golden cat | 0.375 | 0.0083 |
| HYT marten-Leopard cat | 0.474 | 0.016 |
| Stoat-Black bear | 0.1176 | 0.00527 |
| Stoat-Golden cat | 0.4615 | 0.0226 |
| Stoat-Leopard cat | 0 | 0.0000014 |
| Black bear-Golden cat | 0.2307 | 0.0452 |
| Black bear-Leopard cat | 0.6471 | 0.00144 |
| Golden cat-Leopard cat | 0.11111 | 0.0031 |
| HYT marten-Large IC | 0.41176 | 0.0067 |
| HYT marten-Masked PC | 0.4706 | 0.0067 |
| Stoat-Large IC | 0 | 0.000009 |
| Stoat-Masked PC | 0.0588 | 0.00105 |
| Black bear-Large IC | 0.6 | 0.00035 |
| Black bear-Masked PC | 0.67 | 0.00021 |
| Golden cat-Large IC | 0.16667 | 0.02608 |
| Golden cat-Masked PC | 0.25 | 0.0521 |
| Large IC-Masked PC | 0.9 | 0.00017 |
| Large IC-Leopard cat | 0.692 | 0.00045 |
| Masked PC-Leopard cat | 0.643 | 0.0021 |

### 5.3.2 Diet

During trail monitorings and sign surveys in different habitats of the *Prek chu* catchment, a total of 827 scat samples were collected. Among these only 695 samples could be assigned species identities based on their morphometery, substrate, deposition pattern and habitat of occurrence. The remaining 132 samples were left unidentified. The identified scats belonged to seven species included snow leopard, red fox, yellow-throated marten, wild dog, leopard cat, Asiatic black bear and golden jackal [Chapter 3 (Table 3.4)]. Due to very low sample size for black bear (n = 7), detailed analysis and inference on its diet could not be made contrary to that of other six species mentioned below.

#### *5.3.2.1 Food habits*

#### A) Snow leopard

Food habit of snow leopard based on the analysis of 114 scats revealed the occurrence of eight prey items in its diet. Blue sheep was the most frequently occurring food item in the scats both in terms of frequency of occurrence (44.7%) and relative occurrence (30.7%) followed by pika, dzo and Himalayan tahr, while in terms of relative biomass dzo contributed maximum (43.8%) followed by blue sheep (28.5%) and Himalayan tahr (14.9%) to the total biomass consumed by the snow leopard (Table 5.9). Jacob’s selectivity index indicated that Himalayan tahr and pika were utilized more than their availability; blue sheep, birds (galliformes) and musk deer were utilized in accordance to their availability, while dzo was utilized less than its availability by snow leopard (Table 5.10).

#### B) Red fox

Diet of red fox revealed after analysing 367 scats indicated the occurrence of nine food items. The frequency of occurrence and relative occurrence of pika (59.9%, 31.5%) was highest in the scats followed by rodent and dung beetles. The relative prey biomass consumed by red fox was also found to be contributed primarily by pika (35.6%) followed rodents (23.6%) and dung beetles (22.6%; Table 5.9). According to the prey selectivity index dzo, blue sheep and Himalayan tahr were utilized less than their availability, while pika, rodents and birds (galliformes) were utilized more (Table 5.10).

### C) Yellow-throated marten

Analysis of 105 scats of yellow-throated marten revealed that its dietary spectrum consisted of 10 different prey items. The frequency of occurrence and relative occurrence of rodents was highest (56.2%, 35.5%) in the scats followed by pika, birds (galliformes) and seeds (fruit remains). The contribution of biomass to the total prey biomass consumed by yellow-throated marten was also maximum for rodent (39.3%) followed by pika (22.3%), galliformes (17.5%) and goral (6.7%), respectively (Table 5.9). Prey selection by yellow-throated marten based on the Jacob's index indicated that rodents, birds (galliformes) and pikas were utilized more that their availability; dzo, Himalayan tahr, barking deer and serow were utilized less, while goral was utilized in accordance to its availability in the area (Table 5.10).

**Table 5.9 Percent frequency of occurrence (FO), relative occurrence (RO) and relative biomass consumed (RB), of prey species in the scats of snow leopard (n=114), red fox (n=367) and yellow-throated marten (n=105).**

| **Prey species** | **Snow leopard** | | | **Red fox** | | | **Yellow-throated marten** | | |
|---|---|---|---|---|---|---|---|---|---|
| | **FO** | **RO** | **RB** | **FO** | **RO** | **RB** | **FO** | **RO** | **RB** |
| Blue sheep | 44.7 | 30.7 | 28.5 | 15.0 | 7.88 | 9.49 | - | - | - |
| Himalayan tahr | 17.5 | 12.0 | 14.9 | 1.91 | 1.03 | 1.25 | 0.95 | 0.6 | 1.62 |
| Musk deer | 3.5 | 2.46 | 0.84 | - | - | - | - | - | - |
| Serow | - | - | - | - | - | - | 2.86 | 1.82 | 5.12 |
| Goral | - | - | - | - | - | - | 4.76 | 3.02 | 6.7 |
| Barking deer | - | - | - | - | - | - | 0.95 | 0.6 | 1.3 |
| Dzo | 24.6 | 16.9 | 43.8 | 4.36 | 2.29 | 3.58 | 2.86 | 1.82 | 6.16 |
| Pika | 25.3 | 17.5 | 10.2 | 59.9 | 31.5 | 35.6 | 34.3 | 21.7 | 22.3 |
| Rodent | - | - | - | 39.8 | 20.9 | 23.6 | 56.2 | 35.5 | 39.3 |
| UF-mammal | 10.5 | 7.23 | - | 7.1 | 3.74 | - | 12.4 | 7.84 | - |
| Bird | 4.4 | 3.01 | 1.76 | 6.54 | 3.44 | 3.88 | 22.9 | 14.5 | 17.5 |
| Dung beetles | - | - | - | 38.1 | 20.1 | 22.6 | - | - | - |
| Vegetation/Seeds | 14.9 | 10.2 | - | 17.4 | 9.12 | - | 20 | 12.6 | - |
| **Total** | 145.4 | 100 | 100 | 190.1 | 100 | 100 | 158.2 | 100 | 100 |

### D) Wild dog

Food habits of wild dog based on the analysis of 41 scats indicated the occurrence of eight prey items in its diet. Both the frequency of occurrence and relative occurrence of rodents was highest in the scats (31.7%, 23.2%) followed by serow, Himalayan tahr and goral (Table 5.11). The proportion of biomass contribution to the total prey biomass consumed by wild dog was maximum for serow (38.6%) followed by dzo

(27.7%) and Himalayan tahr (23.37%). Wild dog selected serow, Himalayan tahr and goral more than their availability in the area. On the contrary, dzo was utilized less, while pika and rodent were utilized in accordance with their availability in the area (Table 5.10).

**Table 5.10 Prey selection by carnivores based on available and consumed biomass as shown by Jacobs's selectivity index and its significance value (*p*).**

| **Prey species** | **Snow leopard** | ***p*** | **Red fox** | ***p*** | **Marten** | ***p*** | **Wild dog** | ***p*** |
|---|---|---|---|---|---|---|---|---|
| Blue sheep | -0.0189 | 0.8859 | -0.5936 | <0.01 | - | - | - | - |
| Himalayan Tahr | 0.623 | <0.01 | -0.5254 | 0.178 | -0.631 | <0.05 | 0.4570 | <0.01 |
| Musk Deer | 0.1317 | 0.8094 | - | - | - | - | - | - |
| Serow | - | - | - | - | -0.46 | 0.076 | 0.5913 | <0.01 |
| Goral | - | - | - | - | -0.0765 | 0.712 | 0.2013 | 0.274 |
| Barking deer | - | - | - | - | -0.5558 | 0.138 | - | - |
| Dzo | -0.396 | <0.05 | -0.9591 | <0.01 | -0.863 | <0.05 | -0.6303 | <0.01 |
| Pika | 0.7415 | <0.01 | 0.9898 | <0.01 | 0.8742 | <0.01 | 0.0567 | 0.988 |
| Rodent | - | - | 0.5968 | 0.065 | 0.955 | <0.01 | 0.0951 | 0.885 |
| Bird | 0.0937 | 0.8063 | 0.4664 | <0.05 | 0.8897 | <0.01 | - | - |

**Table 5.11 Percent frequency of occurrence (FO), relative occurrence (RO) and relative biomass consumed (RB), of prey species in the scats of wild dog (n=41), leopard cat (n=37) and golden jackal (n=24).**

| **Prey species** | **Wild dog** | | | **Leopard cat** | | **Jackal** | |
|---|---|---|---|---|---|---|---|
| | **FO** | **RO** | **RB** | **FO** | **RO** | **FO** | **RO** |
| Himalayan tahr | 21.95 | 16.06 | 23.37 | - | - | - | - |
| Serow | 26.83 | 19.64 | 38.63 | 2.7 | 1.75 | 8.33 | 5.14 |
| Goral | 19.51 | 14.29 | 8.33 | 5.4 | 3.51 | 20.83 | 12.82 |
| Barking deer | - | - | - | - | - | 12.5 | 7.69 |
| Langur | - | - | - | - | - | 4.17 | 2.56 |
| Pika | 19.51 | 14.29 | 0.78 | 21.62 | 14.04 | 12.5 | 7.69 |
| Rodent | 31.71 | 23.21 | 1.19 | 89.19 | 57.89 | 54.17 | 33.34 |
| Dzo | 7.32 | 5.36 | 27.7 | 5.4 | 3.51 | - | - |
| UF-mammal | 2.44 | 1.79 | - | 10.81 | 7.02 | 12.5 | 7.69 |
| Bird | - | - | - | 10.81 | 7.02 | 20.83 | 12.82 |
| Vegetation/Seeds | 7.32 | 5.36 | - | 8.11 | 5.26 | 16.67 | 10.25 |
| **Total** | 136.6 | 100 | 100 | 154.04 | 100 | 162.5 | 100 |

### E) Leopard cat and Jackal

Based on the analysis of a limited number of leopard cat (37) and jackal (24) scats, only preliminary exploration of their diet was possible. Diet of leopard cat consisted of eight different prey items and small mammals (rodents and pikas) constituted the main prey (Table 5.11). The frequency of occurrence and relative occurrence of rodents (89.2%, 57.9%) was highest in the scats followed by pika and birds (galliformes). Nine different prey items were identified from the scats of golden jackal. Frequency of occurrence and relative occurrence of rodents was highest (54.17%, 33.34%) followed by goral and birds (galliformes) [Table 5.11].

### *5.3.2.2 Niche breadth and dietary overlap*

Levin's standardized index indicated that wild dog (0.722) and snow leopard (0.629) had the most diverse dietary niche breadths among the studied carnivore species followed by jackal and red fox (Table 5.12). Pianka niche overlap index indicated generally high overlaps among the diets of carnivores. Although, dietary overlaps were observed to be more that 50% in most of the cases based on the relative occurrence of prey items in the carnivore scats, maximum and significant overlap was observed for leopard cat (91%, $p = 0.0103$), jackal (90%; $p = 0.0213$), red fox (77%; $p = 0.0217$) and wild dog (69%; $p = 0.0221$) with yellow-throated marten, respectively (Table 5.13).

**Table 5.12 Levin's Standardized Niche Breadth Index of carnivores based on relative frequencies of occurrence (RO) and relative biomass (RB) of prey species in scats.**

| Species | Bsta (RO) | Bsta (RB) |
|---|---|---|
| Snow leopard | 0.629 | 0.4526 |
| Red fox | 0.4965 | 0.5125 |
| HYT Marten | 0.3985 | 0.4376 |
| Wild dog | 0.722 | 0.495 |
| Leopard cat | 0.243 | - |
| Jackal | 0.5871 | - |

**Table 5.13 Dietary overlap between carnivores based on relative occurrence (RO) and relative biomass (RB) of prey species in their scats depicted through Pianka index at significance value ($p = 0.05$).**

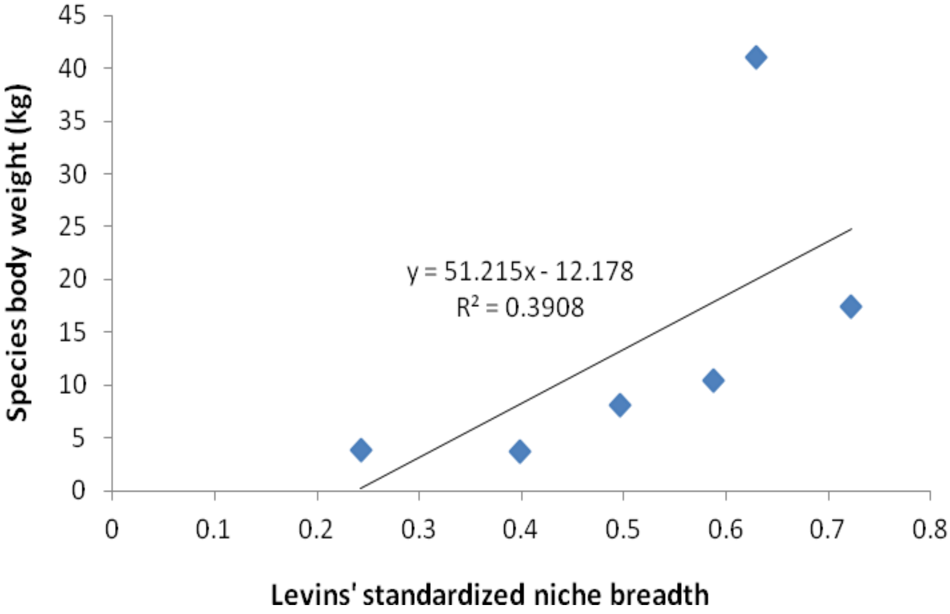
| | Relative occurrence (RO) | | Relative biomass (RB) | |
|---|---|---|---|---|
| **Species** | **Pianka index** | ***p*** | **Pianka index** | ***p*** |
| Snow leopard-Red fox | 0.5052 | 0.4371 | 0.2958 | 0.7252 |
| Snow leopard-Wild dog | 0.3439 | 0.8445 | 0.5295 | 0.2230 |
| Snow leopard-Marten | 0.3217 | 0.6899 | 0.2004 | 0.7339 |
| Red fox-Wild dog | 0.5596 | 0.3362 | 0.0693 | 0.8927 |
| Red fox-Marten | 0.7767 | 0.0217* | 0.7381 | 0.0981 |
| Wild dog-Marten | 0.6927 | 0.0221* | 0.1976 | 0.8253 |
| Leopard cat-Jackal | 0.8981 | 0.0618 | - | - |
| Leopard cat-Wild dog | 0.6828 | 0.1521 | - | - |
| Leopard cat-Marten | 0.9168 | 0.0103* | - | - |
| Wild dog-Jackal | 0.7260 | 0.0915 | - | - |
| Marten-Jackal | 0.9095 | 0.0213* | - | - |

* indicates *p* is significant at 0.05 level

### *5.3.2.3 Niche breadth and body size (weight)*

The Levins' standardized niche breadth and body weight of the carnivores showed an almost linear relationship ($R^2 = 0.391$, Figure 5.3). This indicated that larger species in the area tend to be more generalist and had a wider dietary spectrum compared to the small body size species which tend to be dietary specialists.

**Figure 5.3 Relationship between Levins' standardized niche breadth and body weight of the carnivore species.**



### 5.3.3 Activity patterns

#### *5.3.3.1 Temporal activity profile*

The activity profile of snow leopard was generally bimodal. Although, more active during later part of the day towards dusk and midnight (Figure 5.4), its activity pattern was overall uniformly distributed (Rayleigh Z = 1.622, $p < 0.199$) with no significant difference between the activities in different times of the day (Watsons $U^2 = 0.186$, $p > 0.05$). Also, the mean time activity of snow leopard was at 20:09 h ± 02:04 with 95% confidence interval (CI) of 16:00-00:09 h (Table 5.14). On the contrary, red fox exhibited a nocturnal activity pattern (Figure 5.4) with mean time activity at 22:22 h ± 00:24. The activity pattern of red fox was not uniformly distributed (Z = 39.51, $p < 0.0001$) and there was a significant different in the intensity of activity during different times of the day ($U^2 = 2.11$, $p < 0.005$; Table 5.15). Both the species of mustelids exhibited a diurnal activity pattern (Figure 5.4), with mean time activities at 11:58 h ± 00:15 (yellow-throated marten) and 11:59 h ± 01:16 (stoat), respectively. Activity profiles of both the species were not uniform with significant differences in the activities in different times of the day.

**Table 5.14 Circular statistics of temporal activity pattern of carnivores.**

| Species | N | Mean vector | S.E | 95% CI | Circular variance |
|---|---|---|---|---|---|
| Snow leopard | 24 | 20:06 | 02:04 | 16:00-00:09 | 0.729 |
| Red fox | 163 | 22:22 | 00:24 | 21:35-23:09 | 0.515 |
| Yellow-throated marten | 214 | 11:58 | 00:15 | 11:27-12:28 | 0.372 |
| Stoat | 34 | 11:59 | 01:16 | 09:28-14:28 | 0.65 |
| Black bear | 25 | 15:20 | 04:00 | 07:28-23:09 | 0.863 |
| Wild dog | 9 | 10:36 | 01:18 | 08:02-13:09 | 0.27 |
| Golden cat | 25 | 08:38 | 01:41 | 05:19-11:56 | 0.688 |
| Leopard cat | 62 | 00:51 | 00:30 | 23:48-01:47 | 0.359 |
| Masked palm civet | 40 | 22:26 | 00:34 | 21:20-23:36 | 0.358 |
| Large Indian civet | 30 | 00:25 | 00:39 | 23:08-01:42 | 0.346 |

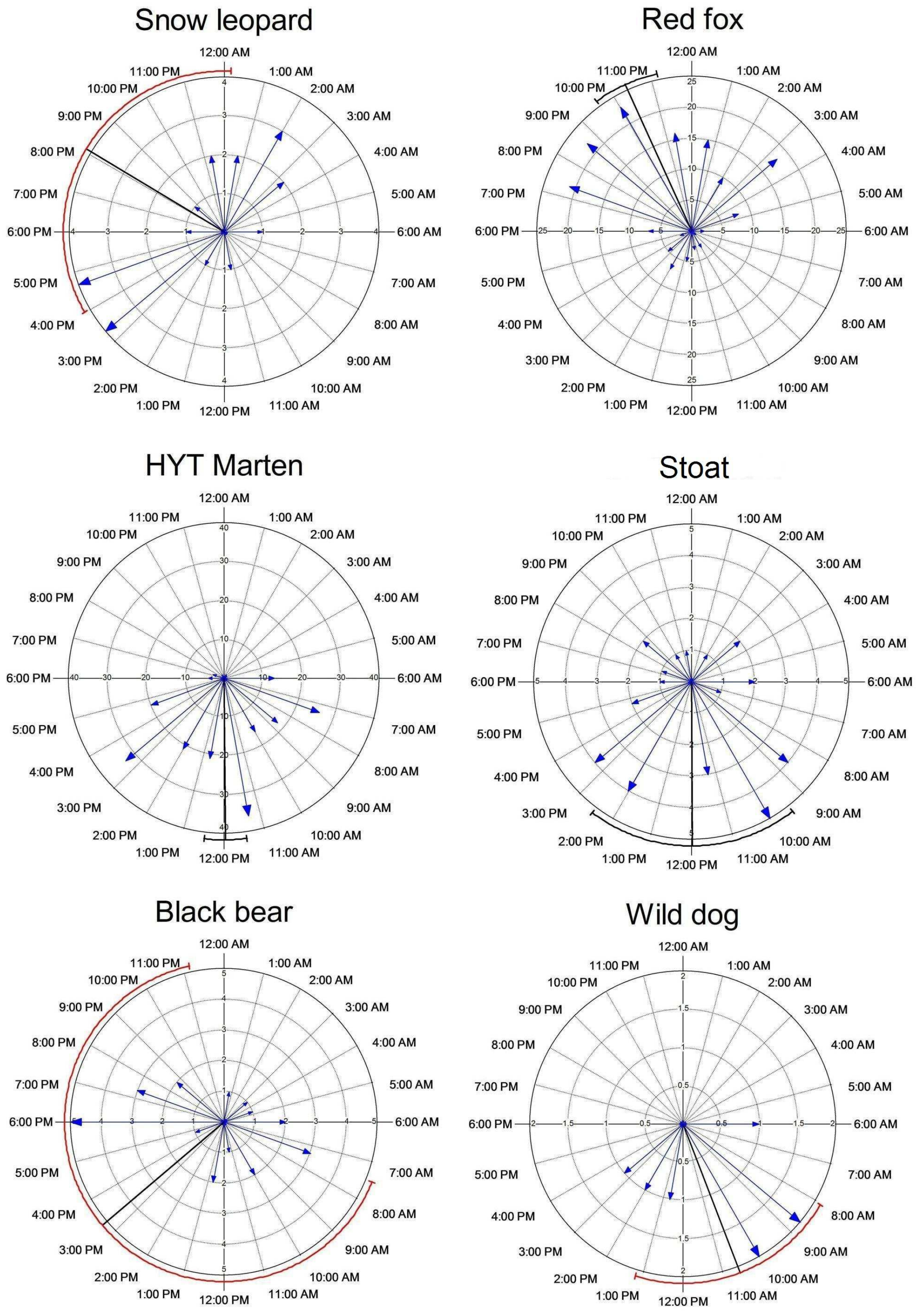
The activity pattern of black bear was arrhythmic while the wild dogs were found to be mainly diurnal in their activity (Figure 5.4). The mean activity of black bear and wild dog ranged from 07:28-23:09 h and 08:02-13:09 h, respectively (Table 5.14). Also, the activity pattern of black bear was uniformly distributed all through the day

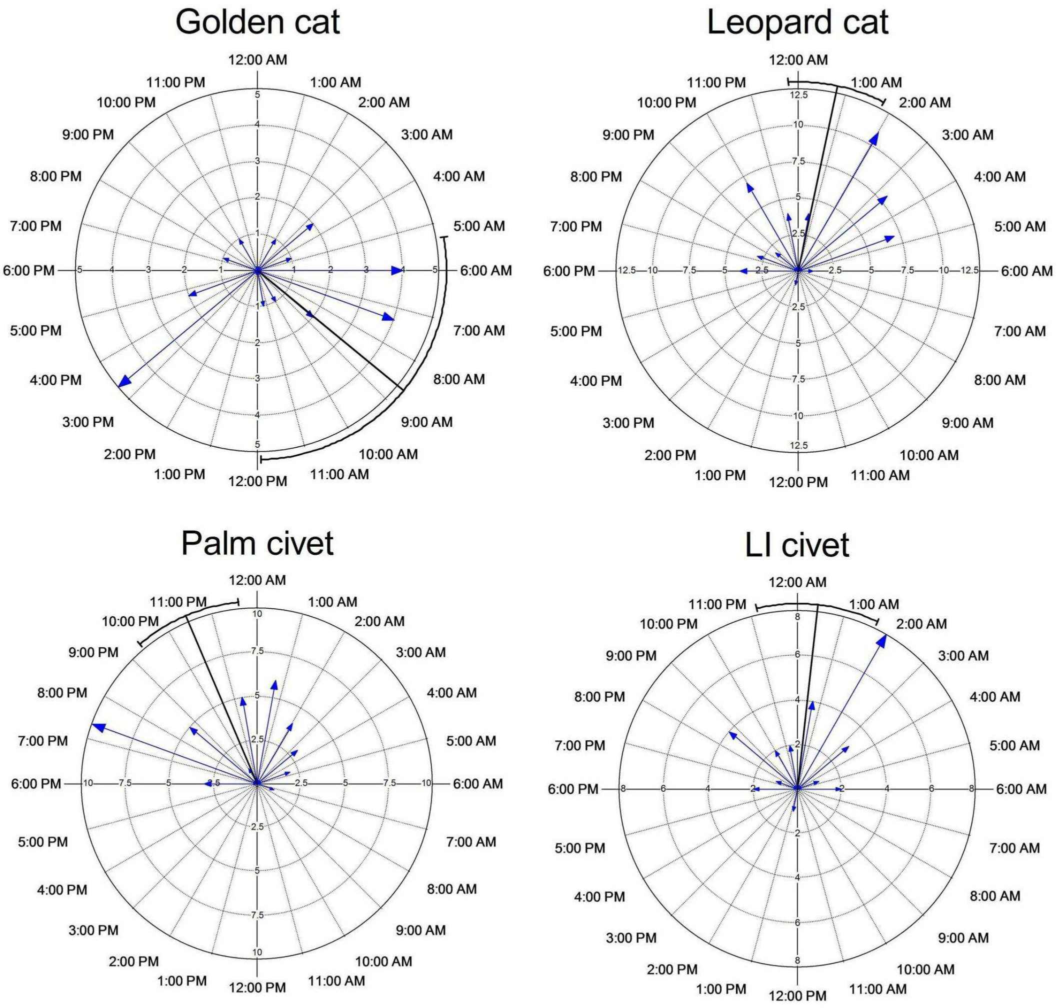
(Z = 0.451, p = 0.624) contrary to that of wild dog (Z = 4.266, p = 0.009). Golden cat exhibited an arrhythmic activity pattern and was more active during crepuscular hours. Although, the activity profile was almost uniformly distributed throughout the day (Z = 2.436, p = 0.087), there was a significant difference in the intensity of activity in different times of the day (Table 5.15) with its mean time activity positioned at 08:38 h ± 01:41. Leopard cat showed typical nocturnal activity pattern (Figure 5.4), with mean time activity at 00:51 h ± 00:30 and significant difference in the activity profile in different times of the day ($U^2 = 1.187$, $p < 0.005$). Both the species of civets exhibited a strict nocturnal activity pattern (Figure 5.4), with their mean time activities positioned at 22:26 h ± 00:34 (masked palm civet) and 00:25 h ± 00:39 (large Indian civet), respectively (Table 5.14). In addition, their activity profiles were not uniformly distributed and were significantly different across different times of the day (Table 5.15).

**Table 5.15 Rayleigh Test (Z) to determine the uniformity in the activity pattern and Watson's Test ($U^2$) to test the differences within the activity pattern of each carnivore species, *p* is the probability of significance**

| **Species** | **Rayleigh Test (Z)** | ***p*** | **Watson's Test ($U^2$)** | ***p*** |
|---|---|---|---|---|
| Snow leopard | 1.622 | 0.199 | 0.186 | >0.05 |
| Red fox | 39.51 | <0.0001 | 2.11 | <0.005 |
| Yellow-throated marten | 84.33 | <0.0001 | 4.361 | <0.005 |
| Stoat | 4.177 | 0.014 | 0.24 | <0.025 |
| Black bear | 0.451 | 0.624 | 0.126 | >0.1 |
| Wild dog | 4.266 | 0.009 | - | - |
| Golden cat | 2.436 | 0.087 | 0.215 | <0.05 |
| Leopard cat | 21.38 | <0.0001 | 1.187 | <0.005 |
| Masked palm civet | 16.5 | <0.0001 | 0.906 | <0.005 |
| Large Indian civet | 12.81 | <0.0001 | 0.712 | <0.005 |

**Figure 5.4 Temporal activity patterns of carnivores in the *Prek chu* catchment.**





### 5.3.3.2 Temporal overlap

The activity profiles of carnivores were different from each other in most of the cases as indicated by the Watson's $U^2$ test (Table 5.16). The most significant difference in the activity patterns were observed between yellow-throated marten and leopard cat ($U^2 = 3.095$, $p < 0.0001$), masked palm civet ($U^2 = 2.507$, $p < 0.0001$) and large Indian civet ($U^2 = 1.874$, $p < 0.0001$), respectively. Pianka overlap index provided weak evidence for overlaps in the activity patterns of species in general (Table 5.16). However, significant overlap was observed only between golden cat and yellow-throated marten (73.6%, $p = 0.030$), leopard cat and large Indian civet (86.4%, $p = 0.0416$), and yellow-throated marten and stoat (79.3%, $p = 0.048$), respectively.

**Table 5.16 Watson's Test ($U^2$) to test the differences between the activity patterns of pairs of carnivores and Pianka overlap index to check the intensity of temporal overlap in activity patterns between species, at appropriate level of significance (*p*).**

| **Species** | **Watson's Test ($U^2$)** | ***p*** | **Pianka index** | ***p*** |
|---|---|---|---|---|
| Golden cat-Leopard cat | 0.865 | <0.001 | 0.298 | 0.724 |
| Golden cat-Yellow-throated marten | 0.171 | >0.1 | 0.736 | 0.030* |
| Golden cat-Stoat | 0.177 | >0.05 | 0.668 | 0.085 |
| Golden cat-Masked palm civet | 0.839 | <0.001 | 0.207 | 0.901 |
| Golden cat-Large Indian civet | 0.699 | <0.001 | 0.269 | 0.563 |
| Golden cat-Black bear | 0.137 | >0.1 | 0.447 | 0.830 |
| Golden cat-Wild dog | 0.216 | <0.05 | 0.50 | 0.199 |
| Leopard cat- Yellow-throated marten | 3.095 | <0.0001 | 0.064 | 0.977 |
| Leopard cat-Masked palm civet | 0.207 | <0.05 | 0.662 | 0.153 |
| Leopard cat-Large Indian civet | 0.069 | >0.5 | 0.864 | 0.042* |
| Leopard cat-Wild dog | 0.55 | <0.001 | 0.031 | 0.946 |
| Leopard cat-Black bear | 0.728 | <0.001 | 0.376 | 0.819 |
| Snow leopard- Yellow-throated marten | 0.781 | <0.001 | 0.534 | 0.294 |
| Snow leopard-Red fox | 0.233 | <0.02 | 0.507 | 0.634 |
| Snow leopard-Wild dog | 0.365 | <0.002 | 0.227 | 0.819 |
| Yellow-throated marten-Stoat | 0.138 | >0.1 | 0.793 | 0.048* |
| Yellow-throated marten-Masked palm civet | 2.507 | <0.0001 | 0.071 | 0.886 |
| Yellow-throated marten-Large Indian civet | 1.874 | <0.0001 | 0.089 | 0.971 |
| Yellow-throated marten-Red fox | 4.921 | <0.001 | 0.219 | 0.921 |
| Yellow-throated marten-Wild dog | 0.063 | >0.5 | 0.629 | 0.212 |
| Yellow-throated marten-Black bear | 0.6 | <0.001 | 0.461 | 0.664 |
| Masked palm civet-Large Indian civet | 0.126 | >0.1 | 0.687 | 0.138 |
| Masked palm civet-Black bear | 0.55 | <0.001 | 0.574 | 0.141 |
| Large Indian civet-Black bear | 0.612 | <0.001 | 0.393 | 0.671 |
| Red fox-Wild dog | 0.482 | <0.001 | 0.153 | 0.875 |
| Wild dog-Black bear | 0.209 | <0.05 | 0.289 | 0.847 |

* indicates *p* is significant at 0.05 level

## 5.4 DISCUSSION

### 5.4.1 Factors governing space use

Landscapes provide the ecological template for the life history and behavioral processes that determine animal spacing patterns (Gough and Rushton 2000) and habitats. The species thereby express their perception of the landscape and its attributes by their habitat-use patterns (Wiens 1976). Habitat specialists therefore exhibit a behaviour that shows a high dependence on particular resources and strong sensitivity to habitat changes and loss, while habitat generalists tend to be very plastic and able to survive in many habitats (Santos and Santos-Reis 2010). However, landscape specific constraints of the environments also play substantial role in dictating the animals' habitat-use strategies in complex landscapes such as Himalayas.

In the present study, occupancy modelling suggested that the intensity of space use by snow leopard in the *Prek chu* catchment was strongly influenced by elevation and the presence of alpine habitat which is characterized by meadows, rocky areas as well as rhododendron scrubs. The strong avoidance of areas with high vegetation cover was obvious since the species is adapted to cold and rocky high altitude Himalayan terrain (Schaller 1977). Corresponding to the present results of occupancy modelling the snow leopard is known to be associated with steep broken mountainous habitat in the alpine and sub-alpine zones where the vegetation is sparse (McCarthy and Chapron 2003). It is generally found in elevations from 3,000-4,500 m inhabiting the timberline-alpine ecotone and occasionally uses areas above 5,500 m but prefers steep terrain broken by cliffs ridges and gullies (Ahlborn and Jackson 1988; Chundawat 1992; McCarthy and Chapron 2003). In the present case its occurrence strictly above 4,000 m (alpine and rocky areas) may be due to prolonged spread of tree line to higher reaches which is a characteristic of eastern Himalaya. Red foxes on the other hand are known to inhabit a wide variety of habitats such as sub-alpine scrub forests, pastures, rocky areas, moorlands, mountains (even above the tree line as they are known to cross alpine passes > 4,500 m), cultivated lands and the vicinity of water courses (Sillero-Zubiri et al. 2004; Macdonald and Reynolds 2008). Although they can adapt to survive in many habitats, their ideal natural habitats are dry, mixed landscape with abundant "edges" of scrub and woodland (Ables 1975; Macdonald and Reynolds 2008). The results of the occupancy modelling in the present study indicating positive influence of *krummholtz* habitats (alpine-subalpine edge) and trekking trails on their space use was hence in accordance with the known information on the species habitat preference, particularly in reference to Himalayas (Aryal et al. 2010). Moreover, since the availability of prey and suitable den sites are also known to affect habitat selection by red foxes as they prefer to locate their dens in forests close to open areas (Voigt and Broadfoot 1983), affinity towards trekking trails can not only enhance the edge effect and connectivity between such habitats but also attract their preferred prey species and hence enhance their survival.

Among mustellids, the occupancy modelling showed highest occupancy rate for yellow-throated marten thereby depicting it to be the most widely distributed carnivore in the area. For comparison, the site occupancy was found higher than that reported in the montane tropical forests in northern Laos (0.29 ± 0.08; Johnson et al. 2009). Its detection probability revealed to be positively influenced by human presence indicated that the species is accustomed to human interference and hence was the most frequently detected carnivore in the area (in terms of direct sightings) and elsewhere (Datta et al. 2008b; Johnson et al. 2009; Mathai et al. 2010). But, its lower detection probability compared to red fox and large Indian civet could be attributed to its occasional arboreal behaviour (Duckworth 1997; Sunquist and Sunquist 2002). The site occupancy of yellow-throated marten being positively influenced by vegetation cover spreading over a wide elevation gradient was also in accordance with the findings elsewhere where it is known to favour dense forests and prefers mixed (spruce and broad-leaved) forests in Russia (Matyushkin 1993) and various other adjacent habitats across a wide altitudinal range in Lao PDR, Myanmar, Thailand and India (Duckworth et al.1999; Lekagul and McNeely 1977; Than Zaw et al. 2008; Choudhury 1997b). The avoidance of trekking trails is hence apparent as these represent edges and open areas. Stoats, on the contrary are known to occupy a wide range of habitats and are often found in successional or forest-edge habitats, in scrub, alpine meadows, marshes, riparian woodlands, hedgerows, and riverbanks that have high densities of small mammals (King 1983; Murphy and Dowding 1995; Martinoli et al. 2001; Smith et al. 2007), while Pulliainen (1999) and Samson and Raymond (1998) mentioned that coniferous and mixed woodlands are preferred. These observations are in accordance with the current results of occupancy modelling which indicated positive and significant influence of elevation, tree cover and presence of coniferous forests on the space use of stoats.

Other canid species including wild dogs and jackals seemed to be casual visitors in the area. Single photo-capture of jackal and distribution of its signs only in the sub-tropical habitats indicated its limited occurrence in the study area mainly confined below 1,500 m altitude in the buffer zone. This pattern of space use was in accordance with its tendency for warmer areas adjacent to human habitations as recorded elsewhere (Poche´ et al. 1987; Jhala and Moehlman 2008). Similarly, no sighting, less number of photo-captures and few sign/scat encounters showed that wild dogs were not only nomadic but also rare in the study area (Bashir et al. 2013a). Results from camera trapping and trail sampling hence indicated that dholes mainly visit sub-alpine and subalpine-alpine edge forests of the area. Dholes being considered as wanderers are reported to inhabit a wide range of habitats except desert regions (Johnsingh 1985; Durbin et al. 2004; Iyengar et al. 2005). Their sparse distribution pattern in the present case from upper temperate to alpine habitats was hence in accordance with their distribution in the rest of the Himalayas, north-west and north-east India where they have been reported to be more fragmented and recorded even from high altitudes of Ladakh at 4,500 m (Bedi and Bedi 1991; Durbin et al. 2008).

Black bears on the contrary due to their omnivorous habit and dependence on plant food, their intensity of habitat use is known to fluctuate with season and availability of resources in the environment (Garshelis and Steinmetz 2008). Their probability of space use being influenced by the presence of broadleaved and coniferous forests as predicted by the occupancy modelling is justified as these forests primarily provide food and denning sites to the bears. Preference for broadleaved and coniferous forests has also been specified in other studies (Izumiyama and Shiraishi 2004; Sathyakumar and Choudhury 2007; Wang et al. 2008; Garshelis and Steinmetz 2008). Moreover, movement of bears to broadleaved forests at lower elevations in the montane zones rich in hard-mast producing trees in autumn and fall (Huygens et al. 2003) has well been documented indicating their preference for oak acorns, beechnuts, walnuts, chestnuts, hazelnuts, or stone pine seeds (Hwang 2003; Hashimoto et al. 2003; Sathyakumar and Viswanath 2003). Although in summer they are also known to seldom move to alpine areas where they rely on succulent plants (Huygens et al. 2003; Garshelis and Steinmetz 2008) which were also observed in the present study. While comparing the results of the occupancy modelling, both detection probability and the probability of site utilization was less than ($\Psi = 0.68 \pm 0.042$ and $p = 0.82 \pm$

0.06) observed in Min Mountains of Sichuan Province, China (Trent 2010), indicating that bears were comparatively less common in the study area.

Among other felids as predicted by the occupancy modelling the probability of site utilization by Asiatic golden cat was influenced mainly by vegetation cover. Although, some studies have suggested that golden cats are less common in montane forests, Wang (2007) has reported its occurrence from areas of dwarf rhododendron and grassland at high elevations in Bhutan which was even observed in the present study (*krummholtz* habitats) at an altitudinal record of 3,960 m (Bashir et al. 2011), indicating its adaptability to a broad range of habitats. The species is however know to mainly inhabit tropical and sub-tropical moist evergreen forest, mixed evergreen hill forest (Grassman et al. 2005a) and dry deciduous forests (Nowell and Jackson 1996). While, comparing the site occupancy with that observed in montane tropical forest in northern Laos ($\Psi = 0.42 \pm 0.14$ and $p = 0.050 \pm 0.01$; Johnson et al. 2009), it was lower in the present study but detection probability was higher, which indicates that golden cats used few areas but intensively. In case of the leopard cat site occupancy being positively influenced by the presence of broadleaved forest and abundance of small mammals and galliformes, strong negative influence of elevation suggested that leopard cats were more functional in lower reaches much below the tree line. The species is otherwise capable of utilizing a broad variety of habitat types from tropical rain forest to temperate broadleaf and marginally coniferous forest, as well as shrub forest and successional grasslands modified agricultural landscapes, logged forests and rubber and oil palm plantations (Nowell and Jackson 1996; Rajaratnam et al. 2007). In the Himalayas, it has been reported to extend into rhododendron-oak-maple forests up to an altitude of 3,254 m in Makalu-Barun NP, Nepal (Ghimirey and Ghimire 2010), however its restricted occurrence only up to 2,750 m in the present study indicated its preference for lower elevations (temperate and sub-tropical habitats) in the wet environments of eastern Himalaya. Insignificant influence of human presence on its detection also indicated its adaptation to human interference which is also in accordance with the findings of Azlan et al. (2009) reporting high photo-capture rates in open areas along roads; and also high detection probabilities on-roads and less off-roads (Mohamed et al. 2013) compared to the present estimates. This argument can be supplemented by the fact that broadleaved forests of temperate and sub-tropical zone occur in proximity to human habitations in the buffer area.

Both the viverids primarily inhabited the lower elevation areas. Use of primarily broadleaved forests (semi-evergreen and evergreen) at lower elevations of the study area (up to 2,700 m) by large Indian civets as predicted by the occupancy modelling were in accordance with the reports elsewhere which demonstrate their occurrence in primary forest (both evergreen and deciduous), secondary forest, plantations and adjacent degraded areas (Duckworth 1997; Duckworth et al. 1999; Azlan 2003). The negative influence of human presence on their detection also indicated avoidance of anthropogenic interference. Moreover, lower site occupancy but higher detection probability compared to that reported in montane tropical forest in northern Laos ($\Psi$ = 0.36 ± 0.15 and p = 0.078 ± 0.04; Johnson et al. 2009) also strengthened the argument that large Indian civets had a more specified habitat use in the study area restricted only to lower semi-evergreen and evergreen forests and confined to dense forest cover away from human habitation. On the contrary, masked palm civets believed to have a differential habitat usage across their range, and known to occupy evergreen, deciduous forest and disturbed habitat (Rabinowitz 1991; Heydon and Bullon 1996; Grassman 1998; Duckworth 1997; Azlan 2003) and even sub-tropical coniferous and broad-leaved forests (Zhou et al. 2008), and reported to have a wider habitat range compared to that of large Indian civets (Johnson et al. 2009); the results of the occupancy modelling in the present study indicated a similar trend in their habitat use. This was evident from non-significant influence of any habitat type on their space use and negative relationship with increasing elevation; thereby indicating usage of even low elevation forests of the area. Moreover, the estimates of site occupancy were almost similar while detection probability higher than that observed in montane tropical forest in northern Laos ($\Psi$ = 0.14 ± 0.04 and p = 0.237 ± 0.05; Johnson et al. 2009). However, while comparing among the two civets in the present study, lower detection probability of masked palm civet may probably be attributed to its partial arboreal behavior (Lekagul and McNeely 1977; Duckworth 1997).

### 5.4.2 Dietary habits

Food habits of carnivores are central to the ecological niche they occupy and play an important role in explaining their social systems, behaviour and factors affecting predator density. For carnivores, the availability and utilization of various food resources are important factors affecting population viability (Fuller and Sievert 2001). Additionally, competitive interactions among carnivore species are common and can have major impacts upon their ecology and management (Creel et al. 2001). Therefore, knowledge of their food selection is critical in understanding the life history strategies and developing sound conservation recommendations (Miquelle et al. 1996), particularly for species inhabiting the least explored fragile high-altitude ecosystems such as the Himalayas.

Studies on snow leopard food habits across their range have shown that their diet primarily depends upon large or medium-sized ungulates, such as blue sheep, Asiatic ibex *Capra ibex*, markhor *Capra falconeri* and Himalayan tahr as well as sciurid rodents such as the Himalayan marmot *Marmota himalayana* and lagomorphs e.g., pika *Ochotona* spp. The occurrence of blue sheep as the most frequent prey species and dzo as the major contributor of prey biomass to the diet of snow leopard in the present study was therefore in accordance with the studies conducted in north-western India (Ladakh: Chundawat and Rawat 1994; Kibber WS: Bagchi and Mishra 2006), Nepal (Annapurna: Oli 1994, Oli et al. 1994; Langu: Jackson 1996; Phu valley: Wegge et al. 2012), Qinghai region of China (Schaller et al. 1988) and parts of Tibet (Jackson et al. 1994) indicating blue sheep as the principal prey. However, the high contribution of livestock to the total biomass consumed by snow leopard inspite of wild species occurring in major proportion of the scats may be attributed to the presumed low diversity (and density) of wild ungulate prey (except blue sheep) in the area evident from very low photo-capture rates of Himalayan tahr (0.09 ± 0.07) and musk deer (0.11 ± 0.07) compared to that of blue sheep (3.81 ± 2.54; Bhattacharya 2013). Such supplementary prey (e.g., livestock; Wang and Macdonald 2006) are known to act as buffers and potentially enhance coexistence among sympatric carnivores (Kok and Nel 2004). However, the indication of Himalayan tahr and pika as the most preferred prey species by the snow leopard based on prey selectivity index of available and consumed biomass was in accordance with the findings of Shrestha (2008) and Lovari et al. (2009) in Sagarmatha National Park, both indicating Himalayan thar to be the main prey of snow leopard in absence of blue sheep and ibex. Since, the present study area inhabited both blue sheep as well as represented the eastern most distribution of Himalayan tahr (Sathyakumar 2002; Bhatnagar and

Lovari 2008); the results of this study can bring in important insights towards understanding the food habits of snow leopard in such unique scenario.

Red foxes on the contrary are generally characterized as generalist opportunistic omnivores and scavengers that eat a wide variety of foods depending on seasonal availability (Sillero-Zubiri et al. 2004; Aryal et al. 2010; Papakosta et al. 2010). In the present study, composition of nine different prey items in its diet ranging from medium to large-sized ungulates, small mammals and arthropods to plant matters validated its omnivorous habit, but revealed pika, rodents, dung beetles and birds (galliformes) as the main prey. Such predominance of rodents and lagomorphs, along with a wide variety of other vertebrate, invertebrate and plant foods has been reported in a number of studies elsewhere. But these studies have also indicated a considerable contribution of wild ungulates and very less contribution by insects to the consumed biomass by red fox (Cagnacci et al. 2003; Lanszki et al. 2007; Papakosta et al. 2010) which is in contradiction to the present findings. Similar preference for pika as main prey has been reported from Dhorpatan hunting reserve, Nepal (Aryal et al. 2010). Moreover, considerable representation of dung beetles (insects) in the red fox diet may be attributed to seasonal fluctuations in the availability of vertebrate prey (for example during winters small mammals undergo hibernation).

Yellow-throated martens are generally known to be dietary generalists (Ramakantha 1994), diverse omnivores (Parr and Duckworth 2007) or even opportunistically frugivorous (Zhou et al. 2011) in their food habits. Similar opportunistic omnivorous diet composition was observed in the present study evident from maximum number of prey items recorded in their scats compared to other carnivores. Although, their diet composition ranged from large-sized ungulates to vegetable matter, the main prey items included rodents, pikas, birds and vegetable matter (seeds) which was in accordance with the findings elsewhere reporting rodents, small mammals, birds, reptiles, fruits as well as honey and bees as common food items (Lekagul and McNeely 1977; Ramakantha 1994; Parr and Duckworth 2007); though there have been reports of chases on Himalayan tahr, Musk deer, Goral and pheasants by yellow-throated martens (Sathyakumar 1999).

Information on the food habits of dholes revealed through this study although showed rodents as most frequently occurring prey in scats, relative biomass consumption indicated that more than 98% of the prey biomass consumed consisted of large to medium size ungulates (Bashir et al. 2013a) which was also verified through prey selectivity indices and broad niche breadth. Almost same dietary preferences for dholes have been established in other studies (Venkataraman et al. 1995; Grassman et al. 2005d; Acharya 2007; Kawanishi and Sunquist 2008; Borah et al. 2009; Kamler et al. 2012). However, the higher occurrence of rodents and pikas in the scats could be due to their ability to flush out and hunt the smaller and cryptic prey species in the bushes (Venkataraman 1996; Kumaraguru et al. 2011). On the other hand, jackals being omnivorous and opportunistic foragers; their diet varies according to season and habitat and is composed of both plant and animal components (Poche′ et al. 1987; Jhala and Moehlman 2008). Food habits revealed in the present study also validated the omnivorous habit of jackals and indicated rodents to be their main prey followed by birds, medium size ungulates and vegetative matter. Generally, mammals are reported to be the major constituents of their diet (Radovic and Kovacic 2010) and particularly small mammals (rodents) as main prey items (Jaeger et al. 2007; Majumder et al. 2011), high contributions of vegetable matter have also been observed in few studies (e.g., Nadeem et al. 2012).

Among other felids, dietary composition of the leopard cat revealed similar results when compared to the food habit studies conducted elsewhere, with rodents (murids) contributing a dominant proportion of the prey (Bashir et al. 2013c). Since murids dominate the terrestrial small-mammal communities in tropical rain forests (Wells et al. 2004), leopard cats have evolved to efficiently hunt small mammals particularly murids as predominant food resources (Kitchener 1991) for rapid energy maximization (Mukherjee et al. 2004). Moreover, the absence of herpetofauna and insects in the diet of leopard cat also indicated high rodent abundance in the area compared to other prey species because of which leopard cats did not require to supplementing their main prey.

### 5.4.3 Activity patterns

Although activity patterns are part of the basic natural history information such data is useful in studying processes that shape ecological communities, especially in knowing whether potentially competing species overlap or avoid each other temporally and understanding how larger species influence activities of their smaller cohorts in the same habitat. The snow leopard in the present study although based on low sample size was mostly active at twilight and during night. The species has otherwise been generally believed to be strongly nocturnal (Roberts 1977; Mallon 1984), but recent studies have also depicted its crepuscular activity pattern (Schaller et al. 1994; Jackson 1996). In coherence with the results of the present study, red foxes have been described as predominantly nocturnal animals (Ables 1975; Blanco 1986), but seasonal variations in activity patterns have also been observed (Ables 1975; Cavallini and Lovari 1991) and hence daytime activities are not uncommon. Moreover, these tend to be diurnal when eating diurnal insects but generally remain nocturnal feeding on nocturnal small mammals and fruits which are available for 24 hours (Cavallini and Lovari 1991).

Among mustellids, the diurnal activity pattern of yellow-throated marten observed in the present study has also been reported by studies conducted elsewhere (Duckworth 1997; Grassman et al. 2005c; Datta et al. 2008b; Than Zaw et al. 2008; Johnson et al. 2009), with few reporting arrhythmic tendencies, i.e., some nocturnal activity during moonlight nights (Duckworth 1997; Grassman et al. 2005c). Besides, crepuscular activity pattern of yellow-throated marten has also been reported in a camera trapping study conducted in Indonesia (van Schaik and Griffiths 1996). In case of stoats, their activity patterns are known to vary with season and habitats they inhabit as Debrot et al. (1985) has reported diurnal activity during summer and nocturnal during winter. However, equal activity during day and night in autumn and more diurnal in spring has also been reported (Alterio and Moller 1997). Primarily diurnal activity in the present study may be considered as a strategy to maximize on prey and avoid competition with other sympatric carnivores.

The diurnal activity pattern of dholes depicted in the present study was in accordance with that observed in primary rainforest of Peninsular Malaysia (Kawanishi and Sunquist 2008). Although, reported to be primarily active during diurnal and crepuscular periods, dholes can hunt at any time of the day or night depending on the spatio-temporal availability of prey (Johnsingh 1982; Grassman et al. 2005d). Similar, temporal relationship was also evident in the present study (as depicted through the activity patterns of major wild ungulates (Bhattacharya et al. 2012; Bhattacharya 2013). Correspondingly, black bears are largely known to be diurnal in their activity patterns as monitored in other areas (e.g., China: Reid et al. 1991; Pakistan: Roberts 1977; and Taiwan: Hwang 2003). The arrhythmic activity pattern with a tendency towards crepuscular activity rhythm in the present study was however in similarly with the findings of Garshelis and Pelton (1980) in the Great Smoky Mountains National Park and Sharma (2012) in Dachigam National Park.

Information on the activity pattern of the leopard cat synchronized with the findings of Cheyne and McDonald (2011) and Rajaratnam (2000) reporting 65% and 85% nocturnal activity; but contradict with those of Austin et al. (2007), Azlan and Sharma (2006), Grassman et al. (2005b) and Rabinowitz (1990) which reported diurnal, crepuscular and arrhythmic activity pattern. Moreover, it has been established that the daily activity of many felids is correlated with the activity pattern of their main prey (Zielinski 1988). This has been observed in leopard cats (Rajaratnam 2000) and Iriomote cats *Prionailurus iriomotensis* (Schmidt et al. 2009) preying upon their main prey (nocturnal murids), and was observed in the present study as well (Bashir et al. 2013c). Asiatic golden cats on the other hand have been known to be nocturnal in activity (Kawanishi and Sunquist 2008). But, as per the present record (arrhythmic and crepuscular activity) and also based on other studies (indicating daytime and crepuscular activity peaks), the species may not be considered exclusively nocturnal as previously thought (Grassman et al. 2005a; Azlan and Sharma 2006).

Contrary to the mustellids, both the species of viverrids exhibited nocturnal activity patterns. The strictly nocturnal activity of large Indian civet observed in the present study was in accordance with that reported in majority of the studies (Lekagul and McNeely 1977; Rabinowitz 1991; Duckworth 1997; Than Zaw et al. 2008) though occasional day-time records of activity have also been observed (Than Zaw et al. 2008). Similarly, primary nocturnal and occasional diurnal activity pattern of masked palm civet in the present study was also found to be in accordance with the earlier reports elsewhere (Lekagul and McNeely 1977; Rabinowitz 1991; Duckworth 1997; Grassman 1997, 1998; Than Zaw et al. 2008; Johnson et al. 2009; Mathai et al. 2010).

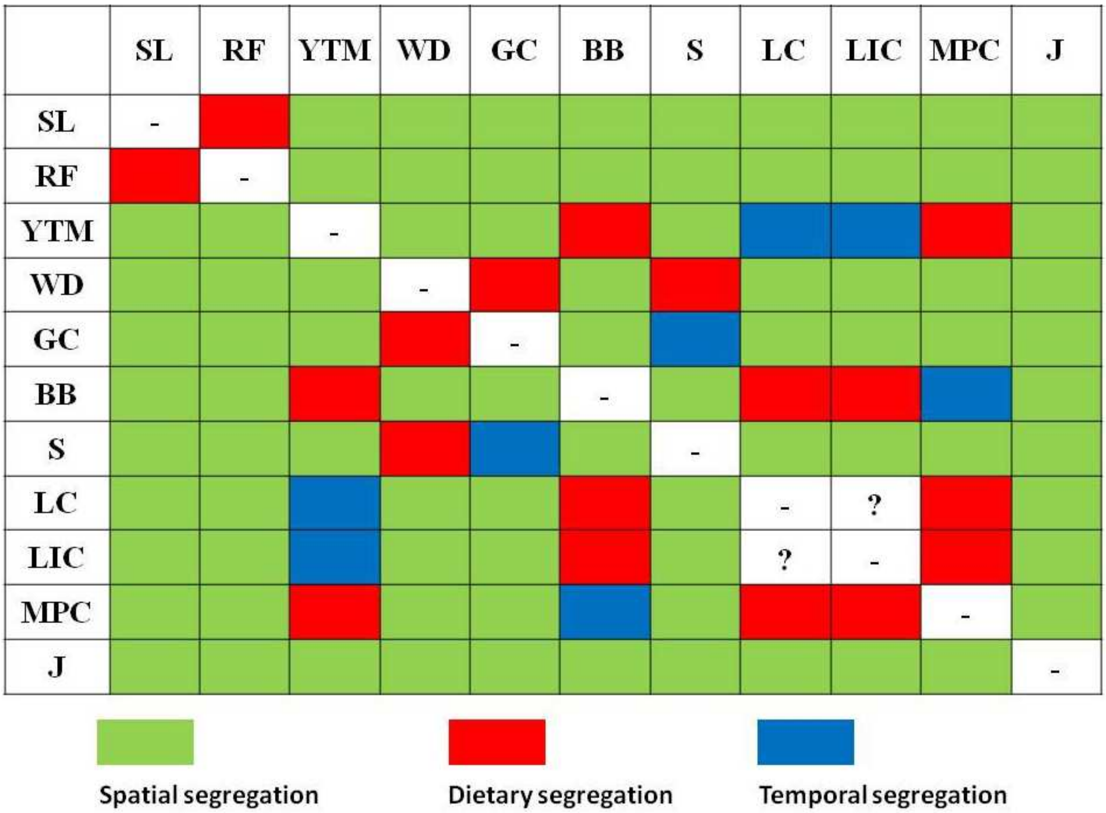
### 5.4.4 Resource partioning and co-existence

As niche theory (Diamond 1978) states that when two or more species co-exist, their habitat use and diet and therefore impact on community structure will not overlap completely. Differential resource selection is hence viewed as a fundamental process enabling co-evolved species to coexist (Rosenzweig 1981). The overall results indicated that the carnivore assemblage inhabiting this high-altitude ecosystem of Khangchendzonga also tends to partition the available resources at three distinct levels *viz.*, space, diet and time as mentioned by Schoener (1974). The area being structured into wide altitudinal gradients and a broad range of habitats and hence being influenced by varied weather conditions; the evolutionary adaptation of carnivores to such conditions favours a differential pattern of space use in such diverse landscapes (Gittleman 1996; Sunquist and Sunquist 2002) and thus allows primary segregation at the spatial scale. Although, the carnivores used a wide range of habitats, occupancy modelling indicated specific habitat preference by species (partitioning the available space) hence permitting their co-existence, but also showed some overlap in use between the habitats. This has resulted in space sharing and co-existence of certain species within a particular habitat type (Figure 5.5), e.g., snow leopard and red fox shared space and exhibited spatial co-existence inhabiting alpine-*krummholtz* habitats. Similarly, golden cat, stoat and wild dog showed co-existence mainly inhabiting sub-alpine areas. Moreover, a larger group consisting of large Indian civet, masked palm civet, leopard cat, yellow-throated marten and black bear exhibited spatial co-existence and were mainly concentrated in the temperate habitat, while the jackal inhabited the temperate-sub-tropical edge.

In order to facilitate co-existence and minimize competition these groups of carnivores tend to partition the available resources within the limited space at a secondary dietary scale through differential foraging habits and prey selection. Since, resource partioning could be enhanced by differences in body size in involving species (i.e., larger animals generally consume larger and more varieties of prey, and have broader trophic niche breadths- Schoener 1974), this was evident based on the relationship between body size and niche breadth of species (Figure 5.3). Co-existence between felids and canids has also been known to be facilitated by evolution of different anatomical adaptations for prey selection (Biknevicius and Van Valkenburgh 1996). Such segregation was quite evident between snow leopard and red fox (Figure 5.5). Although, significant differences in their body size could play an important role in minimizing competition, broader niche breadth of snow leopard also supplemented in facilitating successful co-existence between the two species. This was achieved through significant consumption and selection of large-medium sized prey species by snow leopard and small sized prey species by red fox, also evident from insignificant dietary overlap between the two species. Similarly, golden cat and wild dog being sympatric species (Kawanishi and Sunquist 2008) might also have facilitated co-existence at the same dietary scale (Figure 5.5). Since, the diet of golden cat is known to compose mainly of small mammals (rodents and other murids), snakes, lizards, birds and seldom of medium-sized prey (Lekagul and McNeely 1977; Nowell and Jackson 1996; Grassman et al. 2005a; Kawanishi and Sunquist 2008), the wild dogs on the contrary partitioned the available food resources by significant consumption and selection of large-medium sized prey species (serow, Himalayan tahr and goral). However, the diet of stoats known to be more or less similar to that of golden cat composing mainly of small rodents, lagomorphs, birds, fruits and invertebrates (Murphy and Dowding 1995; Martinoli et al. 2001), the two species seem to compete for the available food resources or segregate at a different scale.

On the other hand, carnivores co-existing in the temperate habitat also tend to partition the available resources at a dietary scale (Figure 5.5). In doing so the black bear being comparatively much larger in size and due to its omnivorous habit partitions the resource by focussing mainly on plant food comprising of *Castanopsis hystrix*, Acorn *(Quercus- glauca, lamellosa)*, *Symplocos* sp and *Rubus ellipticus*. On the contrary, yellow-throated marten and leopard cat did not substantially partition their dietary niches and both mainly consumed rodents, pikas and birds hence exhibiting significant dietary overlap. Moreover, large Indian civets being also known to be mostly carnivorous primarily foraging on small mammals, birds, lizards and invertebrates (Lekagul and McNeely 1977), there seems to be a possible competitive interaction among the three species. Contrary to this, masked palm civets known to eat primarily fruits and seldom small mammals, birds and invertebrates (Lekagul and McNeely 1977; Grassman 1997; Zhou et al. 2008) could minimize competition with other three sympatric carnivores and facilitate co-existence through its frugivorous habit. But, since both black bear and masked palm civet focus primarily on plant food, these two species also seem to compete for available food resources or segregate at a different scale.

**Figure 5.5 Chart depicting the level of segregation among the carnivore species at different scales.**



| | SL | RF | YTM | WD | GC | BB | S | LC | LIC | MPC | J |
|---|---|---|---|---|---|---|---|---|---|---|---|
| **SL** | - | 🟥 | 🟩 | 🟩 | 🟩 | 🟩 | 🟩 | 🟩 | 🟩 | 🟩 | 🟩 |
| **RF** | 🟥 | - | 🟩 | 🟩 | 🟩 | 🟩 | 🟩 | 🟩 | 🟩 | 🟩 | 🟩 |
| **YTM** | 🟩 | 🟩 | - | 🟩 | 🟩 | 🟥 | 🟩 | 🟦 | 🟦 | 🟥 | 🟩 |
| **WD** | 🟩 | 🟩 | 🟩 | - | 🟥 | 🟩 | 🟥 | 🟩 | 🟩 | 🟩 | 🟩 |
| **GC** | 🟩 | 🟩 | 🟩 | 🟥 | - | 🟩 | 🟦 | 🟩 | 🟩 | 🟩 | 🟩 |
| **BB** | 🟩 | 🟩 | 🟥 | 🟩 | 🟩 | - | 🟩 | 🟥 | 🟥 | 🟦 | 🟩 |
| **S** | 🟩 | 🟩 | 🟩 | 🟥 | 🟦 | 🟩 | - | 🟩 | 🟩 | 🟩 | 🟩 |
| **LC** | 🟩 | 🟩 | 🟦 | 🟩 | 🟩 | 🟥 | 🟩 | - | ? | 🟥 | 🟩 |
| **LIC** | 🟩 | 🟩 | 🟦 | 🟩 | 🟩 | 🟥 | 🟩 | ? | - | 🟥 | 🟩 |
| **MPC** | 🟩 | 🟩 | 🟥 | 🟩 | 🟩 | 🟦 | 🟩 | 🟥 | 🟥 | - | 🟩 |
| **J** | 🟩 | 🟩 | 🟩 | 🟩 | 🟩 | 🟩 | 🟩 | 🟩 | 🟩 | 🟩 | - |

🟩 Spatial segregation

🟥 Dietary segregation

🟦 Temporal segregation

*** SL = Snow leopard, RF = Red fox, YTM = Yellow-throated marten, WD = Wild dog, GC = Golden cat, BB = Black bear, S = Stoat, LC = Leopard cat, LIC = Large Indian civet, MPC = Masked palm civet, J = Jackal**

Correspondingly, activity time has been considered one of the most important axes of niche space (after habitat and food) along which organisms most frequently segregate (Schoener 1974). Partitioning at a tertiary temporal scale among species with similar overlapping dietary habits is necessary to minimize competition and facilitate co-existence. In this regard, predators are also known to synchronize predatory activities with activity periods of their primary prey and co-predators (Kronfeld-Schor and Dayan 2003; Harmsen et al. 2011). In the present case, stoat and golden cat minimized dietary competition through temporal partitioning of resources by exhibiting diurnal (stoat) and arrhythmic-crepuscular (golden cat) activity patterns (Figure 5.5). Black bear (arrhythmic-mainly diurnal) and masked palm civet (nocturnal) also segregated at the temporal scale (Figure 5.5). And yellow-throated marten, leopard cat and large Indian civet also minimized the possible competitive interactions in the similar way by exhibiting strict diurnal (yellow-throated marten) and nocturnal (leopard cat and large Indian civet) activity patterns (Figure 5.5). However, the study limits to clearly explain the mechanism by which leopard cats and large Indian civets partitioned the use of available resources (Figure 5.5), as both shared same space, exhibited similar activity patterns, and are also known to share similar dietary habits elsewhere. The degree of overlap in diet although represents a major interspecific relation that strongly affects the potential for competition, higher food overlap however does not necessarily indicate competition (Lanszki et al. 2007). Since, present study could provide dietary information only on few species detailed study on the food habits of the remaining carnivore species recorded can provide suitable answers to such queries and also help in better understanding of mechanisms by which carnivore communities structure themselves in such intricate ecosystems.

However, the information generated during this study did not only provide useful insight of carnivore ecology, but it can also aids in preparing spatial distribution models which can be used to mitigate the local as well as landscape level conservation issues to chalk out a proper management plan for the protection of suitable habitats. The next chapter (Chapter 6) attempts to check whether these findings on habitat use patterns by carnivores in the *Prek chu* catchment are adequate and wide-ranging enough for predicting their habitat suitability in the extent of the entire KBR.

## Chapter 6

# HABITAT SUITABILITY MODELLING

## 6.1 INTRODUCTION

Efficient conservation and management of a species/group and its/their habitats across a landscape requires comprehensive information on its distribution so as to elucidate its status and identify priority areas for conservation action (Wikramanayake et al. 1998; Nagendra 2001). Carnivores besides being ecologically pivotal organisms and considered as indicators of habitat quality also serve as umbrella species because their distributional patterns often strongly reflect regional-scale population processes, and hence can act as effective focal species for landscape or ecosystem level conservation (Carroll et al. 2001; Gittleman et al. 2001; Larson 2008). But, acquiring such information on carnivores purely on the basis of field assessment and monitoring across a landscape is nearly impossible (Heywood 1995), particularly in the context of intricate and inaccessible Himalayan landscape (Schaller 1977; 1998). Moreover, the distribution and habitat use by carnivores is known to vary primarily with the availability, distribution and abundance of prey and competitive species than other habitat features (Bothma et al. 1984; Sunquist et al. 1989; Johnson et al. 1996). But, in high altitude landscapes, both biological and geographical factors equally affect their use of an area (Jackson and Ahlborn 1984; Chundawat and Qureshi 1999; Chen et al. 2009), because in these environments resource availability, climatic conditions and biotic as well as abiotic disturbances are highly variable in space and time.

As a way-out, ecological information on a species gathered over an area can aid in predicting its geographical distribution over a larger area. But, the quantification of species-environment relationships represent the core of such predictive geographical modelling since these models are generally based on various hypotheses as to how environmental factors control the distribution of species and communities (Guisan and Zimmermann 2000). A simple approach towards estimating the potential geographic distribution of a species can be to characterize the environmental conditions that are suitable for its presence, and then identify such suitable environments distributed in space (Pearson 2007). With the rise of powerful statistical techniques and application of sophisticated technologies particularly Remote Sensing (RS) and geographical information system (GIS) the development of predictive habitat distribution modelling and mapping of species and their habitats has rapidly increased (Worah et al. 1989; Buckland and Elston 1993; Nagendra and Gadgil 1999; Gough and Rushton 2000; Guisan and Zimmermann 2000; Ramesh 2003; Kushwaha et al. 2004; Habib et al. 2006; Singh et al. 2009). These models statistically relate field observations to a set of environmental variables, most probably reflecting some key factors of the niche, such as climate, topography, geology or land-cover to predict species distributions (Hirzel et al. 2006).

In order to enhance quality and credibility of habitat suitability modelling, predictive techniques have not only become numerous but also improved in recent years (Guisan and Zimmerman 2000) and can be broadly categorized into three types as: (i) expert-based (non-statistical, non-empirical) spatial modelling of species distribution, (ii) non-spatial statistical quantification of species-environment relationship based on empirical data which includes linear regression (Augustin et al. 2001), logistic regression models (Manly et al. 1993), generalized linear models (Guisan et al. 1999), generalized additive models (Seoane et al. 2004) and GRASP ([generalized regression analysis and spatial prediction]- Lehmann et al. 2003), and (iii) spatially explicit statistical and empirical modelling of species distribution involving the delimitation of an hyperspace based on ecogeographical variables (environmental envelope models) and includes BIOCLIM (Busby 1991), HABITAT (Walker and Cocks 1991), DOMAIN (Carpenter et al. 1993), ENFA (Hirzel et al. 2002) and MaxEnt (Phillips et al. 2006). Besides, these another kind of models have recently evolved which are based on Bayesian inference (Ellison 2004) for predicting species or community distributions. All these types of models use simple presence, presence-absence or abundance observations based on random or stratified field sampling to represent the species data required for model preparation, although opportunistic observations on the species occurrence such as those in natural history collections can also serve the purpose (Graham et al. 2004). But generating this presence/absence data is a crucial part of the process. Absence data in particular is often difficult to obtain accurately, since there is a possibility of incorporating false absences due to imperfect detection (McArdle 1990) and historical absence in suitable habitats that can also introduce considerable bias in the analysis (Hirzel et al. 2002).

In the present study, among the above mentioned three categories of species distribution models, spatially explicit statistical and empirical species distribution modelling was attempted to predict the distribution patterns of carnivores in the Khangchendzonga BR. As the statistical species distribution modelling can use two types of species occurrence data (only presence or presence/absence), in this study just the presence-only model was tried since it is effective when there is no reliable absence data available and also requires much less collection effort (Tsoar et al. 2007). Ecological-Niche Factor Analysis (ENFA) operates on similar principles (Hirzel et al. 2002) and also assumes that the environmental conditions are optimal where species is more frequently found (Hirzel et al. 2001). This program is quite efficient and has been used in several studies (Hirzel and Arlettaz 2003) to predict the potential habitat in a large range of animal taxons (plants: e.g., Zaniewski et al. 2002; insects: e.g., Gallego et al. 2004; birds: e.g., Hirzel et al. 2004 and mammals: e.g., Dettki et al. 2003; Zimmermann 2004; Paliwal 2008).

Moreover, as the status of several Himalayan carnivores including the study species such as the charismatic snow leopard, Asiatic black bear, Asiatic golden cat and large Indian civet is vulnerable, and their distribution ranges are either restricted or very narrow; the spatial distribution map of these species in Khangchendzonga BR can not only help in understanding their ecological associations and appropriateness of habitat areas (Lenton et al. 2000), but also help in implementing the necessary conservation actions for protecting their suitable habitats. This chapter therefore deals with an attempt towards generating a spatial database on carnivore distributions in Khangchendzonga BR based on the knowledge from intensive field research combined with spatially explicit statistical models and with the help of advanced GIS technology. Within the extensive range of different modelling approaches, the focus was on the following relevant questions for elucidating the present distribution of carnivores in KBR and hence providing a foundation for proper conservation actions and Protected Area management.

1. How different ecogeographic variables influence the distribution of carnivores in the Khangchendzonga BR landscape?
2. What is the distribution and extent of suitable area for different carnivores in Khangchendzonga BR?

## 6.2 METHODOLOGY

### 6.2.1 Background

Ecological Niche Factor Analysis (ENFA) is based on ecological niche theory and operates on the principle of using reliable presence-only information to compute suitability functions (Hirzel et al. 2002) which besides removing the possibility of incorporating false absences due to imperfect detection (McArdle 1990) also requires much less collection efforts (Tsoar et al. 2007). With the data on species presence in a particular geographic area as input, the ENFA computes suitability functions by matching the species distribution in the Eco-geographic variable (EGV) space with that of the whole set of cells present in that particular geographic area. For each Eco-geographic variable (e.g., slope or elevation) it compares the mean value in cells used by the species with the global mean across the study area to identify the marginality of the species. Similarly, ENFA also compares the variance of each variable in cells used by the species with the variable's variance across the study area to generate an index of specialization (Hirzel et al. 2002). ENFA only weights the EGVs and never reject any of them (contrary to conventional logistic regression techniques), hence avoids the loss of many important Eco-geographic variables due to their spurious correlation with other Eco-geographic variables. This is followed by extraction of a combination of variables that are most important in determining a species' marginality and specialization using factor analysis. The ENFA thereby generates scores for each cell by weighting their corresponding marginality and specialization values (Hirzel et al. 2002). A combination of these scores for all variables is used to generate an overall suitability index for each focal cell, with rankings ranging from zero to one (Hirzel et al. 2002).

### 6.2.2 Method

#### *6.2.2.1 Data preparation*

GIS data-layers for nine Eco-geographic variables believed to possibly influence distribution of carnivores in KBR (Table 6.1) were generated. ArcGIS 9.3 was used to determine slope, elevation, aspect and ruggedness index for each cell of the study area, using a 30-m cell resolution Digital Elevation Model (DEM) [ASTER data, version 2]. Aspect variable was linearized by transforming it into northness i.e., taking cosine of the angle (+ for northern and – for southern). As a surrogate to the vegetation cover, Normalized Differential Vegetation Index (NDVI) for each cell was calculated from the satellite imagery of the study area (LANDSAT data 1990-- No recent image than 1990 were cloud free). Distance maps to different land-cover classes were prepared using ArcGIS 9.3 from the vegetation class map as reported by Sathyakumar et al. (2009). Based on the results of diet analysis of few carnivores (Chapter 5), the information on their major prey was also considered and hence their distribution in the study area was included as an important ecological variable to develop a habitat suitability model for each carnivore, respectively. Using the aforementioned topographical variables and vegetation index, a predictive habitat suitability map of these major prey species in the study area was prepared in Biomapper 4.0 (Hirzel et al. 2007). Presence locations of prey species (blue sheep = 134, musk deer = 49, serow = 79, goral = 78, barking deer = 99, wild pig = 77, [small mammals mainly rodents and pika = 59] and blood pheasant = 101), as recorded from all the habitat types and elevation zones of KBR through trail sampling and camera trap surveys were used for the analysis (Bhattacharya 2013). McArthur's broken-stick model was followed to select the number of factors to be used in preparing the habitat suitability model (Hirzel et al. 2007). These habitat suitability maps for major prey species were cross-validated following the k-fold cross-validation process as described by Boyce et al. (2002). A high Boyce index (blue sheep = 0.72±0.30, musk deer = 0.76±0.34, serow = 0.75±0.22, goral = 0.70±0.41, barking deer = 0.74±0.25, wild pig = 0.71±0.28 (Bhattacharya 2013), [small mammals mainly rodents and pika = 0.92±0.02] and blood pheasant = 0.84±0.23) indicated the validity of these models. The respective habitat suitability maps for major prey species were used as eco-geographical variables along with other nine eco-geographical variables for ecological niche factor analysis and habitat suitability map preparation for carnivores.

**Table 6.1 Descriptions of eco-geographical variables used for ENFA of carnivores.**

| **Eco-geographic variable [a]** | **Description** |
|---|---|
| Elevation | Elevation (m) of individual 30 $m^2$ pixel |
| Slope | Slope (%) of individual 30 $m^2$ pixel |
| Aspect | Aspect transformed into northness as cos (aspect) |
| NDVI | Normalized Difference Vegetation Index reflectance based on satellite data |
| Distance to water | Shortest distance (m) from every 30 $m^2$ pixel to water sources (eg. river, stream, lake) |
| Distance to forest | Shortest distance (m) from every 30 $m^2$ pixel to the nearest pixel classified as forests (including subalpine, temperate and subtropical) |
| Distance to alpine | Shortest distance (m) from every 30 $m^2$ pixel to the nearest pixel classified as alpine(including alpine meadow and alpine scrub) |
| Distance to snow | Shortest distance (m) from every 30 $m^2$ pixel to the nearest pixel classified as open (including rocky area and snow) |
| Ruggedness index | Index of ruggedness centered on each 30 $m^2$ pixel |
| HSI of barking deer | Habitat suitability index (0–100) of individual 30 $m^2$ pixel |
| HSI of goral | Habitat suitability index (0–100) of individual 30 $m^2$ pixel |
| HSI of serow | Habitat suitability index (0–100) of individual 30 $m^2$ pixel |
| HSI of wild pig | Habitat suitability index (0–100) of individual 30 $m^2$ pixel |
| HSI of blue sheep | Habitat suitability index (0–100) of individual 30 $m^2$ pixel |
| HSI of rodents | Habitat suitability index (0–100) of individual 30 $m^2$ pixel |
| HSI of blood pheasant | Habitat suitability index (0–100) of individual 30 $m^2$ pixel |

[a] Box-cox transformation was carried out

#### *6.2.2.2 ENFA and Habitat Suitability map preparation*

Presence locations of different carnivores such as snow leopard (80), red fox (122), yellow-throated marten (116), stoat (38), leopard cat (67), large Indian civet (47), masked palm civet (54), Asiatic golden cat (41) and Asiatic black bear (49) as recorded using GPS during trail sampling and camera trap surveys were used in the analysis to generate the species map. Eco-geographical variables were first normalized as far as possible through Box-Cox transformation (Sokal and Rohlf 1981; Hirzel et al. 2002). Correlations between the EGVs are presented by the global correlation tree (Figure 6.1) as calculated in Biomapper 4.0. Correlation between elevation and distance to snow as well as NDVI and distance to forest were higher than the rest of the EGVs but within the cut off level of correlation ($r > 70$) as following Traill and Bigalke (2007); hence all nine EGVs were primarily used for

ENFA. However, due to the occurrence of very high eigen-values and as suggested by software warnings, few eco-geographic variables respective to different carnivores were discarded from the analysis. Major steps involved in the process of habitat suitability map preparation are described by schematic representation in Figure 6.2. ENFA as similar to Principal Component Analysis (PCA) summarizes all information into a few standardized and uncorrelated factors which unlike PCA contain ecological information (Hirzel et al. 2002). McArthur's broken-stick model was followed to select the number of factors to be used in preparing the habitat suitability model (Hirzel et al. 2002). Geometric mean algorithm was used for the computation of habitat suitability index which computes habitat suitability in environmental space without assuming any distribution of the species points and performs better than the median algorithm to produce a better fit for the bimodal species distribution (Hirzel et al. 2002; Hirzel and Arlettaz 2003).

**Figure 6.1 Global correlation tree depicting correlation structures between eco-geographic variables used for ENFA of carnivores**

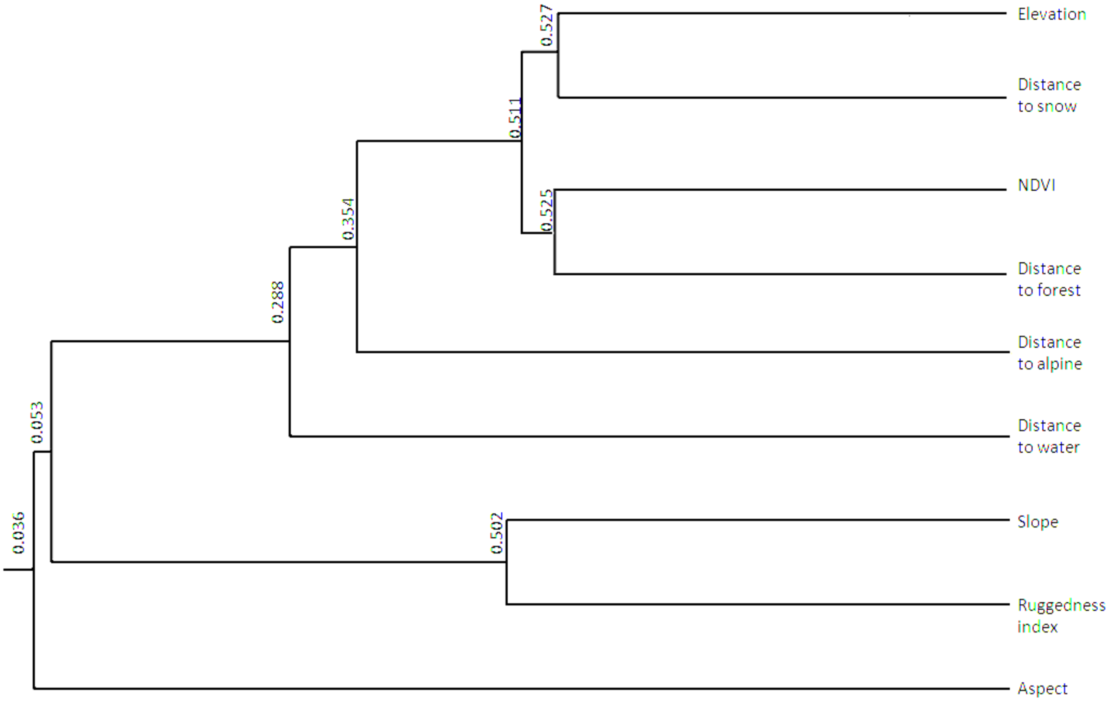
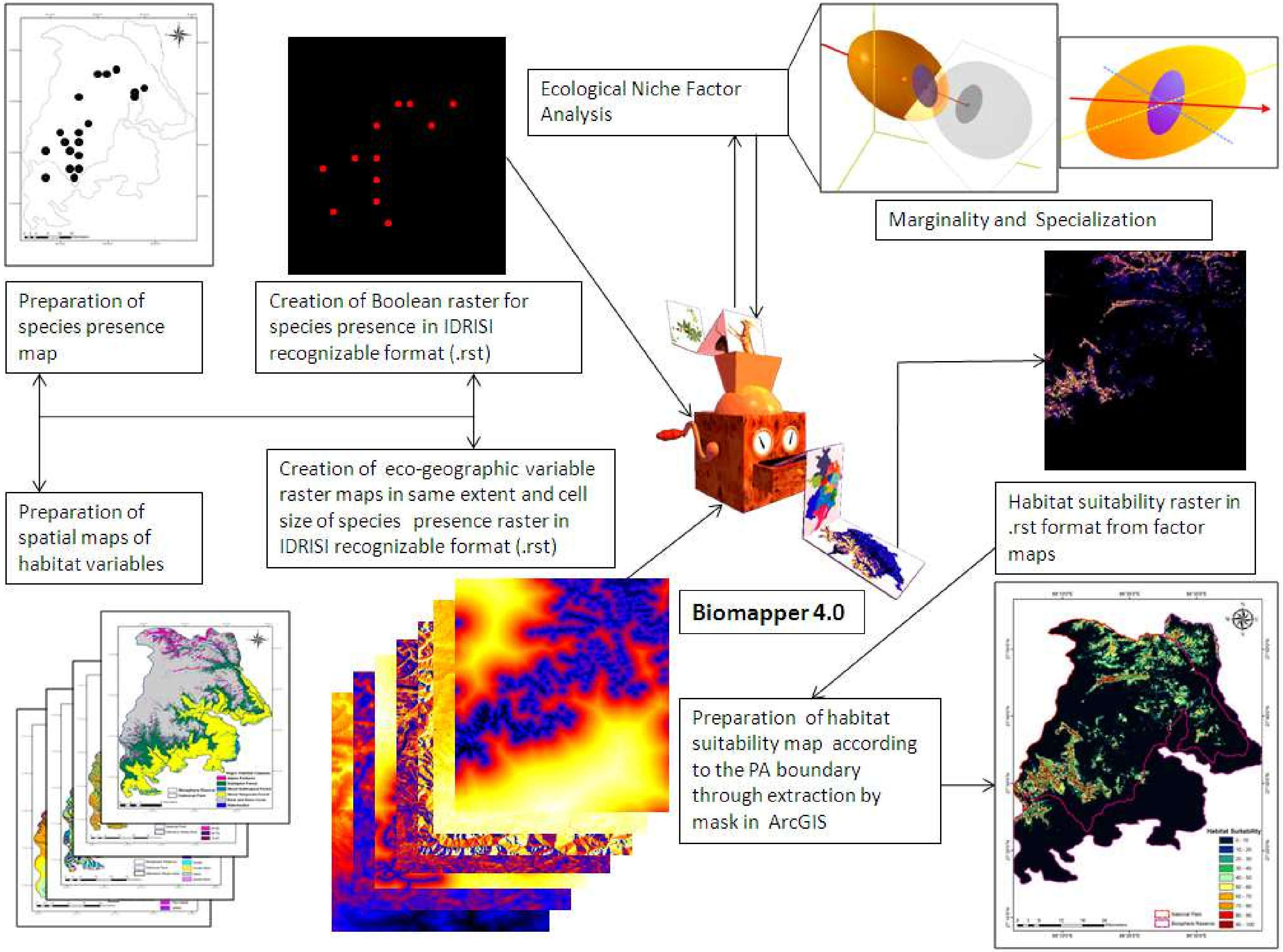


**Figure 6.2 Schematic representation of steps involved in ENFA and model preparation for carnivore habitat suitability in KBR**



### *6.2.2.3 Cross validation of Habitat suitability models*

In order to evaluate the predictive power and accuracy of HS models a Jackknifed 10-fold cross-validation procedure (Fielding and Bell 1997; Ayala et al. 2009) available in Biomapper 4.0 was followed. In this procedure, k=10 mutually exclusive sets are used to partition the species locations randomly. To compute a HS model, k-1 partitions (by default geographically non-overlapping to be more robust to spatial auto-correlation) are used. To validate the HS model on independent data, the remaining one partition is used. The entire process is repeated for k times and their predictive power is assessed by comparing the fluctuations of these HS maps. For each repetition, four model evaluation indices were calculated and represented by their mean and standard deviation across the replicates (Ayala et al. 2009). These were Absolute Validation Index (AVI; 0 to 1), Contrast Validation Index (CVI; 0 to AVI), continuous Boyce index (-1 to +1, 0 indicating a random model) and sequential Boyce index B4 (-1 to +1, 0 indicating a random model). The Absolute Validation Index is the proportion of presence locations in the evaluation partition falling in spatial units with a HSI more than 50. The AVI indicates how efficiently the model can discriminate between high-suitability and low-suitability areas (Ayala et al. 2009; Sattler et al. 2007; Hirzel et al. 2004; Hirzel and Arlettaz 2003). The Contrast Validation Index is the difference between the AVI of a replicate and the AVI of a null model that would predict habitat suitability at random. The CVI indicates how much the model differs from a random model of habitat suitability. These two measures depend on the choice of an arbitrary threshold (in this case, HSI = 50). The threshold-independent Boyce index (Hirzel et al. 2006; Boyce et al. 2002) was used to provide a more continuous and more reliable measure of the accuracy of the model predictions. The continuous Boyce index, calculated through the use of a 'sliding window' across the range of HSI values, measures the monotonic increase of the Predicted to Expected (P/E) frequency ratio with increasing habitat suitability and is computed by the Spearman's rank correlation coefficient between P/E and HSI. This index varies from -1 to 1, with 0 indicating a random model (Hirzel et al. 2006; Boyce et al. 2002). Similarly, the sequential Boyce index B4 was used to assess the predictive power and robustness of the habitat suitability map with best bin combinations of Boyce index B4, and was computed as the Spearman's rank correlation coefficient between P/E and mean HSI value for each of the four HSI classes [Unsuitable (P/E=0), less suitable (0<P/E<1), medium and highly suitable (P/E>1)] averaged over 10-fold resampling (Ayala et al. 2009). Comparison of these four validation indices of HSI models prepared for nine carnivore species indicated the robustness and accuracy of the predictive power of these models.

## 6.3 RESULTS

### 6.3.1 Ecological Niche Factor Analysis

Presence only models being considered to be robust in predicting species distributions; their precision and accuracy (ENFA and HS maps) however depend upon the number and distribution of presence points in the data set. Since, adequate number of presence locations (>30) were not available for all carnivore species recorded from the study area ENFA was done only for nine species of carnivores including snow leopard, red fox, yellow-throated marten, stoat, Asiatic golden cat, Asiatic black bear, leopard cat, large Indian civet and masked palm civet. All these carnivores exhibited high global marginality values (0.84 to 1.78 as shown in Table 6.2), indicating that they occupied a relatively small portion of the set of environmental conditions (defined by the EGVs) available in Khangchendzonga BR. However, black bear, stoat and yellow-throated marten exhibited comparatively lower global marginality values (0.84 to 0.94) suggesting these species to be more 'generalist' relative to other carnivores (Table 6.2). Global tolerance indices (specialization) were generally low, ranging from 0.18 to 0.44 in (Table 6.3) indicating low tolerance towards deviations from the species optimal conditions. Black bear and yellow-throated marten had the highest values of tolerance (0.44 and 0.42), which indicates that they are more likely to colonize sub-optimal habitats than other species. The coefficients of each EGV on the marginality and first specialization factors for the nine carnivore species in Khangchendzonga BR as depicted in Table 6.2 & 6.3, indicates that the marginality factor signifies the best combination of all EGVs illustrating the ecological niche of each carnivore at the geographical scale.

Marginality was particularly important for snow leopard and to a greater extent for large Indian civet and leopard cat as it explained 78%, 58% and 53% of their total model variance, respectively. The occurrence of all carnivores was negatively correlated with distance to water sources (maximum by black bear), and positively correlated with NDVI except for snow leopard (Table 6.2). Model results also indicated that except snow leopard, red fox and stoat, most of the carnivores showed negative correlation with elevation (maximum by civets), and positive correlation with slope and ruggedness index. Further inspection of marginality coefficients of carnivore species revealed a clear pattern of altitudinal distribution as the level of negative correlation decreased gradually as- large Indian civet> masked palm civet> leopard cat> Asiatic black bear> yellow-throated marten> Asiatic golden cat> stoat> red fox> snow leopard. Snow leopard occurrence showed highly positive correlation with the distribution of its major prey (blue sheep), while for red fox the correlation was positive with the distribution of blue sheep, rodents and blood pheasant. Similarly, yellow-throated marten, stoat and leopard cat occurrences showed high positive correlation with the distribution of rodents. The occurrences of Asiatic golden cat showed positive relationship with a number of prey distribution patterns including serow>goral>rodents>barking deer indicating its diverse niche breadth. On the contrary, Asiatic black bear and the two civets did not show any relationship with the prey distributions included in the model. Although, snow leopard, red fox and stoat showed negative correlation with distance to alpine habitat, their slight positive relationship with distance to snow and negative correlation with distance to forest as depicted by the coefficients of EGVs (yellow-throated marten>stoat>red fox>snow leopard), clearly elucidated their pattern of niche preferences.

Inspection of the first specialization factor provided further insights on the ecological niche breadth of the nine carnivore species in Khangchendzonga BR (Table 6.3). This factor alone explained 63% of variation in red fox, however very low percentage of explained variations for this first specialization factor was observed for snow leopard (14%), yellow-throated marten (22%) and leopard cat (23%), suggesting its low effect on niche breadth of these species. All the nine carnivores showed very high level of specialization for elevation, suggesting this EGV to have a major impact in limiting their habitat suitability. Inspection of this first specialization factor for a particular species revealed that civets showed high level of specialization for non-alpine areas. Moreover, high NDVI and ruggedness index were important for leopard cat. However, distance to alpine, distance to water and distance to forest were to some extent important for yellow-throated marten, stoat, golden cat and black bear. In addition, negative correlation with slope and ruggedness index for snow leopard, red fox and stoat, and positive for leopard cat and civets in their marginality coefficients may indicate high preference for flat terrains at high altitudes in case of snow leopard, red fox and stoat and vice versa (steep and rugged terrain at low elevations in case of leopard cat and civets), but their respective coefficients for first factor specialization indicated very low influence of these EGVs on their habitat preferences.

**Table 6.2 Contribution of eco-geographical variables to the Marginality factors of the ENFA for carnivores in Khangchendzonga BR.**

| | MARGINALITY | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | **SL** | **RF** | **YTM** | **S** | **AGC** | **ABB** | **LC** | **LIC** | **MPC** |
| **Variance explained** | 78% | 18% | 48% | 34% | 30% | 35% | 53% | 58% | 38% |
| **Global values** | 1.67 | 1.78 | 0.93 | 0.92 | 1.59 | 0.84 | 1.32 | 1.09 | 1.02 |
| **Elevation** | + | + | --- | 0 | - | --- | --- | ----- | ---- |
| **NDVI** | - | 0 | ++++ | +++ | +++ | +++++ | ++++ | +++++ | +++++ |
| **Aspect** | + | + | + | 0 | 0 | +++ | ++ | ++ | + |
| **Slope** | --- | -- | 0 | - | 0 | 0 | + | + | + |
| **Ruggedness index** | --- | -- | 0 | -- | 0 | 0 | + | + | + |
| **Distance to snow** | + | + | NA | +++ | NA | NA | NA | NA | NA |
| **Distance to alpine** | ---- | ---- | NA | ---- | 0 | - | NA | +++ | +++ |
| **Distance to water** | -- | -- | 0 | ---- | --- | -------- | ------ | ------ | ------- |
| **Distance to forest** | + | - | ------ | ---- | NA | NA | NA | NA | NA |
| **HSI of barking deer** | NA | NA | NA | NA | ++ | NA | NA | NA | NA |
| **HSI of goral** | NA | NA | NA | NA | ++++ | NA | NA | NA | NA |
| **HSI of serow** | NA | NA | NA | NA | +++++++ | NA | NA | NA | NA |
| **HSI of wild pig** | NA | NA | NA | NA | ++ | NA | NA | NA | NA |
| **HSI of blue sheep** | +++++++++ | ++++++ | NA | NA | NA | NA | NA | NA | NA |
| **HSI of rodents** | NA | +++ | ++++++ | ++++++ | +++ | NA | ++++++ | NA | NA |
| **HSI of blood pheasant** | NA | ++++ | NA | NA | NA | NA | NA | NA | NA |

Where; +, ++, +++… indicates intensity of positive influence, -, --, ---… indicates intensity of negative influence, 0 indicates no influence & NA indicates not included in the final model

**Table 6.3 Contribution of eco-geographical variables to the Specialization factors of the ENFA for carnivores in Khangchendzonga BR.**

| | **SPECIALIZATION** | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | **SL** | **RF** | **YTM** | **S** | **AGC** | **ABB** | **LC** | **LIC** | **MPC** |
| **Variance explained** | 14% | 63% | 22% | 46% | 50% | 31% | 23% | 32% | 47% |
| **Global values** | 0.21 | 0.34 | 0.42 | 0.24 | 0.27 | 0.44 | 0.27 | 0.18 | 0.21 |
| **Elevation** | ********** | ********** | ******* | ********* | ********* | ******** | ******* | *** | *** |
| **NDVI** | 0 | 0 | ** | * | *** | ** | *** | ** | ** |
| **Aspect** | 0 | 0 | * | * | * | *** | ***** | 0 | 0 |
| **Slope** | * | 0 | 0 | * | * | 0 | 0 | * | * |
| **Ruggedness index** | * | * | * | 0 | * | ** | **** | 0 | * |
| **Distance to snow** | ** | 0 | NA | 0 | NA | NA | NA | NA | NA |
| **Distance to alpine** | 0 | 0 | NA | ** | ** | *** | NA | ********* | ********* |
| **Distance to water** | * | * | **** | ** | ** | *** | * | 0 | 0 |
| **Distance to forest** | * | * | ***** | ** | NA | NA | NA | NA | NA |
| **HSI of barking deer** | NA | NA | NA | NA | * | NA | NA | NA | NA |
| **HSI of goral** | NA | NA | NA | NA | 0 | NA | NA | NA | NA |
| **HSI of serow** | NA | NA | NA | NA | * | NA | NA | NA | NA |
| **HSI of wild pig** | NA | NA | NA | NA | 0 | NA | NA | NA | NA |
| **HSI of blue sheep** | 0 | 0 | NA | NA | NA | NA | NA | NA | NA |
| **HSI of rodents** | NA | 0 | 0 | * | 0 | NA | 0 | NA | NA |
| **HSI of blood pheasant** | NA | 0 | NA | NA | NA | NA | NA | NA | NA |

Where; *, **, ***… indicates intensity of influence, 0 indicates no influence & NA indicates not included in the final model

### 6.3.2 Habitat Suitability maps

The Habitat Suitability (HS) map of snow leopard predicted a total area of 1224 km² as suitable habitats in Khangchendzonga BR (Table 6.4) out of which 135.7 km² area was delineated as highly suitable (with 60 to 100% probability of species occurrence). Major part of this predicted suitable habitat falls in the south-western part of Khangchendzonga BR indicating the occurrence of a single major zone of continuous favourable habitat confined mainly to *Prek* and *Churong* catchments (Figure 6.3). However, in the northern part of Khangchendzonga BR few interrupted and highly suitable areas restricted to a few patches of trans-Himalayan portion of *Lhonak* valley were also identified. The HS map clearly depicted the flat elevated areas around the *Onglaktang* glacier in southwest and few patches near the *Zema* glacier in north as the best areas for snow leopard with 80 to 100% probability of occurrence. The overall predicted distribution of snow leopard in Khangchendzonga BR was fragmented with a vast chunk of unsuitable habitat including the entire *Rangyang* valley. Most of the favourable snow leopard habitats were inside the NP boundary except few areas at the northern most trans-Himalayan part of the BR depicting high habitat suitability for snow leopard.

Similar to that of snow leopard, the HS map of red fox (Figure 6.4) predicted a more fragmented distribution. The map also identified a single favourable patch of suitable habitat for red fox at the south-western part of Khangchendzonga BR. Apart from this area, the map also predicted the occurrence of few small suitable patches in the northern (*Zemu* catchment) and the eastern (including portions of *Rangyang* and *Lachen* catchments) parts of the BR. Out of the total predicted suitable area of 1163.62 km$^2$ only 108.18 km$^2$ area was highly favourable for red fox (Table 6.4). Most of the high altitude areas including alpine, *krummholtz* and certain portions of sub-alpine habitats of the delineated area were predicted as favourable habitats. In addition, most of the predicted highly suitable area occurred inside the NP boundary except some portion on the south-western part of the BR continuous with the *Churong* catchments.

Contrary to snow leopard and red fox, HS map for yellow-throated marten (Figure 6.5) depicted a continuous distribution across the southwest up to the northeastern parts of the BR. The map predicted yellow-throated marten distribution to be mainly confined to sub-tropical, temperate to upper sub-alpine forests of the KBR. This continuous suitable habitat included parts of *Prek*, *Churong*, *Rangit*, *Rangyang* and *Lachen* catchments. The HS map also delineated a total suitable area of 1360.41 km² (Table 6.4), out of almost 25% (315.6 km²) was predicted to be highly suitable. Moreover, most of the vast chunk of suitable habitat predicted by the HS map laid outside the NP boundary and comprised mainly of the sub-tropical and temperate forests.

HS map for stoat (Figure 6.6) indicated a finely fragmented distribution pattern for the species represented by numerous small patches of high suitability inside the Khangchendzonga BR. This pattern of predicted occurrence was however distribution mainly in southwestern, southern, eastern and partly central parts of the BR. Although, the map predicted a large area of 1332.65 km² as suitable, almost 80% (1030.32 km²) was delineated as less suitable (Table 6.4). In addition, most of this suitable area occurred in the sub-alpine and alpine habitat, particularly along a narrow belt of *krummholtz* edge (highly suitable area). Inspite of the habitat specificity shown by the stoat, a major portion of its predicted suitable habitat (nearly 50%) occurred outside the NP boundary.

Asiatic golden cat as depicted from the HS map showed a similar distribution as that of the yellow-throated marten (Figure 6.7), but was confined mainly to lower altitudes encompassing sub-tropical, temperate and to some extent subalpine forests. Although, the predicted distribution was more or less continuous the highly suitable areas were patchy and sparse, mostly distributed in the southwestern part of the BR. The HS map predicted a total area of 1339.4 km² as suitable out of which only 131.45 km² was delineated as highly suitable (Table 6.4). Moreover, these finite areas of high suitability were situated mainly in the temperate habitat. Similar, to that of yellow-throated marten, most of the predicted suitable habitat of Asiatic golden cat was located outside the NP boundary.

HS map for Asiatic black bear (Figure 6.8) delineated a vast area of 1333.5 km$^2$ (50% of the entire area of Khangchendzonga BR) as suitable for black bear. Inspite of this, almost 70% (906 km²) was predicted as less suitable ([1-30% probability of species occurrence] Table 6.4). Inside the NP, the temperate and lower subalpine forests of the south-western part were predicted as the most favourable habitat for black bear. The overall predicted distribution of black bear was somewhat continuous confined more intensely towards the lower altitudes even outside the NP boundary. Moreover, the distribution of predicted highly suitable patches was less sparse and mainly concentrated in the temperate habitats of southwestern part of the BR.

The HS map for the leopard cat (Figure 6.9) showed majority of the habitat as unsuitable (1,959.44 km²) and only 77.12 km² area was delineated as highly suitable for its occurrence (Table 6.4). The map also revealed a fragmented distribution pattern for leopard cat mainly occurring along the boundary of the BR indicating that most of the suitable habitat occurred outside the NP boundary. Close inspection of the HS map also revealed that all the highly suitable areas/patches were sparse and confined to the lower-elevation temperate forests situated in the southern or south-western parts of the BR and near to the streams.

HS map of large Indian civet (Figure 6.10) depicted the distribution to be most fragmented compared to other carnivores. The suitable areas were confined to seven pockets distributed exactly along the boundary of the BR from the south-western, southern and eastern portions of the Khangchendzonga BR. The HS map delineated a maximum area of 2061.31 km² (almost 80% of the Khangchendzonga BR area) as unsuitable and a total area of just 82.36 km² (minimum among carnivore species) as medium-highly suitable (Table 6.4), thus indicating it’s highly habitat specificity. All the predicted highly suitable areas comprised mainly of sub-tropical forests situated outside the NP.

Similar to the large Indian civet, the HS map of masked palm civet also revealed a highly fragmented distribution for the species (Figure 6.11). This was evident from the delineation of just 25.72 km² as highly suitable out of the total 845.46 km² of predicted suitable habitat (Table 6.4). The suitable areas/patches were also distributed along the boundary of the BR from the south-western, southern and eastern portions of the Khangchendzonga BR. The HS map also revealed that all the highly suitable patches were mainly confined in lower temperate and sub-tropical forests, mostly situated outside the NP boundary.

### 6.3.3 Cross validation of HS maps

The presence-only evaluators AVI and CVI were around 0.5 for each carnivore species, indicating that the HS models were able to discriminate between suitable and unsuitable habitats and that the set of EGVs allowed distinguish specific habitats preferred by each species from the overall habitat available in Khangchendzonga BR. However, high standard deviations for the HS maps of red fox and masked palm civet indicated rather low robustness. Boyce's indices provided a more continuous assessment of the model and predictive map accuracy. The values of these indices were positive and high for all the species. Large standard deviation around the estimates for HS map of snow leopard, red fox and large Indian civet reflected low robustness, especially in the case of the continuous model. However, reclassified HS maps proved more reliable in predicting the distribution of carnivores throughout Khangchendzonga BR for most species (except for snow leopard), as indicated by high and positive values of the Boyce *B4* index- maximal in the case of red fox, yellow-throated marten and leopard cat (Table 6.5). The lowest *B4* index associated with high standard deviation in case of snow leopard suggests that the reclassified HS map for this species should be considered provisional at this stage.

**Table 6.4 Area of suitable habitats (km$^2$) for different carnivore species in KBR as predicted by habitat suitability maps produced from factors of ENFA**

| Species | Unsuitable (<1) | Less (1-30) | Medium (31-60) | High (61-100) | Total suitable area |
|---|---|---|---|---|---|
| Snow leopard | 1395.94 | 948.07 | 140.22 | 135.7 | 1223.98 |
| Red fox | 1456.29 | 864.58 | 190.86 | 108.18 | 1163.62 |
| YT marten | 1259.50 | 636.15 | 408.68 | 315.59 | 1360.41 |
| Stoat | 1287.27 | 1030.3 | 90.72 | 211.61 | 1332.65 |
| Asiatic golden cat | 1479.98 | 494.59 | 513.89 | 131.45 | 1139.94 |
| Asiatic black bear | 1286.43 | 905.98 | 287.40 | 140.10 | 1333.49 |
| Leopard cat | 1917.75 | 541.13 | 83.90 | 77.13 | 702.17 |
| Large Indian civet | 2061.31 | 476.24 | 43.26 | 39.11 | 558.60 |
| Masked palm civet | 1774.46 | 703.08 | 116.66 | 25.72 | 845.45 |

**Table 6.5 Model evaluation statistics for the habitat suitability maps of nine carnivores in KBR. Absolute Validation Index (AVI; 0 to 1), Contrast Validation Index (CVI; 0 to AVI), continuous Boyce index (-1 to +1, 0 indicating a random model) and Boyce index with four bin for cross validation**

| **Species** | **AVI (SD)** | **CVI (SD)** | **Boyce (SD)** | **Boyce B4 (SD)** |
|---|---|---|---|---|
| Snow leopard | 0.52 (0.19) | 0.49 (0.19) | 0.74 (0.43) | 0.48 (0.33) |
| Red fox | 0.50 (0.24) | 0.45 (0.23) | 0.66 (0.32) | 0.96 (0.09) |
| YT marten | 0.49 (0.17) | 038 (0.14) | 0.72 (0.12) | 0.92 (0.11) |
| Stoat | 0.57 (0.10) | 0.47 (0.08) | 0.60 (0.12) | 0.73 (0.31) |
| Golden cat | 0.51 (0.17) | 0.45 (0.16) | 0.84 (0.14) | 0.71 (0.37) |
| Black bear | 0.57 (0.21) | 0.51 (0.19) | 0.71 (0.20) | 0.73 (0.33) |
| Leopard cat | 0.53 (0.15) | 0.52 (0.15) | 0.72 (0.22) | 0.92 (0.11) |
| LI civet | 0.54 (0.14) | 0.52 (0.15) | 0.65 (0.32) | 0.80 (0.28) |
| MP civet | 0.54 (0.32) | 0.52 (0.31) | 0.84 (0.18) | 0.76 (0.22) |

**Figure 6.3 Habitat suitability map predicted for snow leopard in KBR based on the factor maps derived through ENFA**

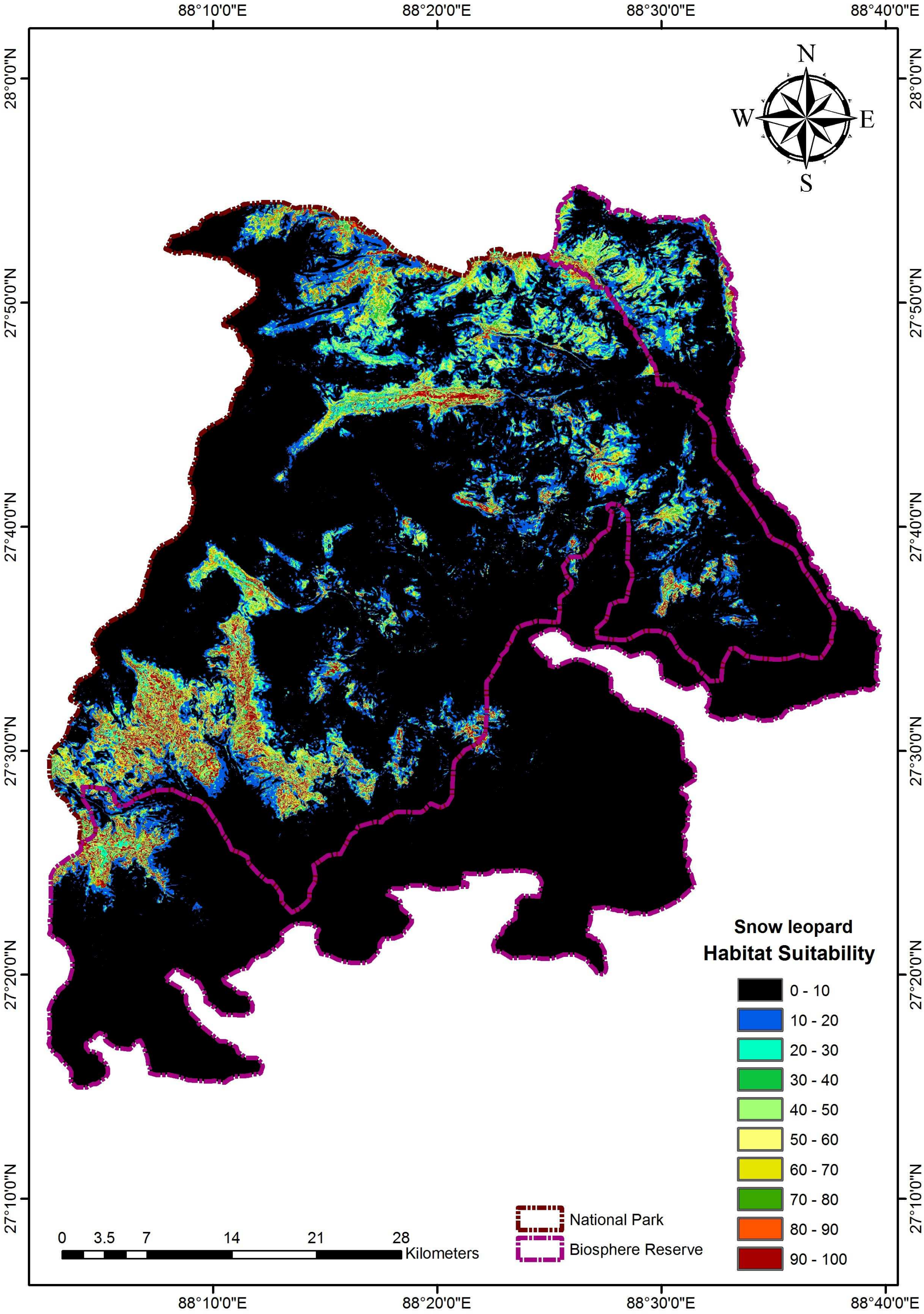


**Figure 6.4 Habitat suitability map predicted for red fox in KBR derived through ENFA**

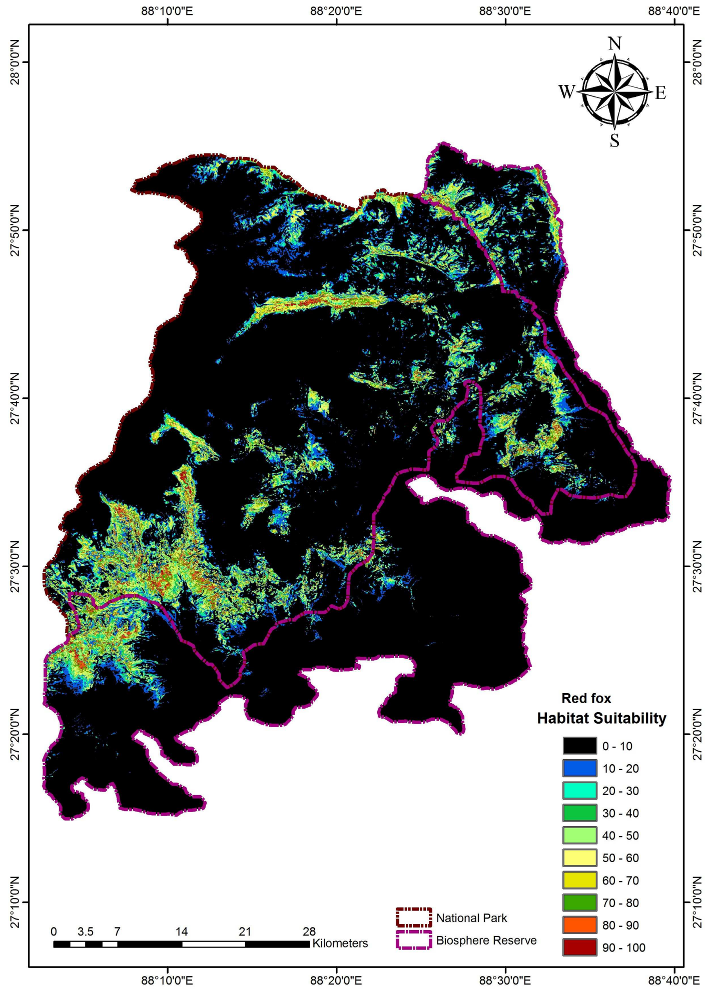


**Figure 6.5 Habitat suitability map predicted for yellow-throated marten in KBR derived through ENFA**

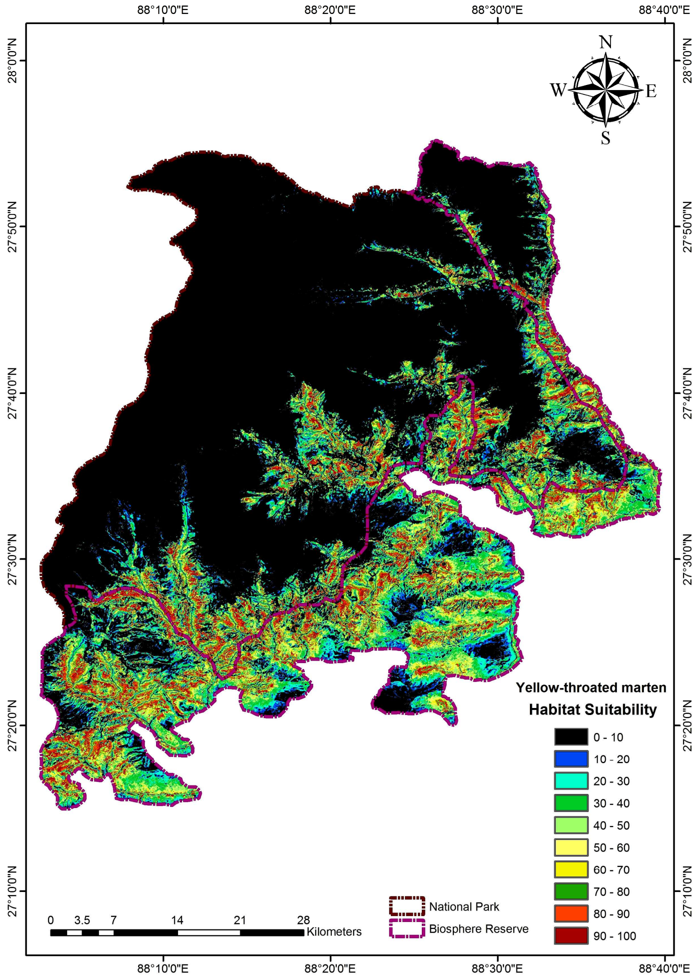


Figure 6.6 Habitat suitability map predicted for stoat in KBR derived through ENFA

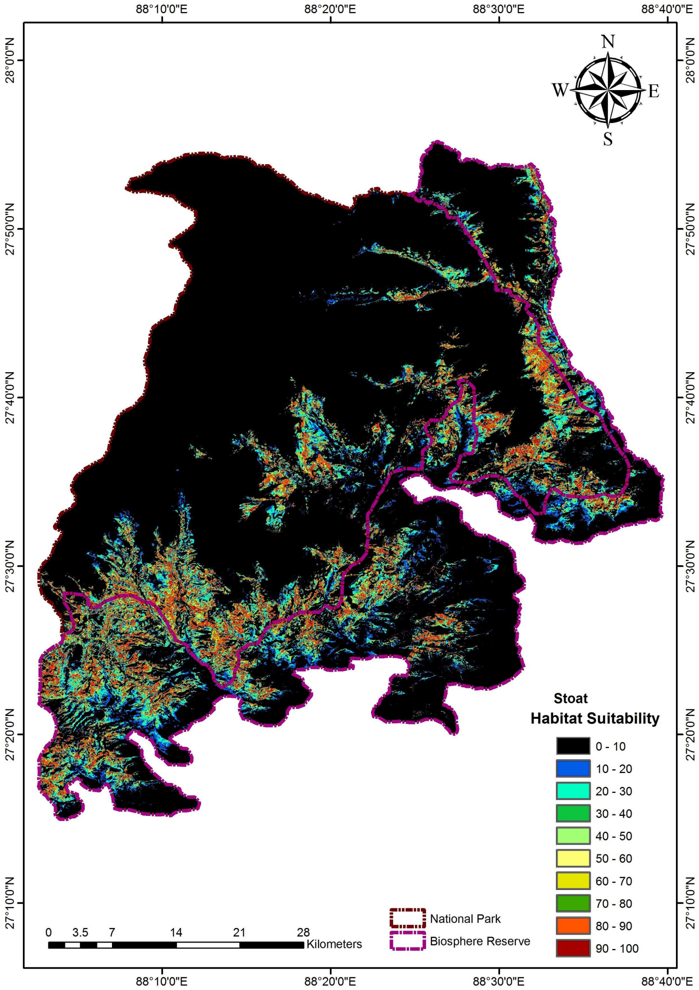


Figure 6.7 Habitat suitability map predicted for Asiatic golden cat in KBR derived through ENFA

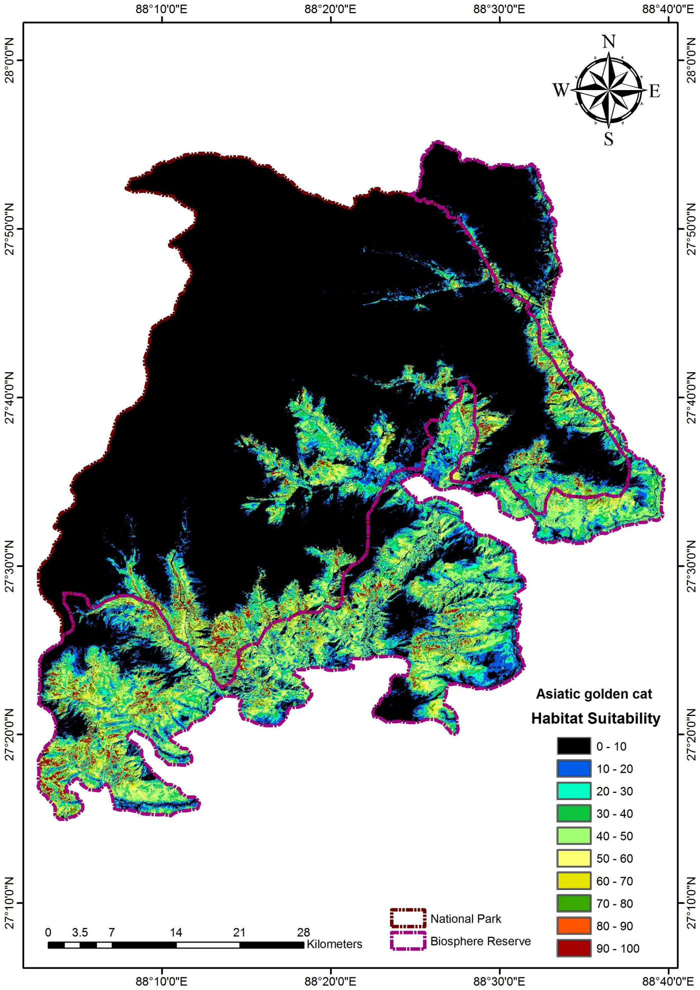


**Figure 6.8 Habitat suitability map predicted for Asiatic black bear in KBR derived through ENFA**

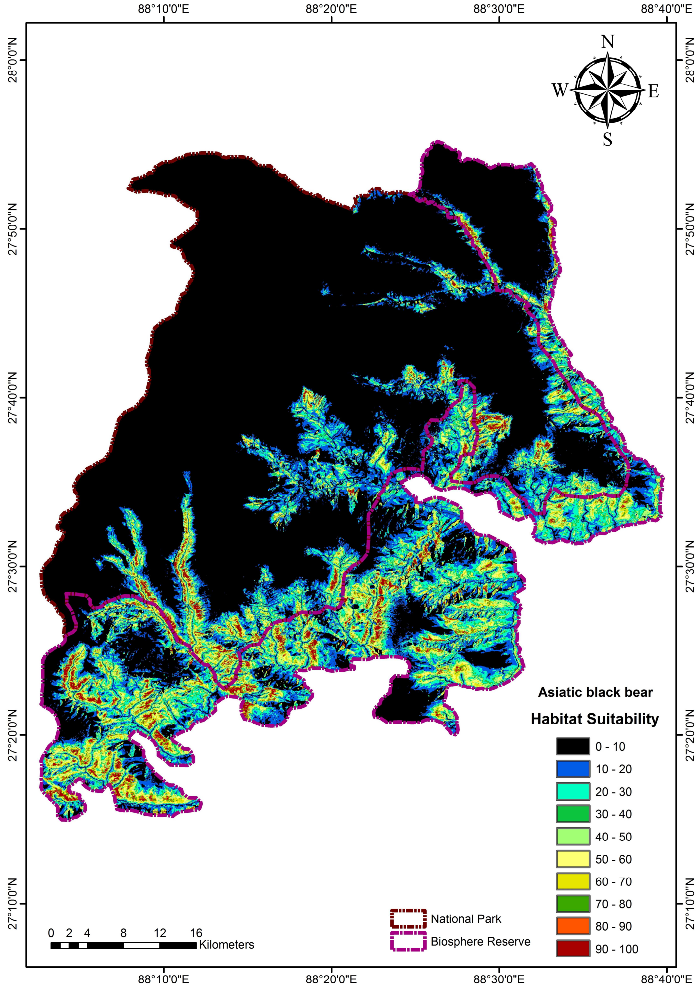


**Figure 6.9 Habitat suitability map predicted for leopard cat in KBR derived through ENFA**

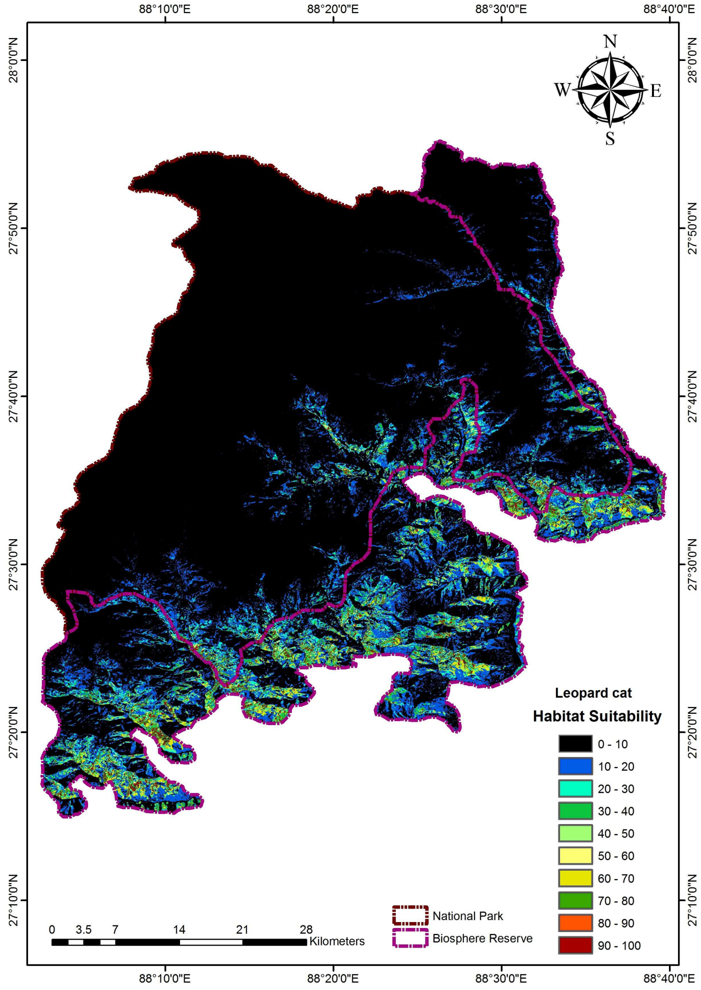


**Figure 6.10 Habitat suitability map predicted for large Indian civet in KBR derived through ENFA**

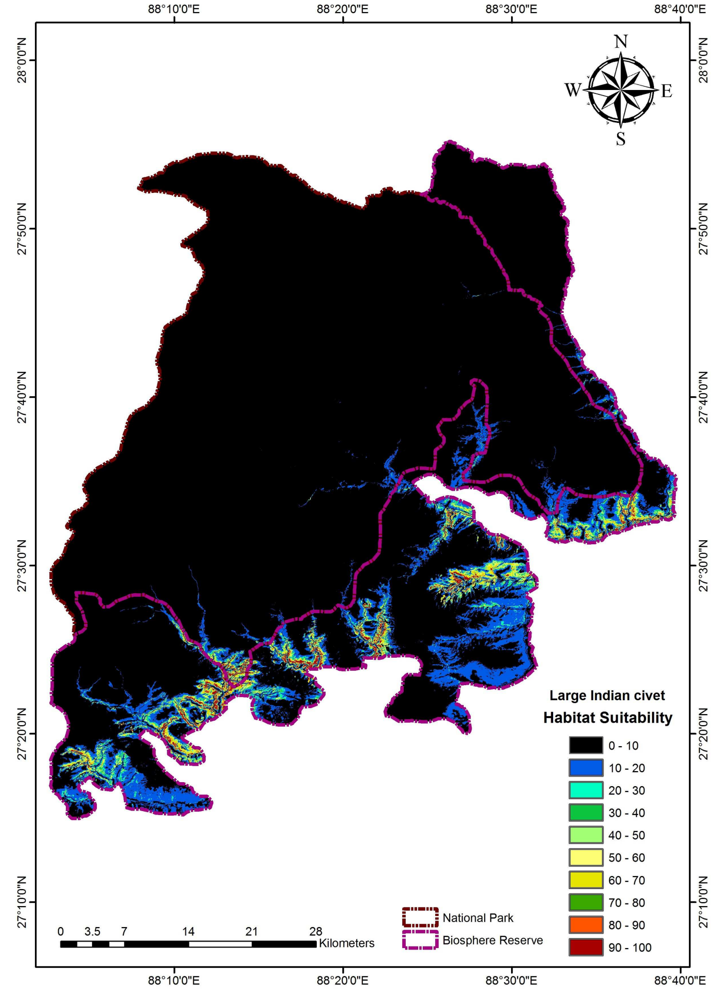
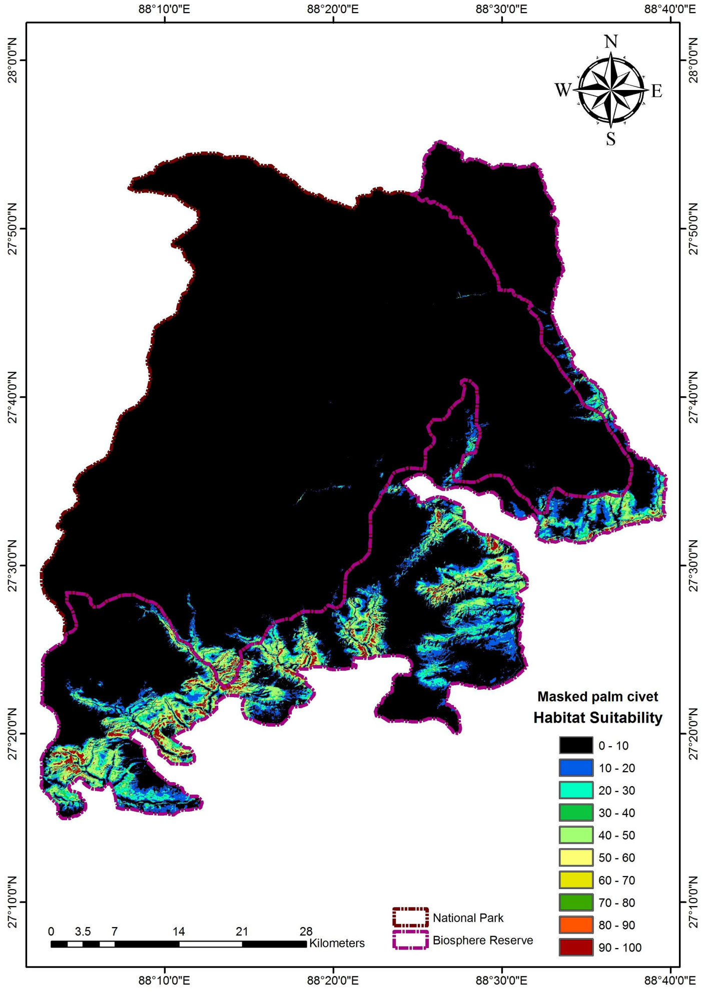


**Figure 6.11 Habitat suitability map predicted for masked palm civet in KBR derived through ENFA**



## 6.4 DISCUSSION

Predictive models of species distributions have progressively being used to address questions related to the ecology, biogeography, and conservation of species (Peterson 2007). While it is increasingly known that the conservation field requires a more evidence-based approach, sound wildlife management policies depend critically on the ability to predict the spatial distribution of species both in their current situation and in the future (Cianfrani et al. 2010). Moreover, for the purpose of wildlife management, research and advocacy, spatially explicit maps can help in a more convincing, advantageous and more easily interpretable way than the numbers and statistics (Buckland and Elston 1993; Cardillo et al. 1999). In this context, habitat suitability models and the maps produced for the distribution of suitable habitats of carnivores in the Khangchendzonga landscape can act as fundamental tools as they support the geographic perspective for conservation strategies (Barbosa 2003; Rondinini et al. 2005). Findings of these models for each species are discussed below:

### 6.4.1 Snow leopard

High global marginality and low specialization (tolerance) values computed by ENFA for snow leopard indicated its occurrence over a relatively small portion of the set of environmental conditions available in Khangchendzonga BR and low tolerance towards deviations from the optimal conditions. Moreover, the results of ENFA model indicated the highest influence of elevation on the distribution of snow leopard. Relationship with other ecogeographical variables indicated its preference for high elevation flat alpine areas distant from forest cover and snow with high distribution of blue sheep as prey. Such pattern of association revealed snow leopard to be a habitat specialist in Khangchendzonga BR with change in elevation acting as a limiting factor for its distribution within its' specialized niche. Similar categorization of unsuitable area for snow leopard with low blue sheep population has been mentioned by Jackson (1996). This habitat preference of snow leopard for the entire Khangchendzonga BR landscape was in accordance with the earlier results of the intensity of space use in the intensive study area (Chapter 5) and also coincided with the findings elsewhere (Jackson and Ahlborn 1984; Jackson 1996; McCarthy and Chapron 2003; McCarthy et al. 2005) except for less preference towards rugged terrain in the present case. Without rocky and broken cover, snow leopards (particularly females with small cubs) could be vulnerable to predation by wolves which are primarily associated with rolling plains and uplands (Jackson 1996). Since, the study did not indicate occurrence of wolves in the south-western part of Khangchendzonga BR but their presence in the northern trans-Himalayan part could act as a threat for the snow leopard population in the present context.

The overall fragmented distribution with very less area predicted as highly suitable, and a vast chunk of unsuitable habitat including the entire *Rangyang* valley (Figure 6.3) and non-connectivity in distribution between the highly suitable zones brings in a matter of concern and illustrates a need for prioritization of suitable areas for the conservation of snow leopard in this landscape. Moreover, the good predictive power, accuracy and robustness of the model as depicted by all four model evaluation statistics (Table 6.5) indicates its appropriateness and hence its applicability for the purpose of conservation and management in the present context. Although, the ENFA map identified a single zone of most suitable area for snow leopard (in the south-western part of the Khangchendzonga BR), it also highlighted some small fragmented areas of high suitability in the northern part of Khangchendzonga BR. Since, this south-western distribution is contiguous with the alpine areas of the adjacent Kanchenjunga Conservation Area, Nepal; and the distribution in the Northern-most boundary (mainly confined in the trans-Himalayan habitats along the *Lhonak* River) is also connected with the transition zone of trans-Himalaya and greater Himalaya situated along the *Zema* glacier and beyond Khangchendzonga BR boundary with the Tibetan plateau, these two areas can be considered of great use for the delineation of areas with high conservation priority. As stated by Forrest et al. (2012) conservation efforts should be prioritized in the places where snow leopards already exist and have high potential, which in the present context refer to the south-western and northern most parts of the BR.

### 6.4.2 Red fox

Similar to that of snow leopard, the ENFA predicted an overall fragmented distribution for red fox and also identified only one major suitable zone in the south-western part of Khangchendzonga BR. Apart from this, it also predicted the occurrence of few small suitable patches in the northern (*Zemu* catchment) and the eastern (including portions of *Rangyang* and *Lachen* catchments) part of the BR most of which occurred inside the NP boundary except some portion on the south-western part of the BR continuous with the *Churong* catchments just above the Barsey Rhododendron Sanctuary. Proper conservation actions are hence needed to maintain the continuity of these suitable areas intact.

High global marginality value of red fox (higher than snow leopard) computed by ENFA also indicated it as a habitat specialist, but comparatively higher value of tolerance index also specified its ability to even adapt in sub-optimal conditions. Moreover, the results of ENFA model indicated similar influence and relationship between red fox distribution and different ecogeographical variables as in case of snow leopard, except for positive relationship with the distribution of rodents and blood pheasant and slight inclination towards forested area (Table 6.2 & 6.3). This pattern of association also validated the overlap in their niches, but also highlighted an element of flexibility in case of red fox and its ability to thrive even in altered habitats (Sillero-Zubiri et al. 2004; Macdonald and Reynolds 2008). These habitat relationships for the entire KBR landscape were also in accordance with the earlier results of occupancy modelling for the intensive study area indicating strong association with *krummholtz* habitat (Chapter 5) and also coincided with the findings elsewhere (Aryal et al. 2010).

### 6.4.3 Yellow-throated marten

By illustrating a continuous distribution for yellow-throated marten mainly comprising of the sub-tropical, temperate up to upper sub-alpine habitats (Figure 6.5) and also delineating a maximum proportion of area as highly suitable for its occurrence compared to other carnivores, the ENFA model depicted the ability of yellow-throated marten to adapt in different habitats. In addition, low values of global marginality and high specialization (tolerance) computed by ENFA also signified it as a habitat generalist with an ability to thrive in a variety of environmental conditions. The habitat relationship of yellow-throated marten calculated by ENFA for the entire Khangchendzonga BR landscape revealing its preference for forested areas at lower elevations rich in rodent distribution were in accordance with the earlier results of occupancy modelling for the intensive study area which indicated strong association with tree cover (Chapter 5) and also coincided with the findings elsewhere as it has been known to favour dense forests with a wide elevation range (Matyushkin 1993; Choudhury 1997b; Than Zaw et al. 2008). Since, the continuous rich belt of suitable area predicted by the ENFA model occurs mainly along the buffer zone of Khangchendzonga BR stretching from south-western to north-eastern parts, this belt demands more attention and protection from the management so as to maintain its connectivity intact as the area also come under the influence of high human interference from the adjoining villages of the buffer zone (Tambe 2007; Tambe and Rawat 2009).

### 6.4.4 Stoat

Although, the ENFA model depicted a fragmented distribution for stoat in KBR and did not specifically identified any distinct highly suitable zones for its occurrence, it revealed that the species tends to occur over a broad elevation gradient covering a diverse range of habitats including sub-alpine and alpine, and particularly along a narrow belt of *krummholtz* edge (highly suitable area). Same pattern of diverse habitat occurrence was depicted by its low global marginality value. Moreover, the positive association with NDVI, forest cover as well as alpine habitat also indicated its adaptability for a wide range of habitats and environmental conditions. These results also coincided with the reports elsewhere indicating its occurrence over a wide range of habitats including forest-edges, alpine meadows riverbanks and coniferous and mixed woodlands (Murphy and Dowding 1995; Samson and Raymond 1998; Martinoli et al. 2001; Smith et al. 2007). However, low global specialization value indicated its low tolerance towards deviations from the optimal conditions. Such relationships with marginality and specialization factors indicate that stoats are widely distributed in Khangchendzonga BR but very sensitive depending heavily on specific factors within this distribution. This was again validated by the prediction of very low proportion (only 20%) of highly suitable habitat by ENFA out of the total delineated potential area in the Khangchendzonga BR.

### 6.4.5 Asiatic golden cat

The ENFA model had a good predictive power, accuracy and robustness as depicted by all four model evaluation statistics particularly the high continuous Boyce index in predicting the distribution and habitat suitability of Asiatic golden cat (Table 6.5). The distribution pattern of golden cat depicted by the ENFA model through HS map (Figure 6.7) was similar to that of yellow-throated marten (throughout the forested areas of Khangchendzonga BR) except the former was confined mainly to lower elevations encompassing sub-tropical, temperate and to some extent subalpine forests. Though, the predicted distribution by ENFA was presumably continuous, areas with high suitability were patchy and sparse and mostly distributed in the southwestern part of the BR. In addition, high global marginality and low specialization (tolerance) values computed by ENFA indicated Asiatic golden cat as a habitat specialist (contrary to that of yellow-throated marten) with its occurrence in a habitat depending heavily on specific factors within that habitat. The influence and relationship with ecogeographical variables on marginality and specialization factors indicated its preference for forested areas at lower elevations along the watercourses. These relationships were in accordance with the earlier findings of occupancy modelling where its occurrence was found to be mainly influenced by vegetation cover (Chapter 5). Moreover, the species has been known to mainly inhabit tropical and sub-tropical moist evergreen forest, mixed evergreen hill forest (Grassman et al. 2005a) and dry deciduous forests (Nowell and Jackson 1996). In addition, positive relationship with the distribution of all forest ungulate species of the area including the rodents (presumed to be possible prey) also indicated their tendency to occupy forested areas rich in mammal prey as they are also know to seldom feed on medium-sized prey (Kawanishi and Sunquist 2008). This brings in the possibility for Asiatic golden cat as the main predator of the forested areas of Khangchendzonga BR in the rare presence of common leopard and clouded leopard (Chapter 4; Sathyakumar et al. 2011a). Moreover, similar to that of yellow-throated marten since most of the predicted suitable habitat patches for Asiatic golden cat occurred outside the NP boundary, these areas demand more protection.

### 6.4.6 Asiatic black bear

The predictions of ENFA model overestimated the continuous proportion and extent of suitable areas for the occurrence of black bear in Khangchendzonga BR represented by a broad belt ranging from sub-tropical to alpine meadows. However, it appropriately delineating the highly suitable patches which were less sparse compared to that of Asiatic golden cat and comprised mainly of temperate forests of the BR along the southern and south-western parts. Moreover, lowest global marginality and highest specialization values (compared to other carnivores) indicated its tendency to occur over a wide range of habitats with high tolerance towards deviations from the optimal habitat conditions. In addition, the relationship with marginality and specialization factors also indicated their preference towards dense forests at low elevations near water sources. But, negative relationship with distance to alpine habitat was ambiguous and might be explained by the possibility of seasonal migration to higher altitudes (Huygens et al. 2003; Garshelis and Steinmetz 2008), recorded in the present case as well (Figure 6.8). Moreover, similar strong association with temperate and coniferous forests was observed in the results of occupancy modelling in the intensive study area (Chapter 5) which also coincided with other studies (Sathyakumar and Choudhury 2007; Izumiyama and Shiraishi 2004; Wang et al. 2008). Although, the highly suitable patches for black bears predicted by the ENFA were comparatively frequent in the Khangchendzonga BR these mostly occurred in the buffer zone. This signals a high possibility of these areas to be susceptible to human-bear conflicts, which hence need to be prioritized, protected and managed efficiently.

### 6.4.7 Leopard cat

The distribution of highly suitable areas for the occurrence of leopard cat was adequately predicted by the ENFA model, highlighting a narrow belt of sub-tropical habitat in the southern and south-eastern buffer zone of Khangchendzonga BR. Predicting a fragmented distribution for leopard cat the ENFA based HS map (Figure 6.9) depicted a very less proportion of the suitable area categorized as most favourable. High global marginality and low specialization values (Table 6.2 & 6.3) also indicated leopard cat to be habitat specialists and less tolerant towards deviations from these optimal conditions at least in the perspective of Khangchendzonga landscape. Moreover, relationships with other ecogeographical variables indicating preference for lower elevation forests were in accordance with the earlier results of occupancy modelling (Chapter 5).

Though this species is widespread in most of its range in India (Mukherjee et al. 2010), in the Khangchendzonga landscape its range seems to be confined to the lower limits of the entire landscape. The preference for areas with high probability of rodent distribution was quite explainable with respect to the results of food habit analysis (Bashir et al. 2013c). Proximity to perennial water bodies has also been regarded as a crucial factor for the distribution of this cat (Sunquist and Sunquist 2002; Mukherjee et al. 2010). However, the preference for lower elevation and cooler western aspects needs to be explained in light of the effect of climatic variables particularly temperature on the leopard cat distribution (Mukherjee et al. 2010). The present global distribution map for leopard cat as recognized by IUCN has depicted the entire Sikkim (including Tibetan marginal mountains) to be inhabited by leopard cats. Contrary to this, a sharp decline in habitat suitability beyond temperate habitat indicates that the lower part of Khangchendzonga BR may represent the upper edge of leopard cat distribution in Sikkim and hence the IUCN map needs to be rectified. Since, most of this highly suitable area for leopard cat occurs in the buffer zone it also needs to be safe guarded in order to check any further fragmentation.

### 6.4.8 Large Indian civet and Masked palm civet

The ENFA model delineated a narrow fragmented belt of suitable habitat for the occurrence of both large Indian civet and masked palm civet in the Khangchendzonga BR landscape including a very less proportion of area predicted as highly suitable. The ENFA model also revealed the distribution of large Indian civet to be confined to just seven distinct small pockets completely separated from each other occurring in sub-tropical forests just along the boundary of the BR from the south-western, southern and eastern portions of the Khangchendzonga BR (Figure 6.10). The highly suitable areas for masked palm civet were also distributed along the boundary of the BR as mentioned above, but in addition included parts of lower temperate forests (Figure 6.11). Moreover, high global marginality and lowest specialization values (compared to other carnivores) calculated by ENFA indicated that both the species were highly habitat specialist (particularly large Indian civet) and also vulnerable to any alteration in their optimal habitat conditions. This was also described through the delineation of very small portion of area as highly favourable for the species (Table 6.4). The relationship of ecogeographical variables with the marginality and specialization factors also indicated their preference for high vegetation areas at lower elevations abundant in water sources. These results were similar to the earlier findings of the occupancy modelling in the intensive study area for the two species (Chapter 5), and elsewhere (Rabinowitz 1991; Duckworth 1997; Azlan 2003; Zhou et al. 2008). Since, the distribution pattern of civets was the most fragmented (particularly large Indian civet), being highly habitat specific and vulnerable to change, these vital patches of occurrence in the buffer zone need to be prioritized and adequately protected.

## 6.5 CONCLUSION

That carnivores will thrive when given habitat that is large, continuous, and protected from poaching and logging seems like common sense. But it is inappropriate to rely on common sense alone to save carnivores particularly in such fragile Himalayan landscapes. Since, the predicted distribution for most of the carnivore species in the extent of Khangchendzonga BR were revealed to be fragmented, such habitat fragmentations can have a wide variety of effects on species including increased mortality, reduced abundance, changes in movement and dispersal patterns, disruption of social structure, reduced population viability, and decreased genetic variation that can lead to reduced adaptability, inbreeding depression, reduced survival and reproduction, and hence increased likelihood of extinction (Dixon et al. 2007). Moreover, the occurrence of most of the highly suitable patches outside the NP boundary in the vicinity of human habitation may instigate threats to these carnivores as similar scenarios of human interference (that modify habitats in ways that alter species interactions) have been reported elsewhere which lead to changes in species distributions and diversity (Collinge 1996; Kurki et al. 1998). Further, since the results demonstrated that carnivores with small portion of suitable habitat even experience higher degree of habitat fragmentation and less habitat connectivity giving rise to smaller and more isolated patches, hence species with less interior habitat and thus higher levels of fragmentation will apparently be at a greater risk of extinction.

Perhaps the greatest long-term threat to carnivores is land conversion (Ginsberg 2001), therefore protection of existing fragments and prevention of further fragmentation is a priority especially for species with higher sensitivity to habitat fragmentation such as snow leopard, large Indian civet, and Asiatic golden cat. Since, the HS models depicted suitable areas outside the NP boundary in the buffer zone, areas adjacent to PA and large fragments are also important and worth protection. Moreover, there is a need for proper delineation of private forest and buffer zone in the Khangchendzonga BR so as to check illegal encroachments and interferences by people inside the PA. Since, the occurrence of suitable areas outside PA as predicted by the models also signals the possibility of carnivore-human conflicts in the delineated areas, the next chapter (Chapter 7) therefore tries to find out answer for this question; whether there exist any conflict between wildlife and humans in the buffer zone of the Khangchendzonga BR.

## Chapter 7

# HUMAN–CARNIVORE CONFLICT

### 7.1 INTRODUCTION

Any ecological study on any species in nature is incomplete without an assessment of human interference because both are dependent and representatives of a single biological community as a whole. All over the world, parks and reserves have become the cornerstone of biodiversity conservation, however in most of the third world countries, conservationists and managers face a series of problems since people are dependent on park resources (Terborgh et al. 2002; Lamarque et al. 2008). This intimate interspersion of people in protected areas has resulted in an increase in human–wildlife conflict over the past few decades (Rodgers 1989; Madhusudan and Mishra 2003; Treves and Karanth 2003). Conflict with humans is a worldwide issue particularly in large carnivore conservation (Nowell and Jackson 1996; Bagchi and Mishra 2004), carnivores often cause serious economic and social losses by preying on livestock, causing damage to property and general community insecurity, and in exceptional cases human injury or even death (Oli et al. 1994; Madhusudan and Mishra 2003; Mishra et al. 2003; Distefano 2005; Ogra 2008).

Conflicts generally arise due to competition for food resources or spatial incompatibility causing direct threat to human or carnivore life; however the most common conflict between humans and carnivores in the Indian subcontinent revolves around livestock and crop damage within protected areas or their buffer zones (Kothari et al. 1989; Kharel 1997; Mishra 1997; Hussain 2003). Since, ungulate species make up a large percentage of their diet the domesticated ungulates present an easy target for predation. However, there are other proximate causes responsible for the escalating levels of carnivore predation on livestock, such as an increase in local abundance of carnivores, increase in livestock populations or decline in wild prey populations (Madhusudan and Mishra 2003). Moreover, as most of the carnivores need vast amounts of space because of their body size, dietary needs, low density, and dispersal distances (Linnell et al. 2001) similar to that of humans due to their large population and high demand for space; spatial competition also leads to conflict.

Conflicts' hence tend to be particularly common in reserve borders where carnivores that rely on extensive territories come into contact with human settlements and hence cause high economic losses (Distefano 2005).

Human-carnivore conflicts pose urgent challenges worldwide because these often pit human communities against carnivores and against those who seek to preserve or restore their populations (Torres et al. 1996). Severe economic losses often generate antagonism towards carnivores' thereby pushing people (livestock owners, farmers, etc) towards retaliatory killings, which substantially affects carnivore populations and thus undermine conservation efforts (Woodroffe et al. 2005; Treves et al. 2006). Such activities have led to the widespread elimination or reduction of carnivore populations in many places around the world (Woodroffe and Ginsberg 1998; Woodroffe 2000). Hence, carnivores not only need conservation but also management that minimizes conflict with humans. Without such management programs large carnivore conservation is likely to be so unpopular that it simply is not feasible (Larson 2008).

An understanding of the human dimensions of resource and ecosystem conservation in Biosphere Reserves is important for preservation and resolution of conflicts related to natural resources (Krishna et al. 2002). Hence, studies centering the quantification of conflict for the sustainable management and conservation of our wild heritage harmoniously with our interests are the need of the hour. A number of attempts have although being made throughout keeping in view these concerns (Oli et al. 1994; Mishra 1997; Hussain 2003; Ikeda 2004; Bagchi and Mishra 2006; Charoo et al. 2011). In the present context, Khangchendzonga BR represents one of the protected areas in India where carnivore-human conflict can be an issue because 1) a major portion of the States geographic area (36.92%) is under BR management (Krishna et al. 2002; Tambe 2007), 2) of the cultural and religious influence on people of the state due to which they are attached towards the natural world and prefer living in proximity with the forests. However, due to increasing development activities and tourism in the state this harmonious relationship is slowly turning unfavorable. In addition, as predicted by the habitat distribution/suitability modelling (Chapter 6), suitable areas for many carnivores laid in the buffer zone of the BR thereby generating a possibility for conflict with people. It was therefore, important to assess the nature and extent of carnivore-human conflict, if any in the BR for proper management. With this background, an attempt was made to answer following research questions during this study:

1. What is the nature and extent of human-carnivore conflict in the Biosphere Reserve?
2. Which are the major conflicting carnivore species of the Biosphere Reserve?
3. What is the status of government compensation programs against losses due to human-carnivore conflict in the Biosphere Reserve?

## 7.2 METHODOLOGY

### 7.2.1 Field surveys and data collection

In year 2009 the extent of carnivore-human conflict in the state of Sikkim was unexpectedly very high (Department of Forest, Environment and Wildlife Management, Government of Sikkim records, *unpublished*). In this regard, field surveys were carried out between June-August and October-December, 2009. In order to assess the level of damage a semi-structured questionnaire (Williams 2003; Karanth 2007) was designed to collect information pertaining to conflict. Data was collected by directly visiting the conflict areas/villages followed by informal interviews with the victims or their family members [Plate 7.1a]. Both close and open-ended questions were administered. The questionnaire was designed in a way to collect information on carnivore species encounters (number, place, and time), crop damage, depredation of their livestock, poultry and human casualties, if any. This also included recording information on GPS location, month and time of the incident, elevation, distance from the nearest forest, crop type and plant part damaged/eaten, number of livestock/poultry depredated/killed, human activity at the time of attack, body part injured and number of casualties, if any. In addition, information on total livestock and poultry holding, expected crop yield, average number of individuals in a household and average monthly income was also recorded. During the interviews, information on different carnivore species sighted in the area and the damage and loss caused in the recent past years was also recorded. Spot checks (Treves and Karanth 2003; Bagchi and Mishra 2006; Sangay and Vernes 2008) were done (wherever possible) to confirm livestock depredation and quantify the damage caused in crop fields. Information was also collected on the protection measures adopted by the villagers to guard their livestock and avoid crop damage caused by wild animals. Specially designed questions were also administered in order to get an overview of the local people’s attitudes towards wildlife (conflicting species in particular), National Park and their views towards conservation (Bahuguna 1986).

In addition, during routine field surveys inside the BR incidents of livestock depredation by carnivores, and information on retaliation, if any was also recorded. Interviews with livestock herders at their herd sites whenever encountered in *Prek* and *Lhonak* catchment inside the BR were also conducted. Telephonic conversation with village heads of remote villages to collect information on the damage was also done. Moreover, newspaper articles published during the June-December 2009 on human-carnivore conflict cases were also used as secondary data sources.

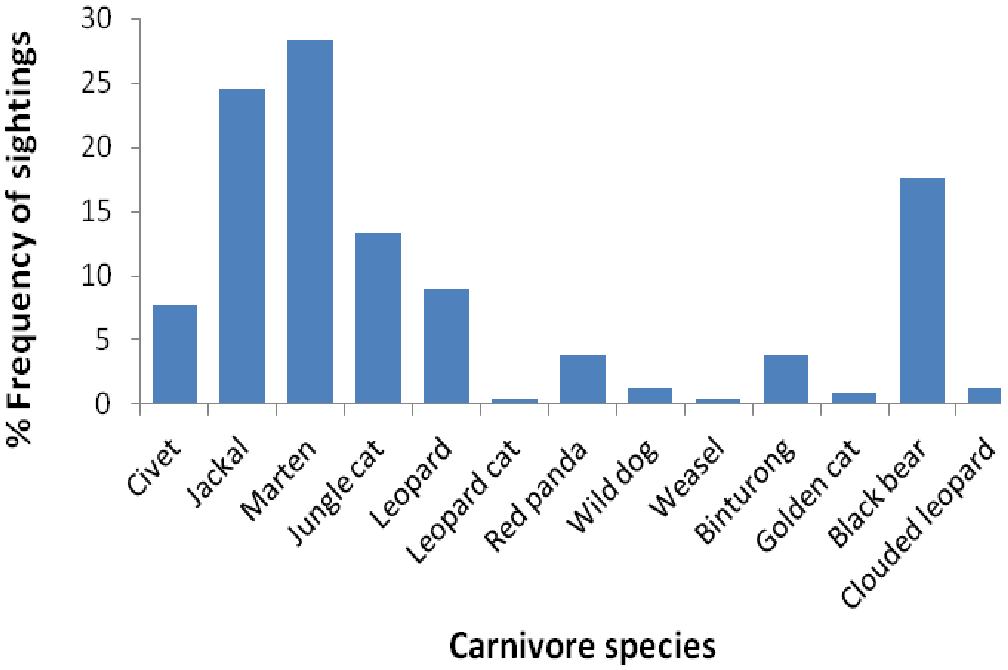
### 7.2.2 Data analysis

Both primary and secondary data recorded was processed and analysed in Microsoft excel and represented in terms of relative frequencies of occurrence. To examine the significance of difference in frequencies of occurrence chi-square tests were employed wherever necessary. Relative frequency of crop raiding along with relative frequency of livestock killing and human attack were pooled and normalized to derive an observed index of damage for each village and the result was presented in a map indicating the highly affected, moderately affected and least affected village. A linear regression model was derived to understand the relationship between conflict incidents with elevation and distance to dense forest. Using this relationship an index was generated for each village predicting its potential of being affected by the conflict. The predicted index was classified as low, medium and high and presented on a map. Finally, a comparison was made between the villages classified according to predicted potential index of carnivore conflict (derived from the model) and the observed index of damage.Economic loss in rupees was also calculated for livestock depredation and crop damage. The average economic value of different classes of livestock species according to their age and sex (cattle, pig, sheep, goat, and horse) and poultry was estimated by interviewing local butchers and livestock traders. Similarly, the economic value for each crop damaged was assessed by considering their rates from the nearby vegetable market.

## 7.3 RESULTS

Over 87 field days were spent in surveying the conflict areas during which 233 affected households belonging to 50 villages [North Sikkim (5), South Sikkim (12), East Sikkim (10), and West Sikkim (23)] were interviewed covering all four districts of the state. On an average 60-100% of the affected households from each village were interviewed. Although, the respondents reported occurrence (through sightings) of at least 20 species of wild animals (including 13 carnivores) in their vicinity (Figure 7.1), the main conflicting carnivore species included black bear, yellow-throated marten, golden jackal, leopard and few small cats. A total of 296 instances of conflict [including crop raiding (207), livestock killing (75), and attack on human (14)] were recorded in the year 2009. The incidents occurred throughout the year with varying frequencies in different months.

**Figure 7.1 Percentage frequencies of different carnivore species sighted by the respondents near their villages.**



### 7.3.1 Distribution of conflict incidents

The distribution of human-carnivore conflict incidents were mainly along the boundaries of protected areas located in four districts of Sikkim. Villages located along the boundary of Khangchendzonga BR experienced the most frequent and highly intense incidents among all three districts (west, south, and north) of its extent

(Figure 7.2). Other affected villages included those located in the vicinity of Barsey Rhododendron Sanctuary in the west, and along the boundary of Fambonglho and Pangolakha Wildlife Sanctuaries in east Sikkim. Number of incidents of conflict were found positively correlated with the elevation of the village and negatively correlated with the distance of the village from dense forest ($R^2$ = 0.59). The relationship is expressed in the linear regression equation as,

$$Y = 2.51 + 1.70E - 4.19D$$

Where, Y= index of human-carnivore conflict, E= Elevation, D= Distance to dense forest

Comparison between classifications of villages according to predicted potential index of human-carnivore conflict (derived from linear regression model; Figure 7.3) and observed index of damage showed 73 % appropriate prediction ($\chi^2$ = 0.081, df = 2, p = 0.96).

**Figure 7.2 Map depicting observed low, medium and high level/index of human-carnivore conflict in different surveyed villages located in four districts of Sikkim in year 2009.**

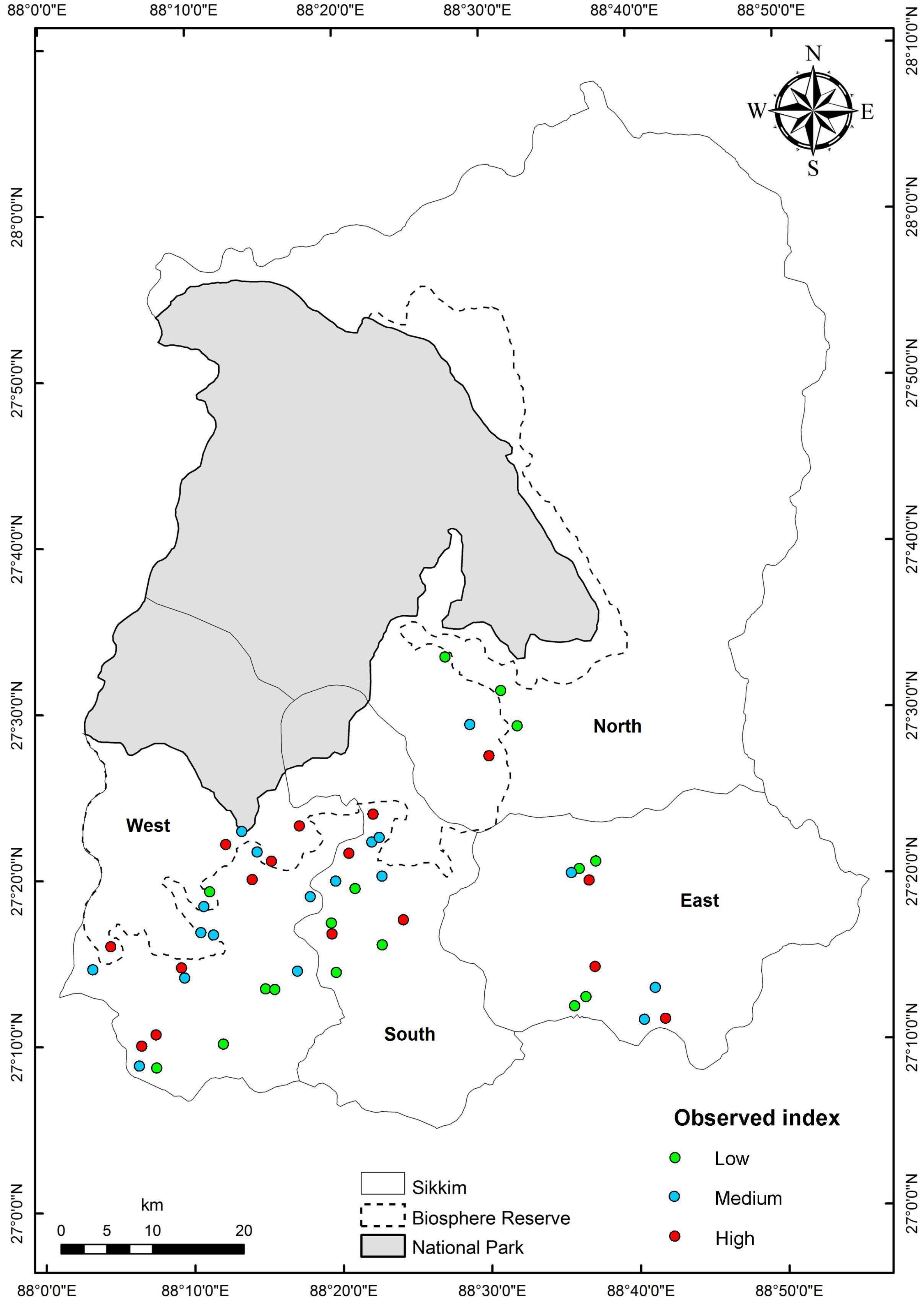
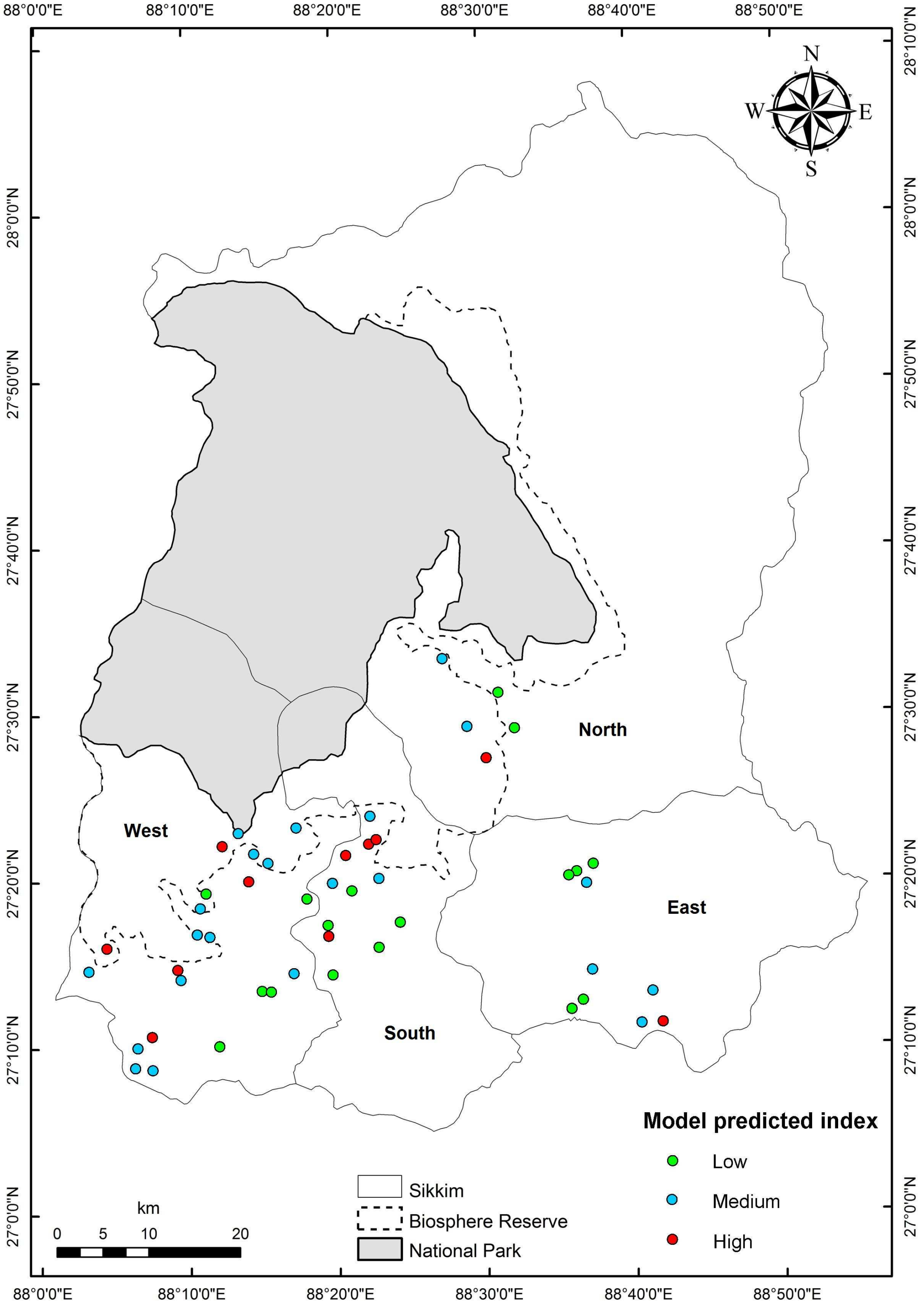


**Figure 7.3 Map depicting predicted low, medium and high level/index of human-carnivore conflict derived through linear regression model in different surveyed villages located in four districts of Sikkim in year 2009.**



### 7.3.2 Crop damage

Incidents of crop damage by carnivores were reported by 89% of the respondents. Among the conflicting carnivore species, crop raiding was done exclusively by black bear. Crop raiding mainly occurred between July and November, but the incidents were mostly frequent (56%) in the months of September-October (Figure 7.4). Total 18 types of agricultural harvests were found to be raided by bears. Maize (*Zea mays*) was the most frequently raided (73% of total incidents) and damaged crop (accounting to about 84% of the total crop damage by weight [kg]) followed by pea (*Pisum sativum*), potato (*Solanum tuberosum*) and squash (*Sechium edule*) [Table 7.1]. In most of the incidents (69%) black bears damaged crops in their mature stage, but in 31% of cases premature corn, leaves and stem of maize, and different parts of other crops such as fruits of squash and peas, tubers of potato, ginger, sweet potato, and roots of other local vegetables, shoots of bamboo in their pre-mature stage were also reported to be consumed and damaged by bears [Plate 7.1b].

**Figure 7.4 Frequency of occurrence of crop damage incidents done by black bear in different months of the year 2009.**

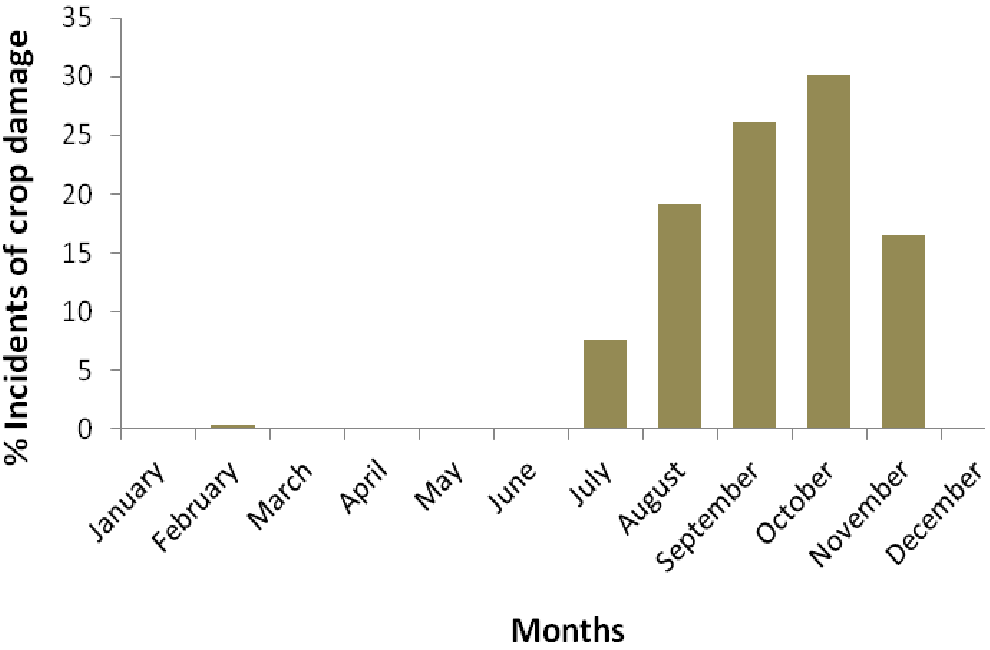


**Table 7.1 List of different crops damaged by bears, number of incidents of damage for each crop (ID), percent incidents of damage (% ID) and percentage contribution by each crop to the total crop damage by weight in kg (% TD).**

| Crop damaged | Common name | ID | % ID | % TD |
|---|---|---|---|---|
| *Dendrocalamus sp* | Bamboo shoot | 3 | 1.45 | 2.43 |
| *Glycine max*.var | Batmas | 1 | 0.48 | 0.11 |
| *Brassica oleracea* | Cabbage | 3 | 1.45 | 0.43 |
| *Daucus carota* | Carrot | 4 | 1.94 | 0.46 |
| *Brassica oleracea* | Cauliflower | 1 | 0.48 | 0.13 |
| *Heracleum sp* | Chimping root | 1 | 0.48 | 0.05 |
| - | Fultarul | 3 | 1.45 | 0.62 |
| *Zingiber officinale* | Ginger root | 3 | 1.45 | 1.40 |
| *Zea mays* | Maize | 152 | 73.43 | 84.80 |
| *Oryza sativa* | Paddy | 2 | 0.97 | 2.97 |
| *Pisum sativum* | Peas | 8 | 3.86 | 1.28 |
| - | Pendalu | 1 | 0.48 | 0.13 |
| *Solanum tuberosum* | Potato | 8 | 3.86 | 2.97 |
| *Cucurbita sp* | Pumpkin | 3 | 1.45 | 0.38 |
| *Glycine max* | Soya bean | 1 | 0.48 | 0.05 |
| *Sechium edule* | Squash | 5 | 2.42 | 0.51 |
| *Ipomoea batatas* | Sweet potato | 3 | 1.45 | 0.30 |
| - | Other leaf vegetables | 5 | 2.42 | 0.97 |

### 7.3.3 Livestock depredation

The incidents of depredation span throughout the year but were more frequent in autumn season during September-November (Figure 7.5). Out of the total 75 incidents of livestock and poultry depredation/killing, most depredation was done by black bear (55%) followed by yellow-throated marten (23%) and jackal (15%) and rest by leopard and small cats [Plate 7.1c]. Among the total number of individuals attacked by carnivores 93% died because of the injuries. Of the total livestock depredated, domestic fowl was killed most frequently (49%) followed by goat (22%) and cow (14%), [Figure 7.6]. Among all the depredated species, domestic fowl were reported to be killed significantly more in proportion to their relative abundance ($\chi^2 = 33.04$, df = 4, $p < 0.0001$). Other livestock such as pig, sheep and ox were also attacked or killed. In terms of number of individuals most of the domestic fowls were depredated by yellow-throated marten (48%) followed by black bear (27%) and jackal (23%), while almost all the depredated livestock including goat (86%), cow (91%), and ox, sheep, pig (all 100%), were killed exclusively by black bear. Overall, the respondents lost 24% of their total livestock (including domestic fowl) to carnivores which included

39% of the total domestic fowl, 22% of pigs and 13% of goats owned by them. In addition, only 11 cases of livestock depredation were recorded in the alpine habitat of the *Prek* catchment during four years of survey attributed to snow leopard, while 4 cases were recorded from the trans-Himalayan part of the Biosphere Reserve in just two months attributed Tibetan wolf [Plate 7.1d].

**Figure 7.5 Frequency of occurrence of depredation incidents done by black bear, yellow-throated marten and golden jackal in different months of the year 2009.**

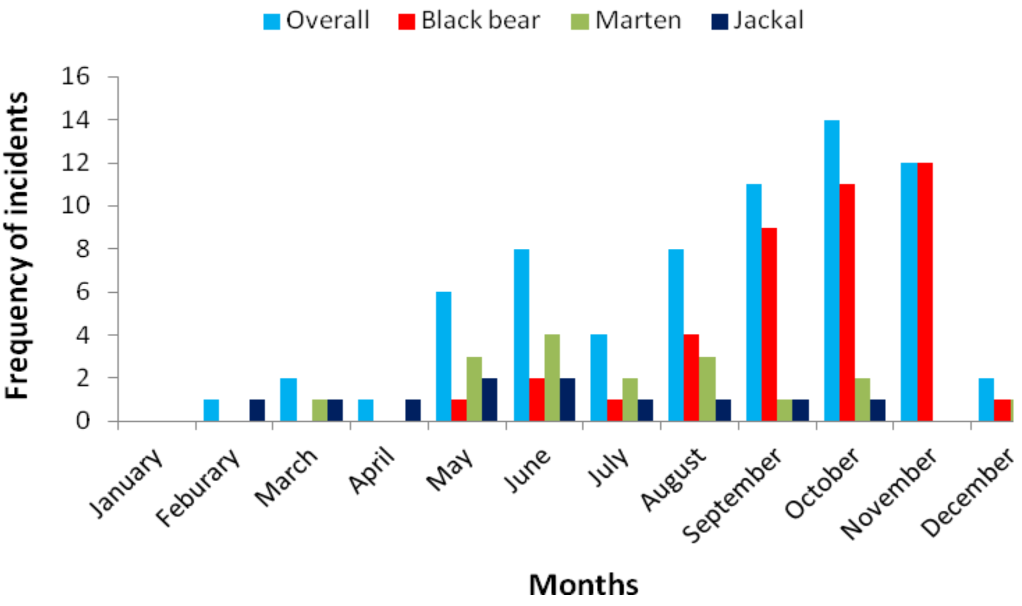
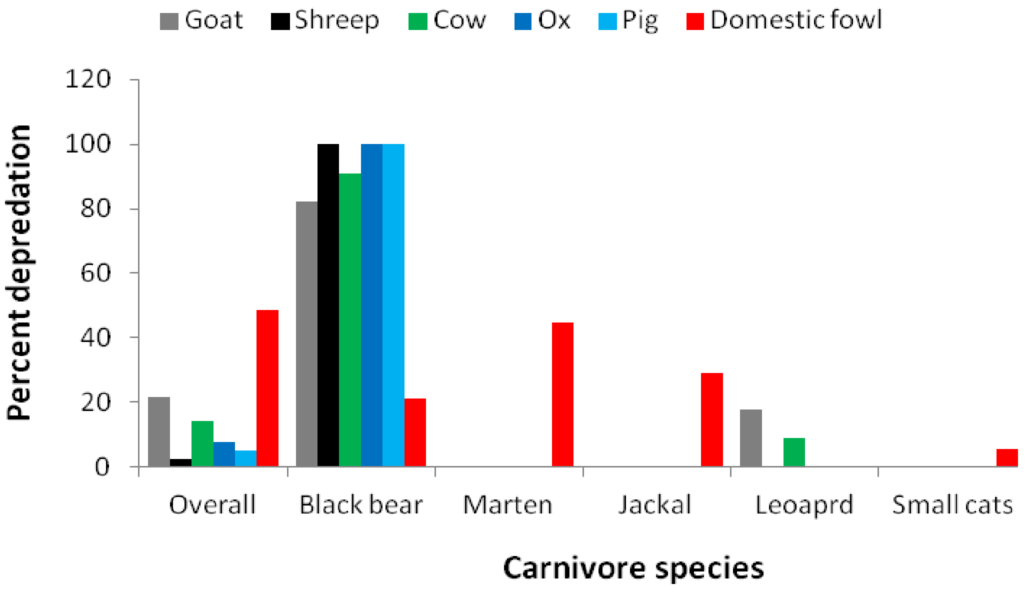


**Figure 7.6 Percent depredation of livestock and domestic fowl by different carnivore species in year 2009.**



### 7.3.4 Human attack

Incidents of human-attack by carnivores were reported by 6% of the respondents. All the incidents of human attack were done by black bears and mostly (92%) took place during the months of October-November, 2009. These attacks mainly occurred in forested areas (65%) when the victims were either busy in collecting fodder/firewood or guarding their livestock inside the forest. However, 20% incidents were also reported from agricultural fields of remote and interior villages and even from densely populated areas (15%) including the state capital *Gangtok* as well. Black bears were sighted by people on 25 occasions (6 occasions with cubs) inside or near the human habitation (20 sightings in villages and 5 in *Gangtok*). The attacks mainly occurred during crepuscular hours between 7-8 hrs in the morning (25%) and 3-5 hrs in the evening (35%). Of the total 14 incidents, 13 were major attacks which in three incidents resulted in the death of victims.

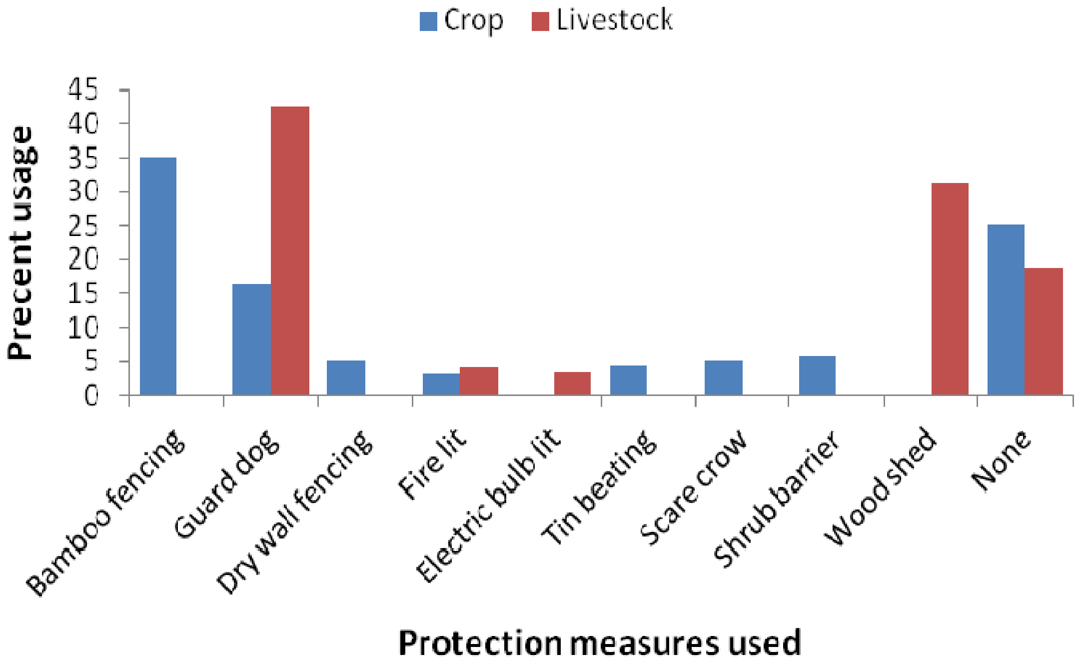
### 7.3.5 Economics of conflict and compensation

The average monthly income per interviewed household was estimated to about ~ 2,800 INR to sustain an average of seven family members (range 1-16). Human-carnivore conflict in the year 2009 resulted in a total loss of ~ 700,000 INR (precisely 700,145 INR). The loss due to crop damage amounted to 391,145 INR (1,679 INR per household), while livestock and poultry depredation by carnivores resulted in a loss of 340,000 INR (4,533 INR per household). The average loss per household was calculated as ~ 3,150 INR (range 200-15,250 INR). In response to this only 20% of the affected households were given compensation by the wildlife department (total amount = 47,800 INR). This included an average amount of ~ 1,000 INR (range 300-3,000 INR) given to each household which was even much less than the loss caused by the conflict.

### 7.3.6 Protection measures against conflict

In order to protect their crops and livestock from damage caused by the conflicting carnivores, the villagers used various strategies (Figure 7.7). These mainly included bamboo fencing of the crop field done by 35% of crop farmers, use of guard dogs (16%) and shrub barriers around the crop fields (6%), while about 25% farmers did not use any type protection measure. For the protection of their livestock, a majority of livestock owners used guard dogs (43%) and wood sheds (31%) for the purpose, however about 19% owners did not use any protection measure. Besides this, small wooden/wire sheds were used by all villagers to safeguard their domestic fowl.

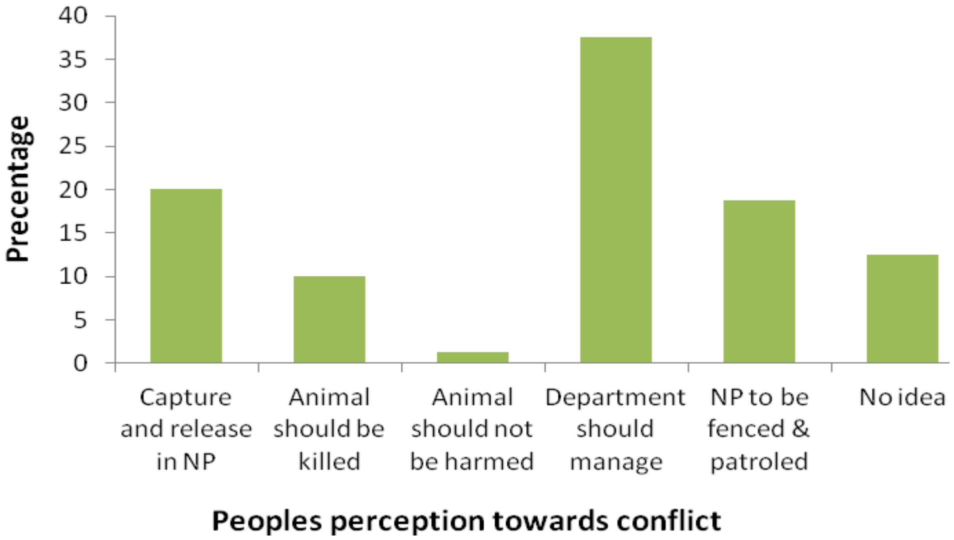
**Figure 7.7 Percentage uses of different protection measures by locals to guard their crops and livestock.**



### 7.3.7 People's perception

About 22% of the respondents had a positive attitude towards the conflicting species suggesting that the animals should not be harmed but captured and released back to the National Park. However, 10% believed that the animal should be killed (Figure 7.8). Although, a major proportion of people (37%) put the blame on wildlife department and urged that they should properly manage these animals. In fact people (19%) even suggested that the National Park should be fenced and regularly patrolled to check any movement of wild animals out of the PA so as to avoid conflicts. However, few incidents of retaliatory persecutions of black bear (6 individuals), snow leopard (4-5 individuals) and leopard cat (1 individual) [Plate 7.1e] by people were also recorded during casual conversations and from secondary sources.

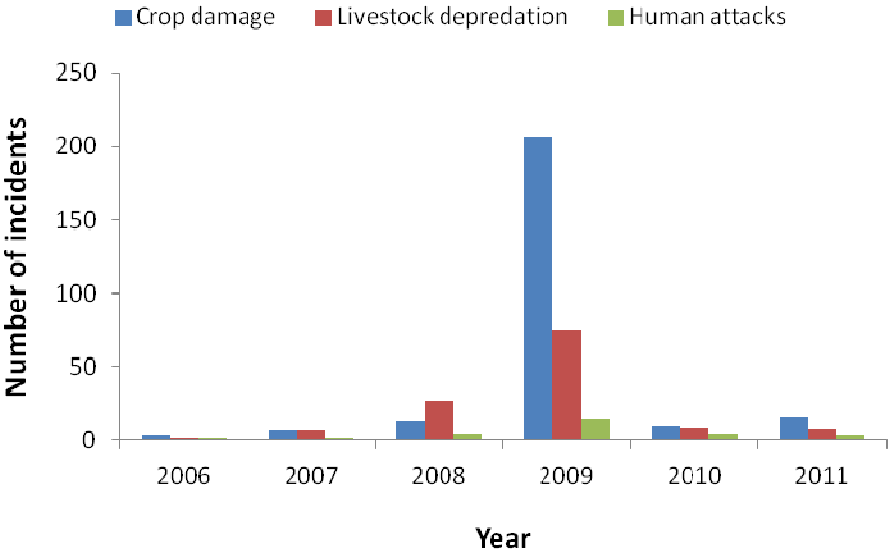
**Figure 7.8 Perception of villagers towards the conflicting animals and suggestions to check conflicts.**



### 7.3.8 Conflict trend over the years

Increasing trend was observed in the number of incidents of conflict in the past few years with highest frequency of occurrence in 2009 (Figure 7.9). Crop raiding was the most common mode of damage compared to other losses except in 2008 where livestock and poultry depredation was more recurrent. Moreover, conflict incidents were also reported in years 2010-11, but were less frequent.

**Figure 7.9 Frequencies of occurrence of crop damage, livestock depredation and human attacks by carnivore species over the years.**



## 7.4 DISCUSSION

Although, the study presents a snapshot of the perceptions and opinions of people living in the fringe villages of the protected areas and lacks actual field quantification of loss, the information generated on the human-carnivore conflict in this study is of high significance not only for understanding the intensity of the conflict but for its appropriate management. The results clearly indicated that the conflict incidents were concentrated mainly along the boundaries of the protected areas evident from the relationship with distance to forests (Distefano 2005; Honda et al. 2009). Moreover, positive relation with elevation (indicating more closeness to wildlife habitats) and high predictive power of the model also demonstrated that villages in close connectivity with wildlife habitats were more susceptibility to conflicts. In similarity, the occurrence of carnivore species in the vicinity of surveyed villages were also reported around two protected areas (Fambonglho Wildlife Sanctuary and Pangolakha Wildlife Sanctuary) in the East Sikkim (WWF 2011). The information generated in this study also validated the increasing trend in conflict incidents in Sikkim over the years with its maximum incidence in 2009 (Department of Forest, Environment and Wildlife Management, Government of Sikkim records, *unpublished*).

### 7.4.1 Damage to crops

Inspite of the fact that other carnivores such as jackal, civets and marten tend to be partially frugivorous in their habits (Poche′ et al. 1987; Zhou et al. 2008, 2011), the crop damage was done primarily by black bears indicating their tendency towards plant food due to their omnivorous habit (Garshelis and Steinmetz 2008). Damage to 18 differed crop types including food grains, tubers, pulses, cereals, and other vegetables and consumption of almost all plant parts (roots, shoots, fruits) even in their premature stage by bears clearly demonstrates the spread and extent of damage. Moreover, the occurrence of crop raiding incidents mainly between July and November were in unison with the production time of majority of the crops particularly maize i.e., September-October. This indicates that attraction towards food resources could be one of the driving forces of crop damage by bears (Honda et al. 2009), reported in case of sloth bears as well (Bargali et al. 2005). Since, maize contributes to over 50 percent of the state’s total food production (Lama 2007), the damage to maize crop directly translate as a considerable loss to both personal consumption and local economy. Potential damage to maize crop by bears was also reported in a parallel study on human-wildlife conflict conducted in east Sikkim (WWF 2011) except for the reason that the study reported macaques as the major culprits. Similar intensity of crop damage by bears as in the present case has also been recorded in Senchal Wildlife Sanctuary, Darjeeling (Sunar et al. 2012), Dachigam National Park, Kashmir (Charoo et al. 2011), and Kanchenjunga Conservation Area, Nepal (Khatiwada 2008).

### 7.4.2 Depredation of livestock and poultry

Livestock depredation by carnivores known to be the most common conflict between humans and carnivores (Thirgood et al. 2005) formed the second major component of human-carnivore conflict in the state of Sikkim after crop damage. Since, a number of carnivore species were involved in conflict, the pattern of livestock and poultry depredation span throughout the year. But the incidents were more frequent in the autumn season during which omnivorous and frugivorous species (black bear, yellow-throated marten, and jackal) get more attracted towards crop fields and villages and thereby cause damage to livestock as well. Most of the livestock damage was done by black bears during the same season (autumn) as reported in Bhutan (Sangay and Vernes 2008). Although yellow-throated martens and jackals also contributed to some proportion of loss, but it primarily included killing of domestic fowl. In coherence with the present results, black bears have been known to be involved in livestock depredation in the Indian Himalayas (Sathyakumar 2001; Chauhan 2003; Sathyakumar and Choudhury 2007) and similar incidents of damage have been reported from Dachigam National Park, Kashmir (Charoo et al. 2011), Senchal Wildlife Sanctuary, Darjeeling (Sunar et al. 2012), and even from Dhorpatan Hunting Reserve, Nepal (Yadav et al. 2009). The reason for very few incidents of depredation by common leopards may be either due to their occurrence in very low densities or distribution confined to lower elevation forests of Sikkim (Chapter 4).

Of the total damage, domestic fowl was killed most frequently followed by goat and cow, and depredation of domestic fowl was primarily done by yellow-throated marten, black bear and golden jackal. Almost, same pattern of depredation was observed in east Sikkim (WWF 2011) reporting about 91% depredation of domestic fowl, thereby identifying yellow-throated marten and jackal as their main predators. Yellow-throated martens are highly adaptable and agile creatures that have made themselves at home in the village outskirts. It almost seemed as predation on domestic fowl went on a regular basis and presumably construction of re-enforced pens and presence of watchdogs did not help substantially. A number of incidents were reported when either a yellow-throated marten or jackal forced itself inside the pen and killed several adult birds including chicks. Poultry depredation by yellow-throated marten and small cats has been reported quiet frequently in Western Aurnachal Pradesh (Mishra et al. 2006; Datta et al. 2008b) and in the Kangchenjunga Conservation Area, Nepal (Khatiwada 2008). Reports from western Pakistan have also stated that yellow-throated martens were found killing domestic chicken (Roberts 1970). Moreover, in accordance with the current results more than 61% domestic fowl kills were attributed to yellow-throated martens in the villages around Singalila National Park, Darjeeling (DLR Prerna 2012). Being regarded as one of the most adaptable canids in the world (Giannatos 2004), golden jackals have also been known to depredate on cattle calves and lambs, but reports about their poultry lifting are more common (Sarker and Ameen 1990; Yom-Tov et al. 1995; Khatiwada 2008; Kait and Sahi 2012).

Since, few incidents of livestock depredation by other carnivores such as snow leopard and Tibetan wolf were also recorded in the high altitude alpine habitats and trans-Himalayan cold deserts of the Khangchendzonga BR where snow leopards seldom depredated on feral yaks and dzos left by the herders in alpine meadows for unsupervised grazing. But, based on the number of incidents and interviews with the people it seemed that the extent of livestock damage was more intense in the trans-Himalayan part of the Biosphere Reserve where Tibetan wolf was regarded as the main species of conflict. This was even confirmed by the semi-nomadic *Dokpa* and *Bhutia* pastoralists, which were based in *Munguthang*, *Thangu*, and *Lachen* villages reporting loss of their yaks and dzos primarily to wolves, though they also reported depredation by snow leopards. Although, livestock depredation by snow leopard and Tibetan wolf is known to be a serious conservation issue and has been reported across the Himalayan region particularly in the trans-Himalayas (Oli et al. 1994; Jackson et al. 1996; Mishra 1997; Mishra et al. 2003; Jackson and Wangchuk 2004; Khatiwada 2004; Bagchi and Mishra 2006; Maheshwari et al. 2012; Suryawanshi et al. 2013), the loss assessed in the present study is comparatively less. However, extensive surveys need to be conducted in order to generate more information on the depredation patterns by these carnivores particularly in the trans-Himalayan part of north Sikkim.

### 7.4.3 Attacks on humans

Attacks on humans are the most serious concerns in human-carnivore conflict because of their adverse effect on the attitude and determination of people towards wildlife conservation. Bear attacks on humans have been reported throughout the Himalayas with varying intensities (Chauhan 2003; Choudhury et al. 2008; Charoo et al. 2011). In the present case, attacks on humans by bears occurred mainly inside forested areas when people were collecting fodder/firewood or grazing their livestock. This indicates heavy encroachment by people into the bear habitats and their dependence on forest products which consequently resulted in these attacks. Supplementing the information generated in this study, overall 49 humans-bears encounters were reported in Sikkim during 2009 resulting is nine human casualties (WWF 2011). The timings of these attacks also coincide with the observed daily activity patterns of bears assessed during the present study (Chapter 5). On the contrary, bear attacks in densely populated areas (such as *Gangtok*) could be attributed to their search for easy food (garbage dumps) even in developed areas as has similarly been reported in western North America (Beckman and Berger 2003).

### 7.4.4 Preventive measures to avoid conflicts

It was observed that in spite of using bamboo fencing, watchdogs and wooden sheds/pens as prime protection measures for guarding crop fields and livestock, a considerable proportion of respondents reported no use of any such protection strategies. Even more staggering statistics were reported from east Sikkim where almost 69% respondents had reported not using any measures in spite of suffering considerable crop and livestock loss (WWF 2011). Since, the efficiency of even the main protection measures was reported to be often challenged by the conflicting carnivores (black bear, yellow-throated marten and golden jackal), the limited use of preventive strategies by villagers further increases the risk of damage to carnivores.

### 7.4.5 Economic loss, compensation, people’s perception and retaliation

The overall estimated economic loss per household due to conflict in the present case was in fact very less compared to what has been reported in other studies (Kargil, Ladakh ~ 27,000 INR- Maheshwari et al. 2012; Jigme Singye Wangchuck National Park, Bhutan ~ 10,000 INR- Wang and Macdonald 2006; Kibber Wildlife Sanctuary ~ 6,400 INR- Mishra 1997). But while considering the economic status of people it seems to be substantial. Moreover, due to very less amount compensation given even to just 20% of the affected households, a sort of negative attitude seems to have developed among the people towards the wildlife department and the wild animals. This was evident from the responses given by about 47% of the victims when they suggested either killing the conflicting animals or blamed the wildlife department for their losses. Since, the occurrence of conflict incidents have increased over the years and the compensation schemes have not been effective in meeting the losses, this growing negative attitude has even caused an increase in the frequency of retaliatory killings in certain cases.

During 2008, high livestock depredation by leopards mainly in the west Sikkim causing heavy economic loss to the villagers was followed by baited retaliatory persecution of the leopards using dynamite (Lachungpa *pers comm*). This might also explain the reduced livestock depredation by leopards in the consecutive years, which are otherwise known to be major predators of livestock in similar habitats (Wang and Macdonald 2006; Sangay and Vernes 2008; Dar et al. 2009). Use of dynamite has also been reported among the agro-pastoralist communities living in villages around the Kibber Wildlife Sanctuary for killing wolf pups (Mishra 1997). Retaliatory killing of snow leopards by the semi-nomadic *Dokpa* and *Bhutia* pastoralists in the trans-Himalayan part of north Sikkim which is known to be one of the major issue in snow leopard conservation throughout its distributional range (Oli et al. 1994; Jackson et al. 1996; Mishra et al. 2003; Jackson and Wangchuk 2004; Bagchi and Mishra 2006; Maheshwari et al. 2012; Suryawanshi et al. 2013), seems to hold for Sikkim as well and needs to be addressed. Moreover, since black bear was the main conflicting carnivore responsible for crop damage, livestock depredation and human attacks, incidents of bear killing were reported from different parts of Sikkim and in one extreme case in *Sangkhu* village (west Sikkim) even its consumption by the villagers in order to destroy the evidence of persecution was also reported. Such incidents of retribution killing of black bears have also been reported in other parts of its distribution range (Stubblefield and Shrestha 2007; Charoo et al. 2011). In addition, presence of baited snare traps for small carnivores in the village fringes and occurrence of a dead leopard cat (with broken hind limb) in the border of *Yuksam* village (west Sikkim) also demonstrated the increasing negative attitude of people towards wildlife.

### 7.4.6 Major cause of conflict

Worldwide human-animal conflict is caused by various animals under various circumstances. Both the scenarios of human-carnivore conflict were observed in Sikkim. One which included movement of wild animals into the villages and causing crop damage and livestock depredation, and the other which involved attacks on humans and livestock inside the wildlife habitats during fodder/firewood collection and livestock grazing. Although, livestock grazing practices have been banned inside the Khangchendzonga National Park (Tambe 2007), unsupervised grazing of pack animals such as feral yaks, dzos and horses still takes place in some areas which makes them vulnerable to predation by carnivores (Tambe and Rawat 2009). In addition, supervised grazing of sheep and goat in the alpine habitats during the autumn season also leads to conflict and loss of livestock to carnivores such as snow leopard. Other reasons that increase the risk of conflict with carnivores are the poor preventive strategies adopted by the villagers for guarding their crop fields and livestock. Since, the organized illegal collection of bear food plants such as bamboo shoots, oak acorns and walnuts by villagers during the autumn season has increased over the years, the incidents of human-black bear conflict also increases towards the end of autumn because this period coincides with the pre-hibernation fattening season of the bears. During this season the bears are known to become highly active and tend to travel long distances in search of food. However, if there is a shortage of wild fruits, especially oak (Hwang and Garshelis 2007; Hwang et al. 2010), they repeatedly end up outside forest areas and walk into nearby human settlements looking for food, and thereby give rise to conflicts. Moreover, illegal encroachments into the forest land due to improper demarcation of private forest and buffer areas around the National Park have also increased the occurrence of conflict incidents.

### 7.4.7 Measures for combating conflict

In order to counter the high intensity of conflict incidences in 2009, the department of Forest, Environment and Wildlife Management, Government of Sikkim had adopted multi-step strategy which included field patrols, quick relief schemes and awareness campaigns. This also included the recruitment of 143 forest guards to enhance the conflict management. Personnel were assigned to different field camps strategically located in vulnerable locations across the state to rapidly follow up any report of black bear intrusion. Awareness camps were also held at key locations across the state and pamphlets detailing preventive measures and contact numbers of key personnel were distributed. Information was also spread though different media mediums. To supplement the black bear diet in the forest and to reduce the probable shortage of natural food, “feeds” were put inside strategic locations in the fringe areas near Protected Areas particularly located in East and West Sikkim. Vide a departmental notification dated 27/2/2009, relief payments to victims of black bear and other main carnivore attacks were increased to 10,000 INR (for minor injury), 50,000 INR (for major injury) and 1, 50000 INR (for human death). Similarly, compensation amounts for the loss of different livestock species and for crops damage were also raised. Many more such conservation measures are necessary that can help in better management of the conflict issues. The next chapter (Chapter 8) addresses few more conservation threats to carnivores in the study area, deals with developing adequate monitoring protocols for these carnivores and also attempts to provide some recommendations for their proper conservation and management in this landscape.

**Plate 7.1 (a) Informal interviews conducted during the surveys, (b) Maize crop damage by black bear, (c) Depredation of domestic goat by leopard, (d) Depredation of feral yak by Tibetan wolf, and (e) Retaliatory killing of leopard cat at *Yuksam***

# Chapter 8

# MONITORING AND CONSERVATION

## 8.1 INTRODUCTION

Across the world, people have set aside some portions of land as protected areas in which habitats are relatively unaltered for the purpose of biodiversity conservation (Larson 2008). These areas have been widely accepted as the most effective means of preserving biodiversity (MacKinnon et al. 1986; IUCN 1994; Licona et al. 2011). But, it has also been well acknowledged that the future of most protected areas depends on the degree to which local people’s concerns, needs, and aspirations are addressed by conservationists (Jackson and Wangchuk 2004). In order to achieve this, the Biosphere Reserve model was applied for the selection and management of 553 protected areas in about 170 countries across the globe (UNESCO 2010). However, to ultimately achieve success in conserving the biodiversity, wildlife management and monitoring needs to be coupled closely to permit articulation of specific objectives for evaluation at the site- or regional level. Given the natural complexity of wildlife populations and habitats they inhabit, and the severe constraints on resources available for monitoring, a key challenge in designing a monitoring program is to successfully determine priorities among all possible indicators so as to select those for measurement that best reflect the status and dynamics of the system under management (Gibbs et al. 1999).

The Khangchendzonga BR in the Sikkim Himalaya offered an opportunity to investigate the implications and possibilities of biosphere reserve concept in the enormously diverse eastern Himalayan landscape. Since its inception as a Biosphere Reserve in 2000, the Khangchendzonga BR landscape experienced several policy level changes and modifications which may have altered the livelihood practices of the local communities and also changed the habitat status of wild animals (Tambe and Rawat 2009). To assess these changes in context of the conservation of wildlife (particularly the key indicators such as large carnivores and their prey) in the core and buffer zones of KBR, there was a serious need for the development of baseline ecological information on these taxa (Tambe 2007, Sathyakumar et al. 2011a).

Moreover, a recent landscape level remote sensing study in KBR (Tambe et al. 2012) also revealed that the park management needs to evolve innovative co-management models, take adequate safeguards while using the riverine zone, strengthen buffer zone management and focus conservation measures in high impact areas for the long-term security of this unique mountain landscape. To achieve these goals, multidisciplinary ecological and socioeconomic research is believed to be needed (Tambe and Rawat 2009). This chapter is based on the findings of the previous chapters (4, 5, 6 and 7) focusing on the implications of the ecological study on carnivores to achieve a larger goal of Biosphere Reserve management by integrating the ecology and conservation of this landscape.

## 8.2 MONITORING OF CARNIVORES

### 8.2.1 Adequacy of monitoring methods for detecting carnivores

The adequacy of different monitoring methods (sign surveys, trail monitoring [sightings] and camera trapping) for detecting carnivore species in different habitat zones of the study area was compared. In order to assess the adequacy of camera trapping, minimum number of camera trap stations and camera days (1 occasion = 10 days) required to detect all carnivore species in each habitat zone was calculated. Since the camera trapping effort was continuous in all habitat types (though the intensity was not similar); calculations for the adequacy of camera trapping effort was done after identifying the best seasons (sampling session) for monitoring carnivores in different habitat zones during which their probability of detection could be maximum and the accessibility and feasibility of the area would be optimal. Similarly, for assessing the adequacy of sign surveys and trail monitoring (sighting), calculations of minimum number of sampling replicates (1 occasion = 10 repeats) required to detect all carnivore species in each habitat zone was carried out. In addition, based on the adequacy and efficiency of different sampling methods to detect different carnivore species in the study area, an attempt was made to compare and recommend the appropriate method (s) for monitoring different carnivore species in this intricate terrain of Sikkim Himalaya.

As mentioned earlier (Chapter 4), the presence of 19 species of carnivores was confirmed from the study area. Among different sampling methods, camera trapping proved to be the most suitable method in detecting more carnivore species than other methods (sign surveys and trail monitoring [sightings]) in all three habitat zones of the study area (Figure 8.1 a, b, c). It was also observed that a minimum camera trapping effort of 10 (alpine), 14 (sub-alpine) and 10 (temperate) sampling occasions are required for the detection of almost all carnivore species in the respective habitat zones of the study area (Figure 8.2). Moreover, for alpine zone at least12 camera trap stations are required for the same purpose (Figure 8.2) and the best season for monitoring is May-August. For sub-alpine zone a minimum nine camera trap stations are required and the most appropriate season for monitoring is January-May. While, in case of temperate zone a minimum of 11 camera trap stations are required for detecting the presence of most carnivore species inhabiting the area and the best season for monitoring is October-December, respectively. A comparison between different methods and recommendation of appropriate method for monitoring different carnivore species in the study area is given in Table 8.1.

**Table 8.1 Comparison and recommendation of methods for monitoring carnivores in Khangchendzonga BR, Sikkim**

| Species | Sign survey | Trail/Transect | Camera trap |
|---|---|---|---|
| Snow leopard | √ | × | √ |
| Common leopard | × | × | √ |
| Clouded leopard | × | × | √ |
| Asiatic golden cat | × | × | √ |
| Jungle cat | × | × | √ |
| Leopard cat | √ | × | √ |
| Red fox | √ | × | √ |
| Tibetan wolf | √ | × | √ |
| Wild dog | √ | × | √ |
| Jackal | √ | √ | √ |
| Asiatic black bear | √ | × | √ |
| Red panda | × | × | √ |
| Masked palm civet | × | × | √ |
| Large Indian civet | × | × | √ |
| Binturong | × | × | × |
| HYT marten | √ | √ | √ |
| Stoat | × | × | √ |
| Pale weasel | × | × | √ |
| Siberian weasel | × | × | √ |

**Figure 8.1 Efficiency of different sampling methods to detect carnivore species in a) alpine, b) sub-alpine, and c) temperate habitats of *Prek chu* catchment, KBR**

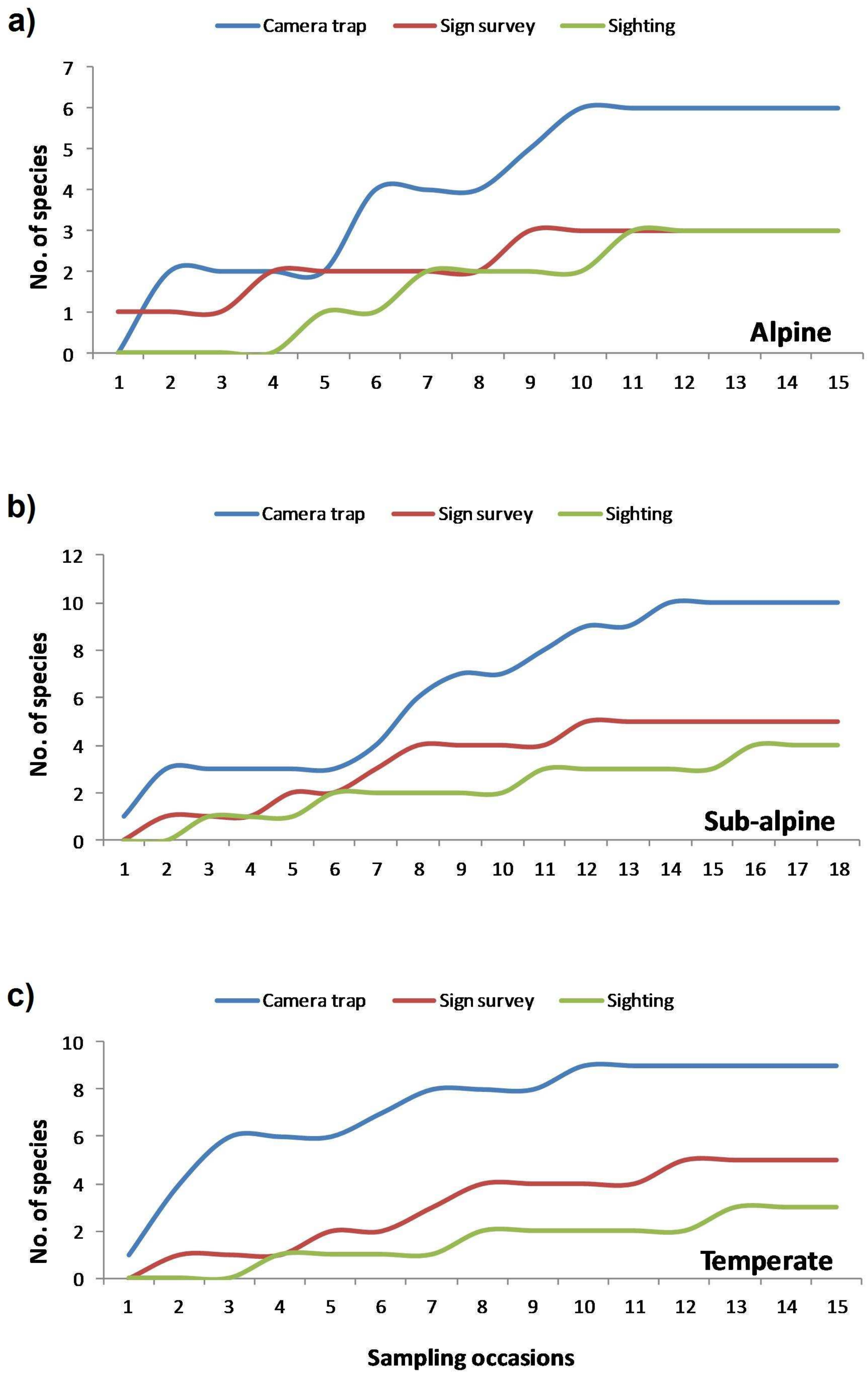
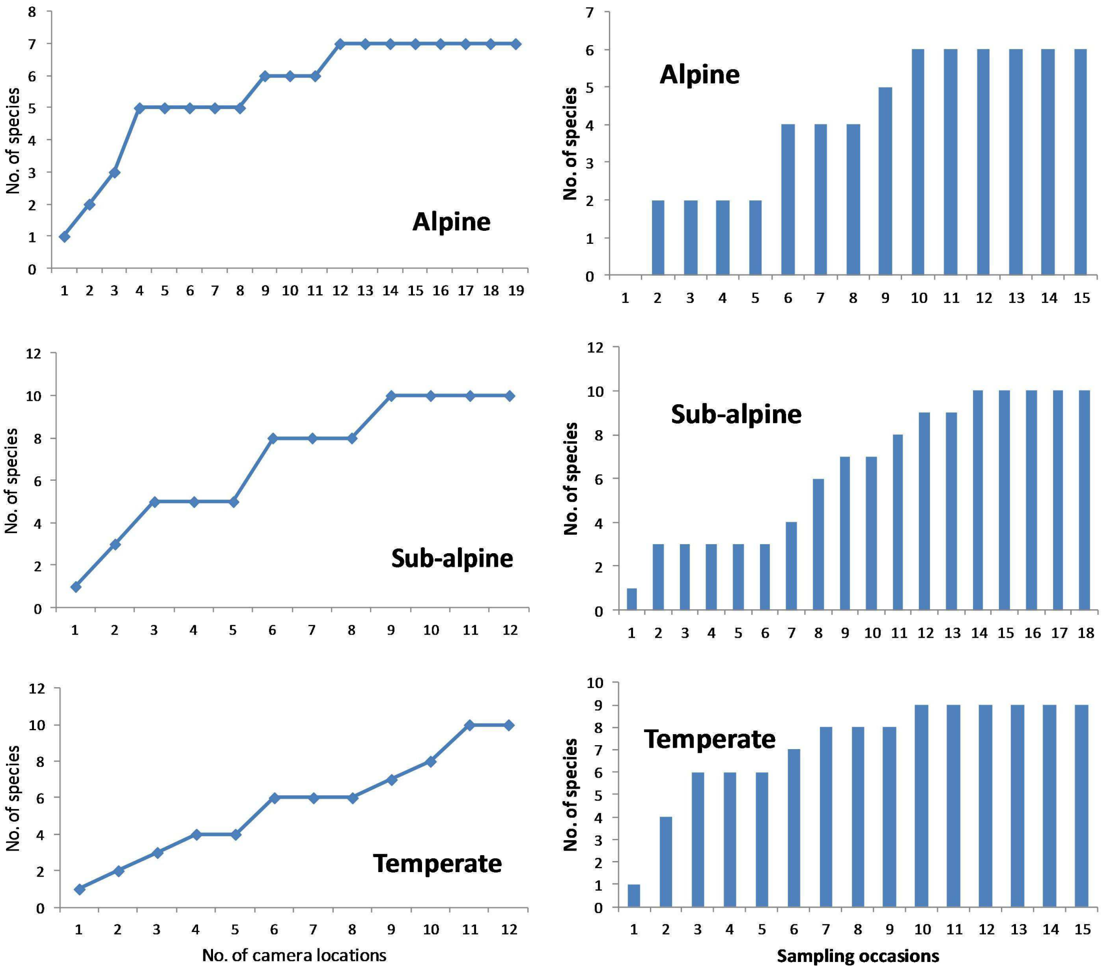


**Figure 8.2 Adequacy of camera trapping in different habitats of *Prek chu* catchment**



## 8.2.2 Population monitoring: change detection at desired power level

Interactions between sample sizes (number of counts), sampling durations (years of monitoring), frequency of surveys, and the variability in counts due to factors such as weather, season and topography can affect the adequacy of monitoring programs. If sample sizes and survey frequencies are insufficient, a monitoring programme will fail to provide the precision needed to detect population changes over time (Walsh et al. 2001). Variability of counts associated with time, climate or sampling error can hence hamper the identification of statistically significant changes in animal populations (Toms et al. 1999). Power analysis in this regard is recommended as a promising tool to address such issues. Power is a statistical measure of the risk of not detecting a trend in a population when it actually exists. Failure to identify such trends with confidence could mean that populations heading towards extinction will go unrecognized. Specifically, power is defined as (1–β) where β is the probability of wrongly accepting a null hypothesis when it is actually false (Type II errors; Gerrodette 1987; Fairweather 1991). It is also expressed as percentage and interpreted as for example, if power = 90%, it means that the statistical power of the monitoring programme is 90% to detect a population trend of a specified magnitude. In this regard, an attempt was made to apply power analysis for designing a long-term monitoring programme for few carnivore populations inhabiting the study area.

Power analysis for camera trap sampling design in order to assess its efficiency as a monitoring program was carried out in program MONITOR (Gibbs 1995) based on the estimates of abundance and variance for few carnivore species including snow leopard, leopard cat and large Indian civet. Estimates of abundance were used from the results of photographic capture-recapture analysis of the carnivore species done earlier (Chapter 4). For population monitoring of these three carnivore species using camera traps different density estimates and their variances with respect to different sampling efforts (effective camera trap days/year) were used (starting from 500 days/year to 4000 days/year in case of snow leopard, from 200 days/year to 1400 days/year in case of leopard cat and from 200 days/year to 1800 days/year for large Indian civet). Powers were estimated at 80% level [based on 500 simulations for 2 tailed tests and for significance level (α) 0.05] for 4, 5, 6, ....., 15 years for snow leopard and for 4, 5, 6, ....., 10 years for leopard cat and large Indian civet, respectively.

The results of the power analysis showed that in order to detect 3% annual decline in snow leopard population with 80% power, a sampling effort of at least 2400 effective camera days/annum is required for a minimum of 14 years. While, for detecting 5% and 10% annual declines in its population with 80% power a minimum effort of 1000 and 850 effective camera days/annum is required for a period of at least 13 and 7 consecutive years, respectively (Figure 8.3). In case of leopard cat, in order to detect 5% annual decline in population with 80% power, a sampling effort of at least 760 effective camera days/annum is required for a minimum of 9 years, while for detecting 10% annual decline in population a minimum sampling effort of 280 effective camera days/annum is required for at least 9 consecutive years (Figure 8.4). However, in order to detect 5% annual decline in large Indian civet population with

80% power, a minimum sampling effort of 520 effective camera days/annum is required for at least 9 years, and for detecting 10% annual decline in its population a sampling effort of at least 350 effective camera days/annum is required for 9 years (Figure 8.5), respectively. Moreover, it is worth mentioning that across all combinations of sampling efforts and sampling periods for snow leopard, leopard cat and large Indian civet with power level of 80% or above, effective detection of population increase can be achieved with less sampling efforts compared to that required for the detection of population decline.

**Figure 8.3 Relationship between number of years of monitoring and minimum sample size needed to achieve 80% power to detect existing changes of 3%, 5% and 10% per annum in snow leopard population in *Prek chu* catchment of Khangchendzonga BR (estimates based on 2 tailed tests, α = 0.05 and 500 simulations)**

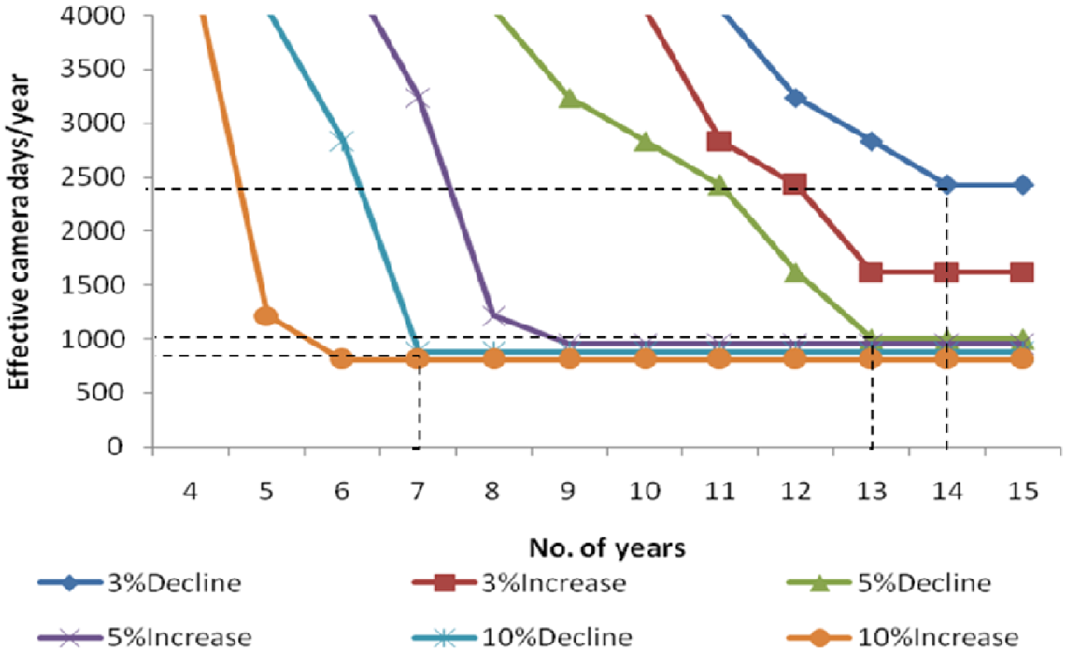


**Figure 8.4 Relationship between number of years of monitoring and minimum sample size needed to achieve 80% power to detect existing changes of 5% and 10% per annum in leopard cat population in *Prek chu* catchment of Khangchendzonga BR (estimates based on 2 tailed tests, α = 0.05 and 500 simulations)**

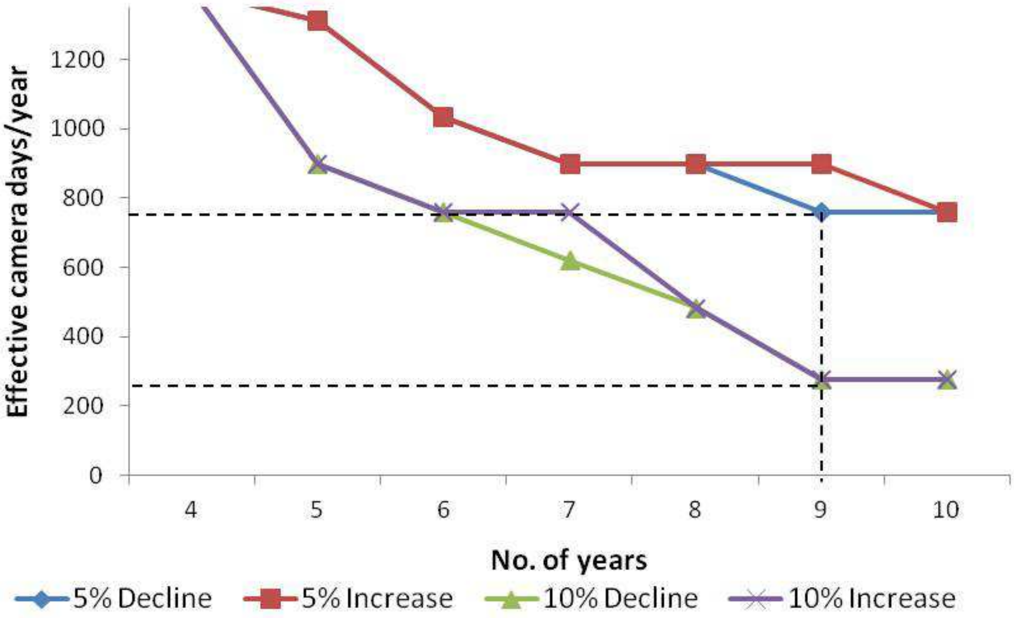
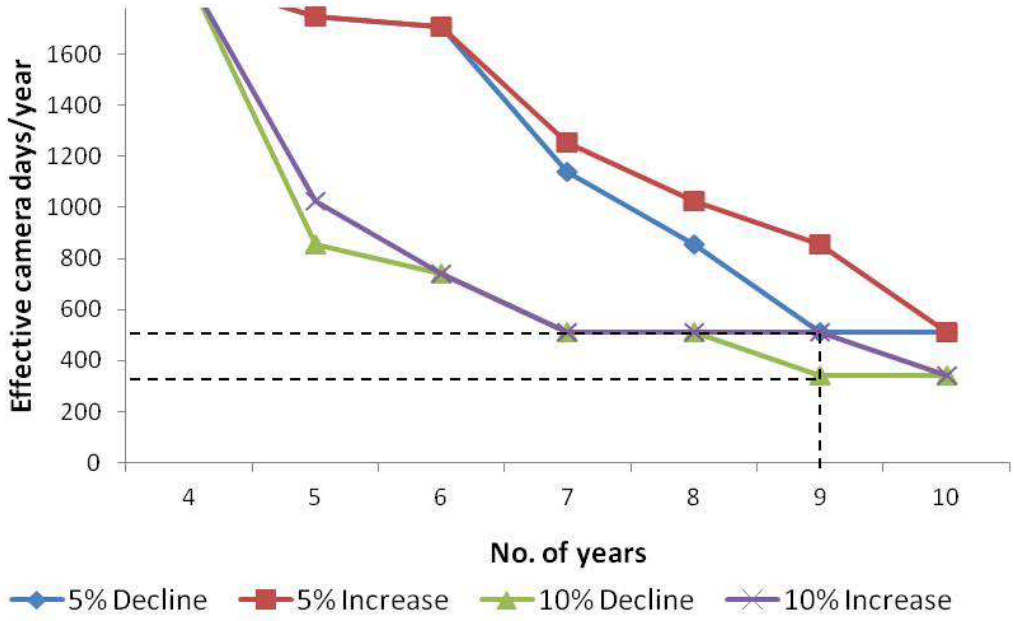


**Figure 8.5 Relationship between number of years of monitoring and minimum sample size needed to achieve 80% power to detect existing changes of 5% and 10% per annum in large Indian civet population in *Prek chu* catchment of Khangchendzonga BR (estimates based on 2 tailed tests, α = 0.05 and 500 simulations)**



### 8.2.3 Expense of monitoring carnivore populations

Since, due to the remoteness of the area and rugged terrain, all the field activities were carried out in expedition mode in the present study, which involved camping in different parts of the intensive study area (Chapters 2 & 3). Each expedition conducted every month resulted in an approximate expenditure of up to 15,000 INR including all the logistic expenses. With this background, to monitor snow leopard population a detection of 3% annual decline (if any) would be appropriate considering its global conservation status. Following the results of the power analysis, it would require an effort of 2400 effective camera days for 14 years. The most feasible way to achieve this both in terms of logistic and inference can be by deploying 20 camera trap units (Cuddeback Attack IR, Model 1156) in the alpine zone of the intensive study area for four months per annum for 14 years. This monitoring period (14 years) is conditional on the number of camera units and the effective camera days and may hence be reduced by increasing the number of camera units and the effective camera days, respectively. The cost of procuring 20 camera traps may be estimated to 300,000 INR and required number of batteries for four months of monitoring may reach up to 20,000 INR. In addition, expenditure for expedition may account to 60,000 INR (15,000 per month), costs as of 2013.

Similarly, for monitoring 5% population decline in leopard cat and large Indian civet populations, an effort of 520-760 effective trap days is required for 9 years as deduced from the power analysis. This can be achieved by deploying 12 camera units in the temperate zone of the intensive study area for two months per annum. Since, the monitoring for leopard cat and large Indian civet has to be in a different season/period than that for snow leopard because of difference in the habitats of their occurrence, this would require an additional expenditure for the purchase of batteries (6,000 INR) and the expedition cost (30,000 INR), although same camera units (deployed for snow leopard monitoring) can be used in this case. Hence, the total expenditure for battery procurement (26,000 INR) and expedition (90,000 INR) per year for monitoring these three carnivore species will account to 116,000 INR. Based on the experience of present study, camera traps in this terrain work efficiently only for two and half years if deployed for continuous monitoring. This indicates that procurement of two new sets of 20 cameras may become necessary, one after a period of five years and the other after 10 years. This accounts for an additional expenditure of 600,000 INR

(300,000 INR for each set). The monitoring cost for first nine years may amount to 1,044,000 INR (116,000 INR × 9) and in the following five years (10-14 years) may amount to 400,000 INR (80,000 INR × 5). And finally, the cost of 20 memory cards (8GB SD cards) that will last for the entire monitoring period may amount to 10,000 INR. These calculations indicate an approximate expense of 2,354,000 INR (~ 2,500,000 INR considering market fluctuations and miscellaneous expenses) for monitoring snow leopard, leopard cat and large Indian civet populations in the *Prek chu* catchment of Khangchendzonga BR.

## 8.3 IDENTIFICATION OF PRIORITY AREAS FOR HABITAT MONITORING AND CONSERVATION

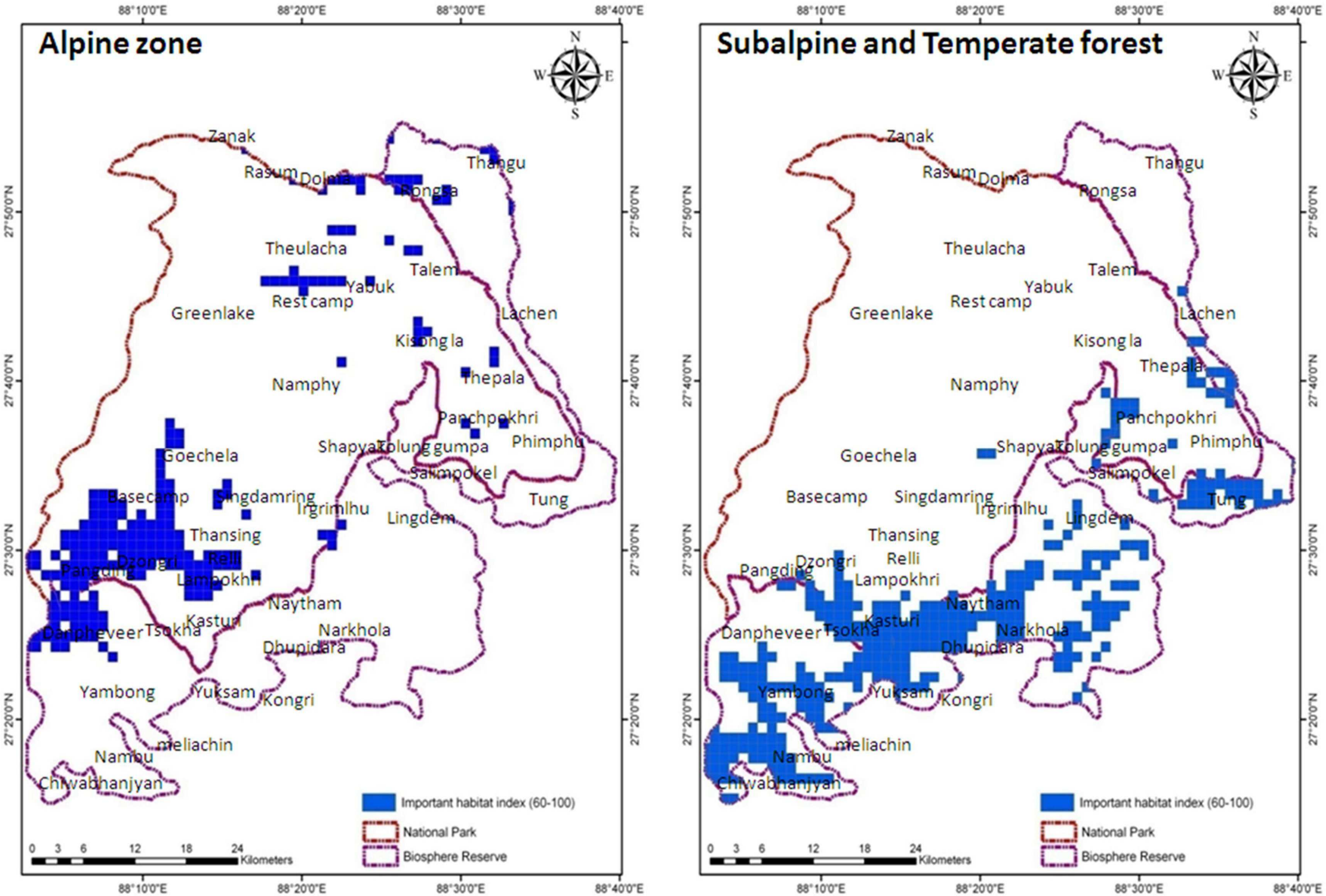
Identification of priority areas for habitat monitorings and conservation is essential for efficient management of a vast intricate landscape such as Khangchendzonga BR. In this context, findings of the habitat suitability models developed for different carnivore species in the present study (Chapter 6) were used for identifying these important areas within the Khangchendzonga landscape. The habitat suitability models had predicted several areas in the Biosphere Reserve as suitable habitats (Suitability index: 60-100) for different carnivores. Habitat suitability indices for the threatened carnivores of the study area [Snow leopard- Endangered; Asiatic black bear- Vulnerable; Golden cat- Near threatened; Large Indian civet- Near threatened (IUCN 2012)] were combined and the mean values were extracted on a 1×1 km$^2$ grid basis for the entire Khangchendzonga BR landscape for alpine and forest habitats. These mean values were further averaged for these species and multiplied by a conversion factor to derive an Important Habitat Index (from 0-100). The most suitable grids (Important Habitat Index 60-100) were identified and the nearest locations were also pointed and labeled on a map.

A schematic map representing the priority areas for monitoring and conservation of carnivores in alpine and forested habitats of the Khangchendzonga BR is presented in Figure 8.6 in a 1×1 km$^2$ grid scale. This approach can prove helpful for the managers for delineating appropriate areas for regular monitoring. In the alpine zone, the map depicted the occurrence of high Important Habitat Index (60-100) grids mainly in the southwestern part of the Biosphere Reserve. These mainly included areas such as, *Lampokhri*, *Relli*, *Gochela*, *Basecamp*, *Pangding* and *Danpheveer*. Areas adjacent to

*Gochela*, *Youngzutar*, *Relli* and *Aralungchok* have already been identified as conservation zones (Tambe 2007), but the present evaluation highlights more suitable areas for snow leopard in this area. In addition, among the trans-Himalayan habitats the areas of *Zanak*, *Rasum*, *Dolma* and *Rongsa* were depicted as the most important habitats for threatened carnivores in Northern part of Khangchendzonga BR. These areas are almost untouched by tourists due to the presence of Indo-Tibetan Border Police (ITBP) for international security reasons although semi-nomadic Tibetan pastoralists and yak herders often use these areas.

Since, the transition zone of subalpine and alpine habitats including area such as *Dzongri*, *Thansing*, upper *Yambong*, *Panchpokhri* and *Thepala* were also identified as most important for the threatened carnivores, the identified grids in *Prek chu* catchment were situated adjacent to the *Yuksam-Dzongri* trekking trail which is a known favorite destination for tourists worldwide. As a consequence, tourism related threats to the habitat such as lopping, pack animal grazing and the accumulation of non-degradable wastes which are inevitable need to be properly managed in order safeguard the habitats of these carnivores. Moreover, since the *Churong chu* watershed (*Yambong* valley trek) also entertains the engagement of local youth in eco-tourism, the magnitude of tourism and hence the affect is very less compared to that in *Prek chu*. In addition, in case of the subalpine and temperate forests most of the important habitat grids were situated along the junction of Biosphere Reserve and National Park (Figure 8.6). These grids mostly occurred in the Biosphere Reserve part connecting or buffering the villages located just outside the Khangchendzonga BR boundary and therefore warrant regular monitoring. Among the habitats at lower elevations areas such as *Nambu*, *Yuksam*, *Sachen*, *Tshoka* and *Kasturi* situated in the western part of Biosphere Reserve and at mid-elevation including areas such as *Narkhola*, lower ridges of *Panchpokhri* and *Lachen-Thepala* zone were also identified as most important for carnivore monitoring and conservation (Figure 8.6). Hence, regular monitoring of the habitats needs to be carried out mostly in the south-western part of the Biosphere Reserve, while for doing so in the northern part active participation of the villagers and yak herders is quite necessary.

Figure 8.6 Identified priority areas (1×1 km² grids) in Alpine and forested habitats of KBR for carnivore monitoring and conservation.



## 8.4 MAJOR THREATS TO CARNIVORES IN THE BIOSPHERE RESERVE

Present study did not provide any strong evidence for direct threats such as illegal hunting for pelt/parts or habitat degradation to carnivores in Khangchendzonga BR, although some conservation concerns were identified that can affect carnivore populations in future, if not managed properly. Major concerns included retaliatory killings of carnivores by villagers and yak herders in response to their livestock losses to conflict (Chapter 7). Increasing intensity of livestock and pack animal grazing inside the National Park due to growing tourism can even exacerbate this scenario in future (Plate 8.1 a, b, c). Although, awareness programs by local NGOs and schools have been successful in improving the attitude of people towards conservation to some extent at least in the *Prek chu* catchment, evidences of occasional hunting using traps and snares inside the National Park for galliformes and ungulates were also recorded (Plate 8.1d), which can reduce the prey base and hence affect the carnivores. Since, ungulates especially musk deer are known to have been facing traditional hunting pressures since long ago in the area (Subba 2000), evidences of blue sheep hunting near *Somiti* Lake using salt baited traps and snares (Plate 8.1 e, f), encounter of a caracass of a trapped serow in *Zema* catchment area (Plate 8.1g), and information on the traditional hunting of Himalayan tahr during local interviews recorded in the present study indicated the persistence of such pressures. In addition, occurrence of snares at different locations such as *Khecheopalri*, *Dubdi*, *Pakholla*, *Dzongri*, *Yambong* valley, and others in different parts of the Biosphere Reserve for hunting galliformes species also depicts the same concern.

Besides these hunting pressures, another conservation concern that can potentially lead to severe prey depletion in the area was also identified. This included the outbreak of a fatal disease particularly affecting the goral and serow populations in the temperate forests. Consequently, frequent deaths of goral and serow (n = 8) were reported during the monsoon of 2008, and subsequent reports were also obtained in 2009 and 2010 (Bhattacharya 2013). Moreover, in 2011 camera traps revealed a goral with spots on the coat and few days later its carcass (with no predation signs, but decomposing signs on coat and mouth) was encountered on the trail (Plate 8.1 h, i). Similar case of goral epidemic has also been reported from Great Himalayan National Park long back which resulted in substantial reduction in goral population in the area

(Vinod and Sathyakumar 1999). Deaths of goral and serow with similar symptoms were reported even during 2012. Presence of feral dogs inside the National Park also poses direct threat to the galliformes species hence depleting the prey base for small carnivores (Plate 8.1 j, k). In addition, illegal encroachments mostly in the lower bordering areas (Chapter 6 & 7) and anthropogenic impacts to habitats due to tourism also represent emerging concerns for carnivores and their habitats in the Biosphere Reserve.

## 8.5 MAJOR RESEARCH FINDINGS OF THE STUDY

The present study generated baseline information on the distribution, abundance, habitat use, food habits and co-existence among major carnivores and human-carnivore interactions at spatial scale in the intricate habitats of Khangchendzonga BR. It was also evident that inspite of different field methods used, camera trapping proved to be most appropriate for the monitoring of carnivores in the area. In addition, the most important habitats for the monitoring and conservation of threatened carnivores in the Khangchendzonga landscape were also identified. The specific findings for major species are presented in a tabulated form (Table 8.2).

**Plate 8.1** Major conservation concerns for carnivores in Khangchendzonga BR: (a) livestock grazing, (b & c) tourism caused pack animal grazing, (d) snares for galliformes, (e & f) salt baited trap for hunting blue sheep, (g) caracass of a trapped serow, (h & i) goral caracass with decomposing signs on coat and mouth, and (j & k) presence of feral dogs inside the National Park

**Table 8.2 Synthesis of the major findings on distribution, abundance, habitat use, food habits, habitat suitability and monitoring methods of major carnivores in Khangchendzonga Biosphere Reserve**

| **Species** | **Distribution** (watershed) | **Abundance/Relative abundance (S.E)** | **Habitat variables** (+ preferred; - avoided) | **Food habits** (major prey species) | **Activity** | **Suitable habitats/areas** | **Monitoring** |
|---|---|---|---|---|---|---|---|
| Snow leopard | *Churong, Prek, Lachen, Zema, Lhonak* | 4.77 (1.81)/100 km² [Density] | Elevation (+), Alpine (+), Vegetation cover (-) | Blue sheep, Himalayan tahr, Dzo, Pika | Bimodal (dusk & midnight) | Dzongri-Goechela-Lampokhri, Lhonak valley | Camera trapping (20 units at 120 days/year) and sign survey |
| Red fox | *Churong, Prek, Lachen, Rangyang, Rangit, Zema* | 18.21 (6)/100 km² [Density] | *Krummholtz* (+), Trekking trails (+) | Pika, Rodent, Dung beetles, Blue sheep | Nocturnal | Dzongri, Thansing, Lampokhri, Yambong, Panchpokhri, Lhonak valley | Camera trapping and sign surveys |
| YT Marten | *Churong, Prek, Rangit, Zema, Lachen* | 33.5 (7.8)/100 km² [Density] | Vegetation cover (+), Trekking trail (-), Human presence (+) | Rodent, Pika, Birds, Fruits | Diurnal | Pakholla-Tsoka, Jamling, Yambong, Kasturi, Nambu, Tung, Narkhola, Lingdem | Camera trapping, sign surveys and trail monitoring |
| Wild dog | *Churong, Prek* | 0.138 (0.06)/100 days [Photo-capture rate] | - | Serow, Himalayan tahr, Dzo | Diurnal | Sub-alpine and *Krummholtz* habitats | Camera trapping and sign surveys |
| Black bear | *Churong, Prek, Rangit, Lachen* | 0.23 (0.08)/100 days [Photo-capture rate] | Broadleaved (+), Conifer (+) | - | Arrhythmic | Yuksam-Tsoka, Kasturi, Yambong, Narkhola, Panchpokhri | Camera trapping |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| Tibetan wolf | *Lhonak* | 4.1 (2.05)/100 days [Photo-capture rate] | - | - | Nocturnal | Muguthang, 20R, Dolma, Rasum, Zanak | Camera trapping and sign surveys |
| Golden cat | *Churong, Prek* | 0.41 (0.13)/100 days [Photo-capture rate] | Vegetation cover (+) | - | Arrhythmic & crepuscular | Yuksam-Sachen, Kasturi, Yambong, Nambu | Camera trapping |
| Leopard cat | *Churong, Prek, Rangit, Lachen* | 17.52 (5.52)/100 km² [Density] | Elevation (-), Broadleaved (+), small prey (+) | Rodent, Pika, Birds | Nocturnal | Yuksam-Sachen-Tsoka, Yambong, Nambu, Tung, Sakmpokel | Camera trapping (12 units at 60 days/year) and sign survey |
| Masked palm civet | *Churong, Prek, Rangit* | 14.03 (6.52)/100 km² [Density] | Elevation (-) | - | Nocturnal | Yuksam-Sachen, Yambong, Nambu, Narkhola, Lingdem | Camera trapping |
| Large Indian civet | *Churong, Prek, Rangit, Lachen* | 10.67 (3.71)/100 km² [Density] | Elevation (-), Broadleaved (+), Human presence (-) | - | Nocturnal | Yuksam-Sachen, Meliachin, Dhupidara, Narkhola, Naytham, Tung | Camera trapping (12 units at 60 days/year) |
| Stoat | *Churong, Prek, Rangit, Rangyang, Lachen* | 10.26 (4.52)/100 km² [Density] | Elevation (+), Vegetation cover (+), Conifer (+) | - | Diurnal | Phedang, Phedi, Aralungchok, Shapyakolung, Phimphu, Thepala | Camera trapping |
| Jackal | *Churong, Prek* | 4.42 (2.2)/100 km [sign encounter rate] | - | Rodent, birds, goral, fruits | - | Lower temperate and sub-tropical habitats | Camera trapping and sign surveys |

## 8.6 LIMITATIONS OF THE STUDY

Rugged terrain and inaccessibility were the main limiting factors for sign surveys, trail monitoring and camera deployment. Washing away of bridges and frequent landslides during monsoon also impeded field research to some extent. Moreover, dense forest cover and illusive nature of the species resulted in low sightings of carnivore and ungulate, thereby hindering the assessment of absolute abundance for most of the species. Similar pattern was observed in terms of their photo-capture rates. Less number of camera trap units for monitoring (maximum of 27) was also a limiting factor in context of the vast Khangchendzonga landscape. Harsh weather conditions and incessant rains resulted in quick decay of carnivore scat samples and caused problems in their identification. This lead to the non-identification of a number of samples (n = 132). Uneven weather, extreme cold conditions, low moving clouds, movement of vegetation due to wind, and movement of tourists, local people, pack animals and feral Yaks resulted in continuous camera exposure thereby reducing the effective trap days and also causing camera malfunctioning and hence loss of data. Moreover, two units of camera traps were stolen from the field and could not be recovered. In addition, continued delay in the issuance of permits for North Sikkim by the Department of Forests, Environment and Wildlife Management limited intensive surveys and camera trapping in the trans-Himalayan habitats of KBR.

## 8.7 CONCLUSION AND RECOMMENDATIONS

This study provides the first scientific information on the ecology of carnivores in the Khangchendzonga BR and is of high significance to managers for efficient conservation and management of these ecologically important species and their habitats in such intricate habitats where accessibility and data collection is a limitation. Regular monitoring and strict vigil of the most suitable habitats (particularly the south-western and northern parts of the Biosphere Reserve) predicted during this study is necessary to safeguard these areas against any such anthropogenic pressures. Proper demarcation of the private forests is required in order to reduce human-carnivore conflict along the fringe villages. Also, efforts should be made to minimize crop cultivation close to the forests and emphasis should be given on the development of effective bio-fences to avoid crop damage and livestock depredation by wild animals. Regular awareness campaigns should also be conducted in the villages to improve their understanding of the importance of local wildlife and Protected Areas. Tourism pressure also needs to be regulated properly so as to minimize its impact on wildlife habitats. Special attention towards the prevention/eradication of the disease of wild ungulates is required to reduce the possibility of substantial decline in major prey species of carnivores in the forested habitats. There is a need for strong cooperation among the Forest Department, local NGOs and the stakeholders from villages in order to achieve successful conservation of carnivores and their habitats in different watersheds of the Biosphere Reserve and reduce any hunting pressures on their prey base.

Future research should focus on intensive surveys and camera trapping in other watersheds of the Biosphere Reserve, particularly in the trans-Himalayan habitats of *Lhonak* catchment to reveal the complete status of carnivores and other mammals of that area. This can also help in validation of the habitat suitability models prepared during the present study. An assessment of small mammal prey populations and the use of molecular technology for scat identification should also be a priority in future studies. Moreover, the response of these carnivores to anthropogenic factors such as disturbances due to eco-tourism is yet to be studied and should be undertaken. A detailed study involving actual assessment of damage in the field should be attempted to back up and corroborate the findings of the present study. Future studies on these aspects can definitely fill the gap in research on carnivores and help managers to develop an efficient management plan for their conservation in the intricate landscape of Khangchendzonga.

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# Precarious status of the Endangered dhole *Cuon alpinus* in the high elevation Eastern Himalayan habitats of Khangchendzonga Biosphere Reserve, Sikkim, India

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**Abstract** During 2008–2010 we investigated the ecology of the Endangered dhole or wild dog *Cuon alpinus* in Khangchendzonga Biosphere Reserve in the Eastern Himalaya in Sikkim, India. We conducted camera trapping (n = 71 sites, 6,278 camera-days) and sign surveying along trails (n = 24; 629.43 km of effort) to assess the relative abundance, distribution and activity pattern of the dhole. Morphological characteristics evident in the 61 camera-trap photographs indicate that the dhole population in the Reserve may be the rare and genetically distinct subspecies *C. alpinus primaevus*. We detected dholes over a wide elevation range (2,501–4,100 m) that encompassed the upper temperate, subalpine, and alpine scrub zones. Dholes were diurnal, with peak activity at 08.00–10.00. Analysis of 41 scats indicated a diet comprising mainly mountain ungulates, rodents and pikas *Ochotona* sp. Although the frequency of occurrence of rodents was highest (32%) in the scats, 98.7% of the total biomass consumed was of mountain ungulates. Historical reports (1888–1894) indicated that the dhole was formerly abundant in Sikkim but was hunted to meet the high demand for its alleged medicinal properties. With no information on the status of the dhole in Sikkim for over a century, our study suggests that the species is now rare in the Reserve. To aid the conservation of the dhole and its main ungulate prey species extensive research and monitoring are required in the Reserve and elsewhere in the Eastern Himalaya.

**Keywords** *Cuon alpinus primaevus*, dhole, diet, Eastern Himalaya, India, mountain ungulates, Sikkim, subalpine forest

## Introduction

The Asiatic wild dog or dhole *Cuon alpinus* is a primarily pack-living, strictly terrestrial large canid, typically weighing 12–20 kg, usually with a reddish or brown coat and a darker bushy tail (Johnsingh, 1985; Durbin et al., 2004). It is the only species in the genus *Cuon,* with 11 subspecies (Durbin et al., 2004; Iyengar et al., 2005), of which five are reported from south Asia (Johnsingh, 1985; Durbin et al., 2004). In some areas it was customarily and incorrectly categorized as vermin and assumed to reduce natural populations of wild ungulates and livestock. Organized reward hunting and poisoning exterminated the species over much of its range (Davidar, 1975). Recently, habitat degradation, extinction of prey populations, and conflicts with and persecution by humans has led to a marked reduction in the dhole's range (Iyengar et al., 2005), and it is categorized as Endangered on the IUCN Red List (Durbin et al., 2008).

Little is known of the dhole in north-east India and north of the river Ganges, and only a few short-term surveys have been carried out (Johnsingh, 1985; Stewart, 1993, 1994). In this area the species has been recorded in the states of Arunachal Pradesh, Assam, Meghalaya, West Bengal (Durbin et al., 2008) and Sikkim (Sathyakumar et al., 2011). Dhole–livestock conflict has been studied in Bhutan (Johnsingh et al., 2007), and the species' ecology is being studied in Pakke Tiger Reserve (Gopi et al., 2010). The presence of two subspecies of dhole was reported in Sikkim in the 19th century (Gammie, 1894). Jerdon (1874) and Blanford (1891) mentioned one species of dhole *Cuon dukhunensis* in Sikkim but that local people believed there were two, differing in colour, size and habits. These were a large type, brownish, with a black muzzle, occurring only in pairs or in groups of three or four, and a small type, reddish in colour, occurring in packs of 10–12 and hunting prey such as wild pig *Sus scrofa*, barking deer *Muntiacus muntjak*, goat *Capra aegagrus* and other livestock. The large type was believed to have medicinal properties and was hunted to meet this demand.

Here we present the first information in c. 100 years on the dhole in the high elevation habitats of Khangchendzonga Biosphere Reserve. Based on camera-trap photographs we present information on the species' morphology, activity patterns and distribution, and using scat analysis we investigate the dhole's diet. We also highlight the need for further research on the species and for appropriate protection to safeguard this particular subspecies.

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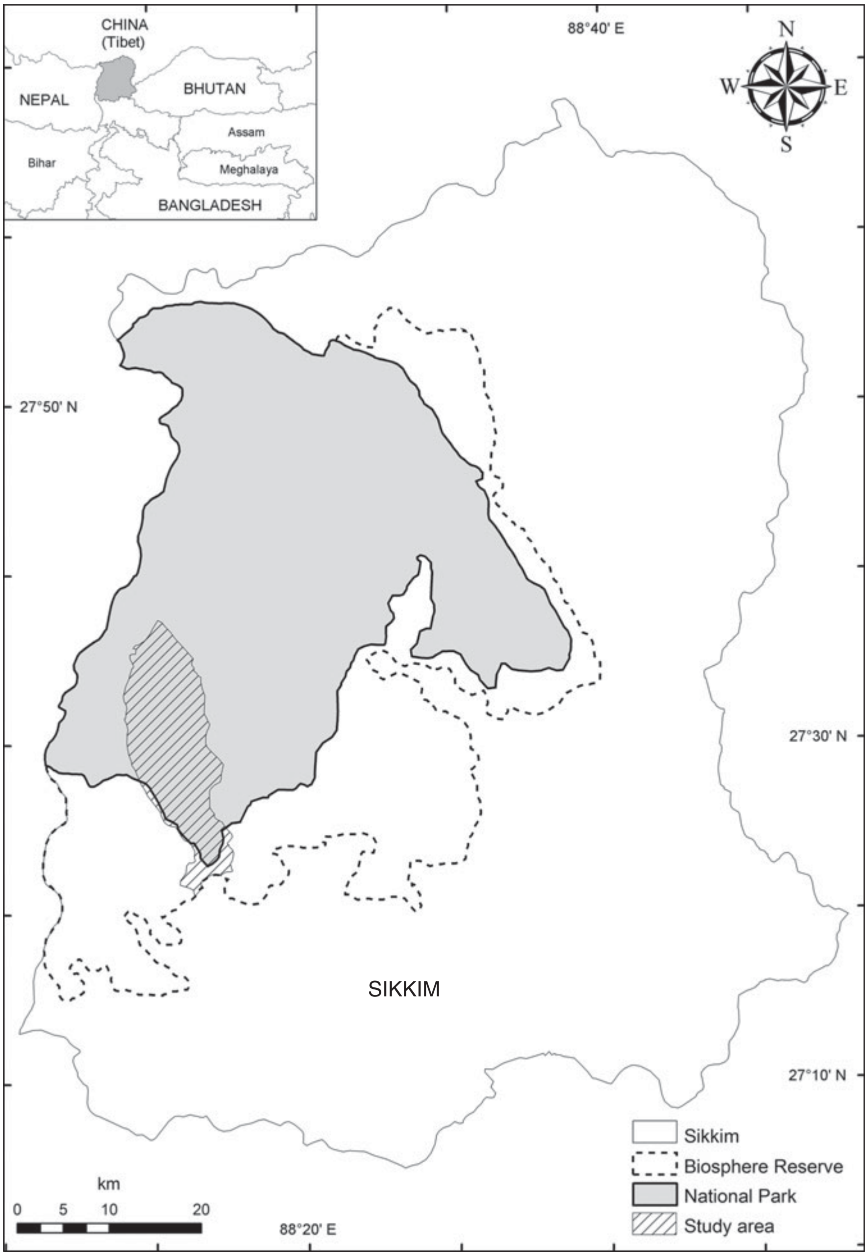

FIG. 1 Location of Khangchendzonga Biosphere Reserve in Sikkim, showing the Khangchendzonga National Park and the catchment of Prek Chu, in which we conducted this study. The shaded area on the inset indicates the location of Sikkim in north-east India.

## Study area

Sikkim, a small mountainous state of India, is the western-most part of the Eastern Himalayan biodiversity hotspot (Myers et al., 2000). This study was carried out in Khangchendzonga National Park and Biosphere Reserve (Fig. 1) from February 2008 to August 2010. The Reserve encompasses temperate, subalpine and alpine habitats (1,220–5,000 m) and rocky slopes, glacial moraines and permafrost zones (>5,000–8,586 m). There are seven water catchments, of which Prek Chu was selected as the study area as it contains all of the habitat types occurring in the Reserve (Sathyakumar et al., 2011). The 182 km$^2$ of Prek Chu lies over 1,220 to 6,691 m and receives a total annual rainfall of 1,750–2,250 mm (Tambe, 2007). The major habitats and their percentage of the total area (Fig. 2) are mixed sub-tropical and mixed temperate (17%), subalpine and krummholtz (36%), alpine pastures (5%), rock and snow cover (41%) and water bodies (1%).

# Methodology

## Camera trapping

Prek Chu was divided into 4 km$^2$ blocks, using *ArcGIS v. 9.0* (ESRI, Redlands, USA), and categorized into three survey zones according to habitat: temperate (1,200–3,000 m), sub-alpine (3,000–4,000 m) and alpine (> 4,000 m). There were seven, 12 and five blocks in the temperate, subalpine and

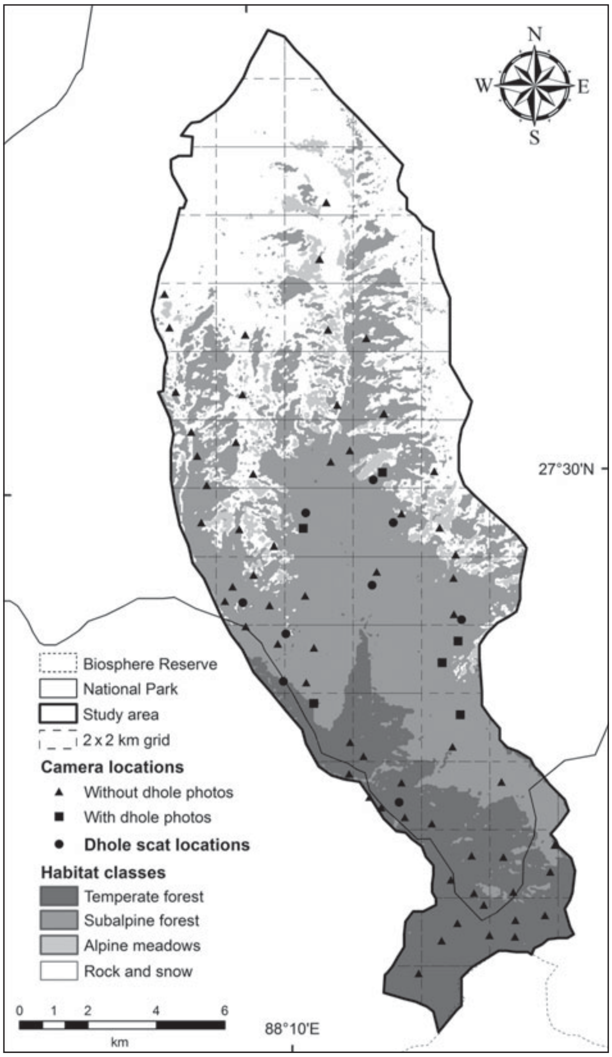

FIG. 2 The study area, Prek Chu catchment (Fig. 1), showing the locations of camera traps in a 2 × 2 km grid and the locations where dhole *Cuon alpinus* scats were found.

alpine zones, respectively. The number of camera traps deployed corresponded to the area of the three zones and their accessibility, with a total of 27 camera traps deployed at 71 sites (24, 27 and 20 in the temperate, subalpine and alpine zones, respectively; Fig. 2) for at least 30 days. We used four models of infrared-triggered camera units: two DeerCam (Non Typical, Inc., Park Falls, USA), two Wildview (Stealth Cam, LLC, Grand Prairie, USA), 18 Stealth Cam (Stealth Cam, LLC, Grand Prairie, USA) and five Moultrie (Moultrie Feeders, Alabaster, USA). As we were focusing on a rare species over a large area our strategy was to survey more trap sites less intensively rather than fewer trap sites more intensively (Mackenzie & Royle, 2005). All cameras were set with a 1 minute delay between photographs and 24 h operation, and in four-picture burst mode.

Camera units were attached to trees 15–30 cm above the ground and 3–5 m from a trail or location where animal movement could be expected. Geographical coordinates, elevation and forest type were recorded at each camera-trap location. Camera traps were monitored at least twice per month, when batteries and memory cards were changed.

## Sign surveys

Repeated surveys for dhole signs were carried out along 24 trails (with lengths of 1.5–7 km). We walked six trails (a total of 67 trail walks, with a total of 201.15 km of effort) in the temperate zone, 12 trails (108 walks, 299.11 km) in the subalpine zone and six trails (48 walks, 129.17 km) in the alpine zone.

## Scat identification, collection and analysis

Identification of dhole scat was straightforward as dholes tend to defecate in the middle of trails, unlike large felids, which defecate along the edges of trails, and because the members of a pack of dholes defecate in the same spot (Kumaraguru et al., 2011). Dhole scats were found in clusters, exposed on soil (Plate 1a), and could be distinguished easily from the scats of felids, which were stickier and deposited on grass (Johnsingh, 1983). We collected only those scats that were deposited in the middle of trails and were either in an aggregation or clustered over 1–1.5 m along a trail. All scats of each aggregation were collected as a group. We excluded all scats found singly, to avoid confusion with scats of red fox *Vulpes vulpes* in the subalpine zone and jackal *Canis aureus* in the temperate zone. Scats were collected during the sign surveys. The location, date and other data such as the occurrence of tracks around scats were also recorded.

A reference key was developed for the identification of prey species on the basis of hair structure and morphology. All scat samples were sun-dried in the field and later oven-dried at 60°C (Sankar & Johnsingh, 2002) for 48 h and preserved in tagged paper bags for later analysis. Each scat was broken, soaked and washed with running water, using fine-mesh (1.0 and 0.5 mm) filters (Juarez & Filho, 2002), to separate prey remains such as hair, bones, hooves, teeth, feathers, nails and other undigested remains. Washed samples of hairs from the scats were dehydrated in absolute alcohol for 12 hours and then placed in xylene for 15–20 minutes (Koppikar & Sabnis, 1976). Hairs were then slide-mounted and examined at 10× and 40× magnification. At least 20 hairs were examined from each scat (Mukherjee et al., 1994) and prey species were identified by comparison with the reference collection, using features such as colour, length, thickness, characteristic medullar configurations (Koppikar & Sabnis, 1976; Reynolds & Aebischer, 1991; Mukherjee et al., 1994) and cortex-to-medulla ratio.

PLATE 1 (a) Dhole *Cuon alpinus* scats found in a cluster in the middle of a trail in temperate forest in the Prek Chu catchment in Khangchendzonga Biosphere Reserve (Figs 1–2). (b) Dhole pack, with one subadult (in the foreground). (c) Adult individual in subalpine–alpine edge forest. (d) Adult in subalpine forest.

## Status, distribution and activity

To estimate the relative abundance of the dhole we calculated the photographic and photo-capture rates. The former is the number of camera days required to capture one independent photographic event of a dhole summed across all camera traps in the study (Carbone et al., 2001). The latter is the number of independent photographic events of dholes divided by the number of trap-days per site. As the camera traps were set in four-picture burst mode we looked at the revisit rate of the known individuals/groups and found that 54 minutes was the maximum time between revisits in a day. Based on this, we considered any capture $> 54$ minutes apart as an independent event for calculating photographic and photo-capture rates. Mean photo-capture rates were calculated for all of Prek Chu and for the three survey zones separately (Carbone et al., 2001). The number of camera-trap days was calculated from the date of deployment to the date of retrieval (if the memory card was not full) or to the date of the final photograph. Time of day of the photographs was used to determine the daily activity pattern (Pei, 1998) of the dhole. A daily activity index was calculated as the number of photographs within a 2-hour duration × 100/total number of photographs. We also calculated the encounter rate of signs per km in each survey zone and used the Mann–Whitney $U$ test to test for differences between zones.

## Food habit analysis

Frequency of occurrence ($F$) of mammalian prey in scats was calculated as $F = n/N$, where $n$ is the number of dhole scats having that particular species and $N$ is the total number of dhole scats analysed (Karanth & Sunquist, 1995). We subjected the results of the scat analysis to resampling using the bootstrap method, with *Simstat* (Peladeau, 2000). Subsamples equalling the original sample size of scats were iterated 10,000 times, to generate means and bias-corrected 95% confidence intervals for percentage frequency of prey items in scats (Mukherjee et al., 2004). The frequency of occurrence of prey species in the scats was converted to relative biomass (Karanth & Sunquist, 1995) as this provides the best approximation of actual diet (Klare et al., 2011). We used $Y = 0.035+0.020X$ (Floyd et al. 1978), where $Y$ is kg of prey consumed per scat and $X$ is the mean

TABLE 1 Details of photographs of the dhole *Cuon alpinus* captured at six camera-trap sites in the Prek Chu watershed of Khangchendzonga Biosphere Reserve (Figs 1–2) during 2009–2010.

| Site | Altitude (m) | Habitat | Total trap-days | No. of photographs | Group size | Group composition |
|---|---|---|---|---|---|---|
| Ghunsa | 3,100 | Subalpine | 281 | 5 | 4 | Adult |
| Chongrigang | 4,100 | Alpine | 178 | 12 | 4 | Adult + 1 subadult |
| | | | | 3 | 1 | Subadult |
| Kockchurong | 3,700 | Subalpine | 71 | 5 | 5 | Adult |
| Phedi top | 3,900 | Subalpine | 135 | 3 | 1 | Adult |
| Phedhi | 3,700 | Subalpine | 271 | 5 | 1 | Adult |
| | | | | 5 | 1 | Adult |
| Kasturi top | 3,300 | Subalpine | 293 | 10 | 1 | Adult |
| | | | | 13 | 1 | Adult |

weight of an individual of a particular prey type (Ackerman et al., 1984). Multiplying each *Y* by the number of scats found to contain a particular prey species gave the relative weight of each prey type consumed. These values were used to estimate the percentage biomass contribution of each prey species in the dhole's diet (Klare et al., 2011).

## Results

A sampling effort of 6,278 camera-days across 71 sample sites was achieved in the three survey zones (1,407, 3,061 and 1,810 camera-days in the temperate, subalpine and alpine zones, respectively), resulting in 4,517 photographs (2,668 of wild animals and 1,849 of domestic animals and people). We recorded 42 mammal species (Sathyakumar et al. 2011), of which three were canids (n = 518 photographs): the red fox (n = 456; 152 photo-capture events), dhole (n = 61; nine photo-capture events) and golden jackal (n = 1; one photo-capture event). Of the 61 dhole photographs, obtained at six camera-trap sites, 22 contained more than one individual (Table 1). Based on a wide-angle photograph depicting one stationary animal investigating the camera trap and four resting on the ground, pack size was probably no more than five. Subadult individuals (Plate 1b) were photo-captured twice, once with a pack and once alone (Table 1). Thirty-nine photographs of the dhole were of a single individual (Table 1). All dhole photo-captures were during daytime, with a peak (45% of the captures) at 08.00–10.00.

All the adult dholes photographed had a long reddish brown coat and a dark bushy tail, and the pinnae were white inside and reddish brown on the outer side. The muzzle was blackish and relatively short and slightly convex in profile. The nose was black and short white whiskers were present over the white upper and lower lips (Plate 1c). The dorsal and lateral pelage was reddish brown with a yellowish tinge and darker at the neck (Plate 1d). The fore neck, chest and underside were white, with long and dense white fur on the upper chest and throat. Long fur, either white or reddish brown, was present on each paw of adult individuals (Plate 1c,d).

The photographic rate (i.e. the minimum number of days required to capture one dhole photographic event) was 541 days. The overall mean photo-capture rate was 0.12 ± SE 0.05 per 100 days and was highest (0.26 ± SE 0.10 per 100 days) in the subalpine zone. Dhole photographs were obtained at five sites (seven photo-capture events) in subalpine forests at elevations of 3,100–3,900 m and at one site (two photo-capture events) at 4,100 m in the alpine zone (Table 1). The sites were dominated by *Abies–Betula–Rhododendron* forest or dwarf *Rhododendron* and *Juniperus* above 3,700 m.

Whilst walking trails we did not encounter dholes and the only signs we found were 41 scats. The encounter rate of scats was significantly higher ($P = 0.03$, Mann–Whitney *U* test) in the subalpine (0.21 ± SE 0.1 scats $km^{-1}$, 31 scats) than in the temperate zone (0.02 ± SE 0.01 scats $km^{-1}$, 10 scats). We did not detect any evidence of canids in the lower temperate zone (1,850–2,500 m), although we made frequent sightings of jackal below 1,850 m.

Only 29.3% of dhole scats in a group had more than one prey item. Undigested matter was present in the following decreasing order: hair (100%, present in all scats), bone (80.5%), nail (24.4%), teeth (24.4%), hoof (7.3%) and undigested remains of grass (7.3%). Unidentified rodents were the most frequently found prey item, followed by the serow *Nemorhaedus sumatraensis*, Himalayan tahr *Hemitragus jemlahicus*, goral *Nemorhaedus goral* and pika *Ochotona* sp., unidentified remains and vegetable matter (Table 2). Serow was the largest prey (mean adult body weight 91–100 kg) consumed, followed by the Himalayan tahr (80 kg) and goral (30 kg; Prater, 1971). The estimated relative biomass of prey and the relative number of individual prey are also presented in Table 2. The biomass estimates indicate that although dholes killed more pikas and rodents, ungulate prey species (serow, Himalayan tahr and goral) contributed 98.7% of the total biomass consumed.

## Discussion

The reddish-brown long coat and long white or reddish brown hairs on the paws indicate that the dhole population

TABLE 2 Diet of the dhole in Khangchendzonga Biosphere Reserve (Fig. 1) as indicated by analysis of 41 scats, with the mean percentage frequency of prey items (with 95% confidence intervals, CI, from bootstrapping), estimated % relative biomass of prey, and relative number of individual prey consumed (see text for further details).

| Prey species | Mean % frequency (95% CI) | % relative biomass | Relative no. of individuals consumed |
|---|---|---|---|
| Unidentified rodents | 32 (14–41) | 1.18 | 57.89 |
| Serow *Nemorhaedus sumatraensis* | 27 (15–41) | 52.05 | 5.23 |
| Himalayan tahr *Hemitragus jemlahicus* | 22 (10–34) | 34.21 | 4.29 |
| Goral *Nemorhaedus goral* | 20 (7–32) | 11.81 | 3.95 |
| Pika *Ochotona* sp. | 20 (10–34) | 0.75 | 28.64 |
| Unidentified | 10 (2–19) | | |
| Vegetable matter | 7 (0–15) | | |

TABLE 3 Number of days required to obtain one camera-trap photograph of the dhole (the photographic rate) in Khangchendzonga Biosphere Reserve (Fig. 1) and in seven other protected areas in south Asia (Datta et al., 2008).

| Protected area | No. of trap days | Photographic rate (days) | Reference |
|---|---|---|---|
| Khangchendzonga Biosphere Reserve, India | 6,278 | 541 | This study (2008–2010) |
| Namdapha National Park, north-east India | 1,537 | 0* | Datta et al. (2008) |
| Hukawng Valley Tiger Reserve, north Myanmar | 8,836 | 4,418 | Lynam (2003) |
| Hkakaborazi National Park, north Myanmar | 1,238 | 29 | Rao et al. (2005) |
| Taman Negara National Park, Peninsular Malaysia | 14,054 | 878 | Kawanishi & Sunquist (2004) |
| Nam Et-Phou Louey National Protected Area, Lao | 3,588 | 359 | Johnson et al. (2006) |
| Bukit Barisan Selatan National Park, Indonesia | 24,045 | 6,024 | O'Brien et al. (2003) |
| Phu Kheio Wildlife Sanctuary, Thailand | 1,224 | 111 | Grassman (2003) |

*Datta et al. (2008) did not secure any photographs of the dhole but confirmed its presence through indirect evidence, and Mishra et al., (2006) reported skins of dholes from six valleys at high altitudes in Arunachal Pradesh.

in Khangchendzonga Biosphere Reserve could be the subspecies *C. alpinus primaevus* (Durbin et al., 2004), which has also been reported at lower and middle altitudes in Bhutan (Johnsingh et al., 2007). The presence of thick white fur on the chest and throat also distinguishes it from the dhole of southern India *C. alpinus dukhunensis*, which is known to be genetically distinct from *C. alpinus primaevus* (Iyengar et al., 2005). Iyengar et al. (2005) did not find evidence for the recognition of all 11 subspecies of dhole and showed there is admixture of subspecies over a vast area, and therefore genetic analysis is required to determine the true identity of the dhole of Khangchendzonga Biosphere Reserve.

The photographic rate for the dhole in Khangchendzonga Biosphere Reserve (541 days) lies within the range (29–6,024 days) of that in seven other camera-trap studies in south Asia (Table 3). Trail surveys in Pakke Tiger Reserve in Arunachal Pradesh, India, resulted in a low encounter rate of dhole signs (0.26 km$^{-1}$; Gopi et al., 2010), similar to our encounter rate (0.21 km$^{-1}$) in Khangchendzonga Biosphere Reserve. Comparison of our results with those of other studies is only tentative, however, as there were differences in study design and camera-trapping effort. More robust methods such as occupancy-based abundance estimation or non-invasive DNA-based capture–recapture studies are required to elucidate the status of the dhole in Khangchendzonga Biosphere Reserve and elsewhere.

Factors that may influence habitat selection by dholes include the availability of medium to large ungulate prey species, water, the presence of other large carnivore species, human population density and suitability of breeding sites (Durbin et al., 2004). In the subalpine forests, serow and goral were relatively abundant and Himalayan tahr was also present (Bhattacharya et al., 2010). No other large or medium-sized carnivore species has been reported from these subalpine forests except the Asiatic golden cat *Pardofelis temminckii* (Bashir et al., 2011), which is reported to be sympatric with the dhole in Peninsular Malaysia (Kawanishi & Sunquist, 2008). Presence of large and medium ungulate prey, absence of large carnivores such as the common leopard *Panthera pardus* and relatively less anthropogenic pressure may be the reasons for the presence of the dhole primarily in subalpine forests in our study area.

Other studies have also indicated that the dhole is diurnal and obtains most of its dietary biomass from ungulate prey (Johnsingh, 1992; Karanth & Sunquist, 1995; Venkataraman et al., 1995; Kawanishi & Sunquist, 2008; Borah et al., 2009; Kumaraguru et al., 2011; Kamler et al., 2012). The presence of pikas and rodent species in the diet of the dhole could be because of its ability to flush out and hunt smaller and cryptic prey species (Venkataraman, 1996; Kumaraguru et al., 2011). Although too small for the pack as a whole, such small prey is sufficient for an individual dhole (Kumaraguru et al., 2011). Dholes also occasionally consume grass and other vegetation (Cohen et al., 1978; Johnsingh, 1983; Durbin et al., 2004), which explains the presence of undigested vegetation remains in some scats.

Our results indicate that probably only one subspecies of dhole, *C. alpinus primaevus*, is present in the subalpine forests of Prek Chu. Iyengar et al., (2005) suggested that this subspecies, reported to be 'very rare' in a survey in the early 1980s (Johnsingh, 1985), and distinct from *C. alpinus dukhunensis*, should be accorded a high priority for conservation action. Other catchments in Khangchendzonga Biosphere Reserve need to be surveyed, with intensive camera trapping, for the dhole. In Bhutan dholes have caused livestock loss because villagers allowed their livestock to graze unsupervised (Johnsingh et al., 2007). We recommend that livestock depredation cases should be monitored in Khangchendzonga Biosphere Reserve as unsupervised grazing is also practised in this area. Our findings have been shared with the relevant State and Central authorities, for enhancement of the conservation management of the dhole in this region, and a further project has now been initiated to monitor the dhole population in Prek Chu and other catchments in Khangchendzonga Biosphere Reserve.

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