Response of tiger population to habitat, wild ungulate prey and human disturbance in Rajaji National Park, Uttarakhand, India

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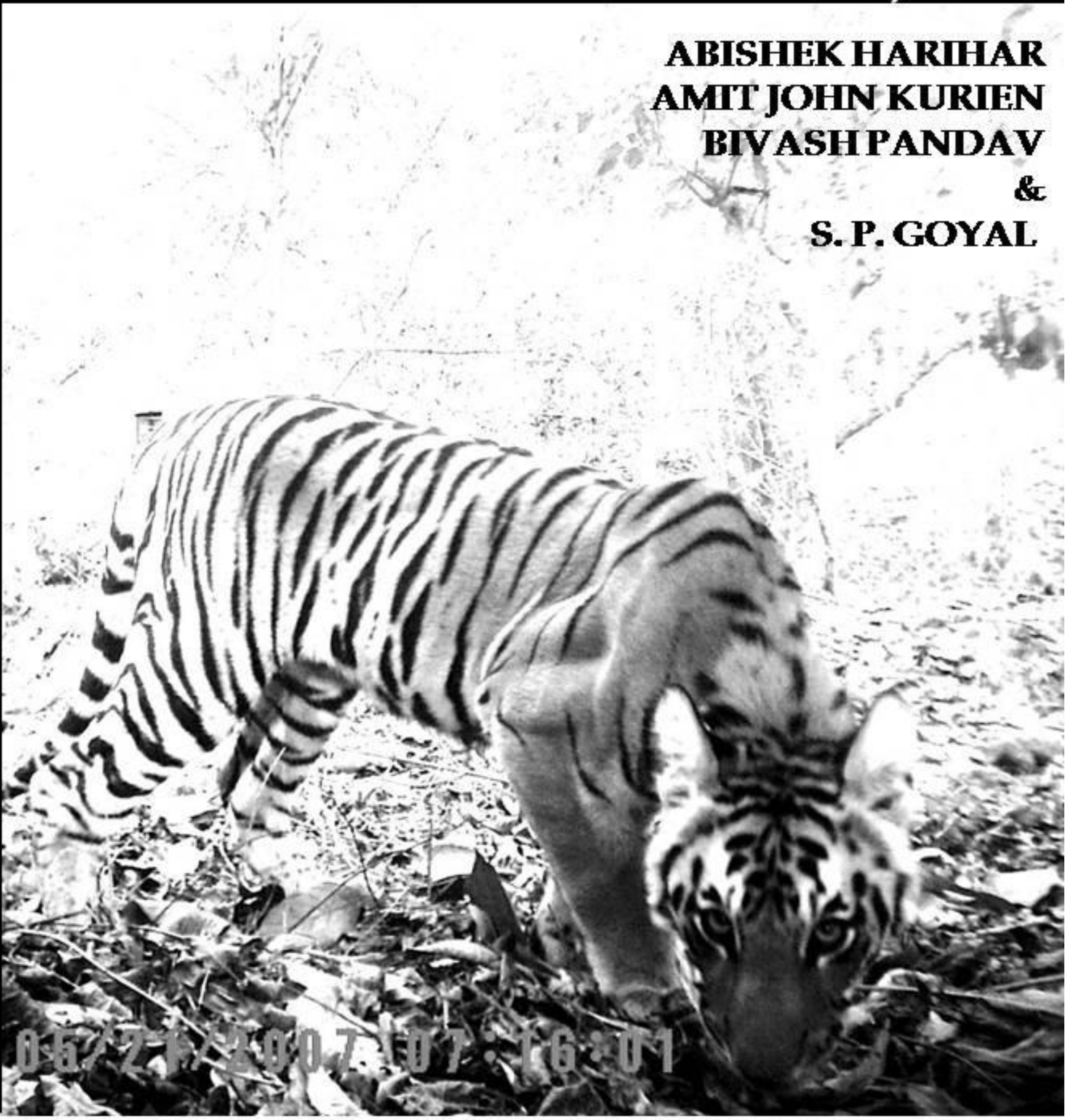

Final Technical Report 2007

भारतीय वन्यजीव संस्थान
Wildlife Institute of India

THE NATIONAL FISH AND WILDLIFE FOUNDATION
SAVE THE TIGER FUND

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We once again express our sincere gratitude to the Government of Uttaranchal for executing this massive relocation program and providing us the privilege to document the changes after relocation of gujjars from the Park.

Project outline

This project funded by Save the Tiger Fund grant no. 2005-0013-027, aimed to assess the status of habitat, wild ungulate prey species and tiger from an area recently made free of human settlements by the Uttarakhand Forest Department since 2003. This project supported two masters' students of the Wildlife Institute of India. Abishek Harihar who submitted a dissertation titled "**Population, Food Habits and Prey Densities of Tiger in Chilla Range, Rajaji National Park, Uttaranchal, India**" and Amit John Kurien who submitted a dissertation titled "**Response of tiger (*Panthera tigris*), prey species and their habitat in relation to human disturbance in and around Chilla Range of Rajaji National Park, Uttaranchal**". The project also supported Abishek Harihar during a period of one year during which he was employed as a Juniour Research Fellow. This project report summerises the work carried out over three years (2004-2007) in six research papers.

The first paper entitled "**Varying Human Disturbance and its effect on tiger, prey and habitat – case study from Chilla, RNP, India**" investigates the patterns of distribution of tiger and its principal prey species in response to human disturbance in the recently evacuated Chilla Range and the adjoining Shyampur Range of the Haridwar Forest Division. The results show a distinction in tiger and prey usage of the two forest areas indicating that human mediated disturbance deters habitat use by these species. The Second paper entitled "**Impact of human disturbance on woody species of four different forest communities in the dry tropical forest of Siwaliks, North India**" studied the effect of human disturbance on four vegetation types of the Chilla Range of Rajaji National Park and showed that miscellaneous forests considered to be suitable ungulate habitat was the most affected by human mediated disturbance. The third paper entitled "**Density of tiger and prey species in Chilla range, Rajaji National Park, Uttarakhand, India**" monitors the population of tiger and its prey in Chilla Range of Rajaji National Park for a period of three years since 2004. The study indicates that both prey and tiger populations are on the rise following the relocation of human settlements. With photographic evidences of breeding tigers being obtained during the course of this study, it is clear that minimizing of anthropogenic pressures could aid recover populations of both wild ungulate prey and tigers. The fourth paper entitled "**Status of tiger and its prey species in Rajaji National Park**" assess the staus of tigers and their prey in the entire area of the Rajaji National Park following the resettlements of many more *Gujjar* settlements. The results indicate that though prey densities are high in the western part of Rajaji National Park (across the ganges) the expected recovery of tiger populations that was noticed along the east bank of Rajaji National Park was not notoiced, therefore bringing to notice that active management interventions would have to be taken to restore the Chilla-Mothichur corridor thereby facilitating tigers from east Rajaji National Park to disperse. The Fifth paper entitled "**Separation between two sympatric carnivores in Chilla range of Rajaji National Park**" assesses the trophic niche overlap between tigers and leopards in the Chilla Range of Rajaji National Park over three years (2004-2007). The study indicates increased competition and possible competitive exculsionn of leopards owing to the increase in the population of tigers. And finally the last paper entitled "**Estimating population size of tigers using camera trap based capture-recapture sampling: minimizing closure violation and improving estimate precision**" outlines methodological insights gained during the camera trapping excersise. The results indicate that in order to minimize closure violation and improve estimate precision trap area would have to be increased and that trap density would have to be maintained high.

Varying Human Disturbance and its effect on tiger, prey and habitat – case study from Chilla, RNP, India

Amit Kurien, S.P. Goyal and Bivash Pandav

***Abstract*:** We studied the effect of presence and release of human disturbance on tiger and prey occurrence and on habitat structure in Chilla Range of Rajaji National Park from November 2004 to April 2005 following a relocation of Gujjar communities from the forest. After thirteen months of remaining undisturbed, we found that tiger occurrence is more compared to the neighboring areas facing human disturbance. Prey occurrence data clearly showed that presence of human settlements clearly deterred use by wild ungulates like Sambar and Chital – the key prey species of tiger. With the absence of lopping and other human activities, the habitat seems to be showing the signs of a successional phase. Shrub density was higher in the human evacuated area and the exotic weed density density was higher in the human occupied area. Distance to hamlets proved a difficult variable to measure lopping effect as a result of very high human density within the forested landscape. PCA results indicated that between the two neighboring areas although a statistically significant difference in disturbance variables was not seen, there exists a sign of a gradient of better habitat structure towards the human evacuated area.

Varying Human Disturbance and its effect on tiger, prey and habitat – case study from Chilla, RNP, India

Amit Kurien, S.P. Goyal and Bivash Pandav

1. Introduction

Conservation of endangered large vertebrates in fragmented landscapes has become a central issue for conservation biologists (Wikramanayake *et al.* 2004). It is by now known that loss and fragmentation of habitat is a major threat to the continued survival of many such species. The fundamental cause of virtually all recent and ongoing declines of mammalian species is the growth of human populations (Cardillo *et al.* 2004). Among the most threatened of mammals are species in the order carnivora (Woodroffe 2000). Among them, most of the larger carnivores, the top predators, are more prone to local extinction as a result of hunting, habitat loss and fragmentation.

The predator in human dominated landscapes

The tiger (*Panthera tigris*) is the largest predator in Asia (Weber & Rabinowitz 1996). As a result of territorial nature and wide movement patterns it requires large areas of suitable habitat to survive in their natural state (Seidensticker 1976). Studies from Chitwan and Nagarhole reveal that the size of breeding female Ranges varied from 13-30 sq. km and that of males from 40-100 sq. km (Smith 1993, Karanth & Sunquist 2000). Recent studies (Eisenberg & Seidensticker 1976, Karanth & Sunquist 1992, Karanth & Nichols 1998, Carbone & Gittleman 2002) suggest that abundance of tigers is directly related to densities of ungulate prey species. The requirement of adequate prey densities and water are the most important aspects of an individual's survival. At a broader scale of a species, vast natural habitats play, perhaps the most vital role apart from the other two factors above mentioned. Tiger behaviour requires adult animals to disperse into different areas (Smith 1993), and this demands good patches of adjoining forests. Studies have indicated that tigers are known to avoid human disturbance (Johnsingh *et al.* 2004). Today, many of the remaining tiger populations are confined to small and isolated forest patches where stochastic events and continuing human impacts are likely to cause local extinction (Smith 1993, Johnsingh & Negi 2003, Kawanishi & Sunquist 2004). Poaching is another reason for the decreasing number of tigers in many of its habitats. In many places the otherwise contiguous tiger populations often span many adjoining jurisdictional units (Smith *et al.* 1998), which make conservation initiatives erratic in implementation. Human presence between natural patches in turn disallows the process of genetic exchange between populations that is vital in maintaining the genetic fitness of the population. Patchy distributions of habitats, as a result of human intrusion are known to alter tiger usage of the area, its breeding habits and general social behaviour. This in turn drives the point home that even with increasing human population, the requirement of large landscapes for an absolute conservation approach of this large cat still remains an imperative.

The prey and habitat in human dominated landscape

Of the many herbivores of the tropical forests, the main tiger prey species are the medium to large sized ungulates (Seidensticker 1976). Many studies, from Schaller

(1967) to Karanth & Sunquist (1995) have mentioned that tiger prefer medium (31-175 kg) to large (>176 kg) sized animals as prey. Medium sized prey would include chital and wild pig, while large prey would mean sambar, nilgai and gaur. In the forests of northern India, the largest available prey animal is sambar, and is a generalist grazer/brower (Schaller 1976, Johnsingh 1983). It is known to have a significant dependency on shrubs and water. Chital on the other hand is known to have significant dependency on surface water and partial cover (Schaller 1967, Johnsingh 1983). They prefer open grasslands during winters and more forested patches during summer when grass is low in abundance (Schaller 1967, Bhat 1993). Nilgai, wild pig and barking deer are also available prey species for tiger but it is found that chital and sambar contribute to about 55-65% of a tiger's diet in most places (Schaller 1967, Johnsingh 1983, Karanth & Sunquist 1995).

The support of high biomass animals require the presence of natural forests with adequate palatable plant species that are available for grazing and browsing. India, has only 5% of the land area under forest cover and these requirements are a problematic proposition to achieve, given that India simultaneously holds the largest domestic herbivore population in the world of 285 million (FAO 2005) a good number of them being close to the protected areas. One of the greatest threats to wild herbivores in a human dominated landscape like India, especially prey species of the tiger like sambar, chital and nilgai is the threat of competition from domestic livestock and induced effects of overgrazing and habitat degradation. According to Mishra *et al.* (2002), when livestock species are introduced into a co-evolved assemblage of native wild herbivore species, they may compete with and even exclude, native wild herbivores. Earlier work on resource competition (Fritz *et al.* 1996) show that domestic livestock, as a result of being derived from wild herbivore ancestry, have similar patterns of resource requirements and utilization. In the case of chital, there is a case for reason – as ruminant grazers it is closer to livestock in terms of its digestive capabilities. Sambar (a forest ruminant browser) is not related to cattle in that respect. However, in terms of losing ground as far as habitat is concerned, sambar face competition directly from human beings and their resource use patterns. Sambar are known to prefer forests with thick under cover (Seidensticker 1976) and most human activities like tree felling for fuel and timber, cattle grazing, conversion of land to agriculture etc end up reducing shrub cover drastically.

Habitat loss therefore is the predominant threat type for the great majority of mammals (Cardillo *et al*. 2004). Competition from livestock and human settlements with unsustainable land use patterns and poaching are major causes for worry in most protected areas of India. Accounting for reserve size and productivity, Rivard *et al*. (2000) showed an adverse effect of local towns on local extinction of species. Pollution of water sources that emerge from the forest is also an impact that cannot be avoided in the presence of abundant human and livestock presence. For prey species and tiger alike, utilizable clean water is a principal resource that they depend on.

Given a scenario of this magnitude, many initiatives by various organizations starting from the World Bank's Eco development initiatives to NGO's have been trying to provide alternatives to reliance on the natural resources of protected areas. In this regard, the Forest Departments have tried their best with the help of other organizations to provide an alternative livelihood to people living within and close to forest patches that are used by tiger and prey species. In a country like India, such efforts are very difficult to undertake, and not many have truly succeeded. Some efforts that are worth the mention are the relocation programme that took place from Kanha National Park, Ranthambore National park and relocation of the pastoralist settlements of the Maldharis from the Gir Lion sanctuary in Gujarat (Khan 1995).

Chilla and the *Gujjars*

The north Indian state of Uttaranchal includes the large landscape of the Siwaliks which is considered to be the north western limit of this conservation flagship. The protected area of Rajaji National park (RNP) and Corbett Tiger Reserve are among the strongholds of good populations of this species. Many areas of the Siwaliks include a group of nomadic pastoralist community called "Gujjars". As on today, some Gujjars are nomadic pastoralists coming down towards the Siwaliks during the winter and returning to higher altitudes during summer. Most other Gujjars have resided permanently within the forests of Siwaliks, many of them within the Rajaji National Park. Their presence in the areas of RNP is evident from atleast 1939 in Coomb's plan (1939-48), and presence of a grazing working circle is mentioned as early as 1924 in Champion's plan (1924-38) (Kumar 1995). For a fairly long time permits were issued to the families within the park to cut grass and lop branches for leaves to provide fodder for their livestock holdings (mostly buffaloes). Over time the actual number of families living in the forest far exceeded the permissible limit. As their buffalo holdings increased over time (from ca. 4000 to ca. 20,000), the requirements for fodder increased proportionately. A cumulative result was the excess lopping of many tree species, overgrazing and the overall consequence was habitat degradation of these forests, which is the repository of the northwestern most population of the tiger and Asian elephant (Johnsingh and Negi 2003).

During October 2003 a process of relocating the human (Gujjar community) settlements took place from Chilla Range, following the notification of the earlier Rajaji Wildlife sanctuary to a National park. The study site of Chilla Range was under the heavy influence of Gujjar communities, who used to feed their livestock, largely buffaloes (*Bubalus bubalis*) by lopping trees and grazing them in the forest. The intensive lopping, firewood extraction and grazing in most of these parts had led to lack of sustainable regeneration and proliferation of weeds (Edgaonkar 1995). As a pilot programme, 74 Gujjar settlements (193 families) were shifted out of ca. 150 sq km of Chilla Range. Void of human settlements, there was very healthy resurgence of ground vegetation indicating promising signs of recovery with respect to utilization of the area by wildlife. Natural history records and a monitoring effort indicated an increased use of the area by wildlife. For the purpose of conservation it is therefore important to undertake research activities to see and understand how a predator, prey and habitat respond in the event of no human disturbance as opposed to areas facing disturbance. Taking into account the seemingly improving status of the protected area and the relieving pressure from human habitation within this Range, this study assesses the effect of human disturbance on the habitat and the relative use of tiger and its prey species in Chilla Range.

2. Study area

The intensive study area was the Chilla Range ($29^o54'$ N to $30^o15'$ N and $77^o50'$ E to $78^o16'$ E) ca. 150 sq. km and Shyampur Range (ca. 90 sq. km) adjoining it, falling southwards. Broadly, both these areas fall in the Siwalik Range. The Siwaliks run parallel to the outer Himalayas from Jammu to Assam and is considered one of the most threatened and fragile ecosystems in the Indian subcontinent and is home to the north western most population of the tiger and the elephant. Chilla falls on the east bank of the Ganges and provides a connectivity between the western regions of the RNP (west of Ganges) and the eastern portions of the Siwalik landscape that encompass the regions of the Corbett National Park. These areas make up the Rajaji Corbett Tiger Conservation Unit (RCTCU) (Johnsingh & Negi 2003). The RCTCU is one among the eleven Level-1 Tiger conservation units identified in the Indian subcontinent for the long-term conservation of the tiger (Wikramanayake *et al*. 1999).

Topography and Vegetation

The layout of the study area is characterized by high undulating topography consisting of hills with V-shaped valleys intersected by dry riverbeds called 'raus'. Broadly, the forests of this region can be categorized as Northern Indian Moist Deciduous Forest and Northern Tropical Dry Deciduous Forest (Champion and Seth 1968). Apart from these, plantations of *Tectona grandis* and *Haplophragma adenophyllum* are found here. The vegetation of this region is a result of various physical and climatic factors including anthropogenic pressures such as human habitation, extensive grazing, lopping and cutting of trees for firewood and fodder (Edgaonkar 1995).

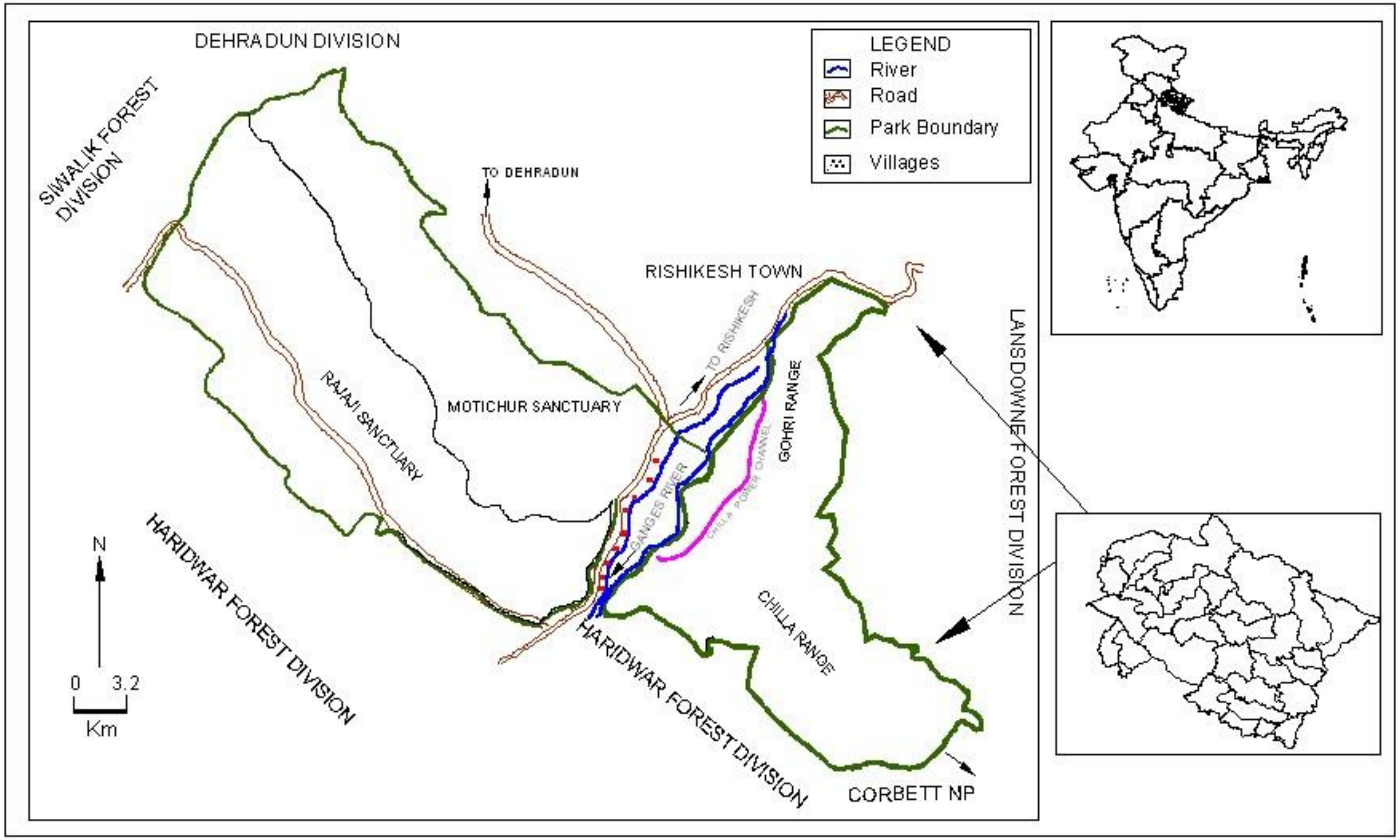


Figure 3.1. Map of Rajaji National Park, showing the study area

The region experiences three seasons clearly – winter (November - February), summer (March - June) and rainy season (July - September). The study period stretched from November to April, through winter and summer. Winter is characteristic of cold nights with dew and frost. As summer approaches, frost vanish, dew decreases. There is usually slight rainfall in between the two seasons. Winter mean monthly temperatures are around 22^{o}c and summer it is as high as 45^{o}C.

Human influences

Over the years anthropogenic activities have increased with the growing human population and its demand for more forestland for agriculture and various developmental projects. This has broken the continuity of the previously large forest patch. The presence of the large town of Haridwar and Rishikesh has been a tremendous pressure to the forests, apart from the Gujjars themselves. After the relocation, Chilla Range presently does not hold human settlements. Shyampur Range still holds numerous Gujjar families.

Fauna

Although bearing the brunt of many questionable activities, the Chilla Range still boasts of a high diversity of vertebrates. Apart from the large diversity of avifauna (including the rare White backed vulture and Long billed vulture), the other mammals found here are leopards, sloth bears, Asiatic black bear, hyenas, sambar, chital, barking deer, goral, wild pig, hanuman langur, rhesus macaque, palm civet, jungle cat, porcupine, small Indian civet, Indian pangolin and Himalayan masked palm civet.

Among them the black bear and the Himalayan masked palm civet are new records for Chilla Range.

3. Methods

Estimation of tiger and prey occurrence

Based on an intensive survey undertaken for assessing tiger status in the Siwalik ecosystem Johnsingh *et al.* (2004) suggested that streambed transects (rau walks) are ideal for monitoring tiger pugmark occurrence. Streambeds or raus were therefore chosen as the line of walk for tiger sign surveys. Most of them inadvertently criss cross the pattern of habitats in the Siwaliks. The objective of the streambed transects/walks was to generate an index that can represent the relative occurrence of tiger and its prey species. Indirect evidences in the form of pugmark, scats and scrapes give a good indication of the relative use by tiger of the study area (Karanth & Nichols 2002). Since the parameter of interest was animal occurrence, all signs (tracks and pellets) were recorded. It is, by now, widely known that tigers use natural trails for their movement (Smith *et al.* 1989), and many studies use track plots for identifying tiger occurrence. A pointer to tiger occurrence can be reported in the form of an index that can indicate a rate of occurrence. Caughley (1977) defined an index as "a measurable correlative of abundance". For the purpose of identifying occurrence patterns in the area by tigers the index essentially needed to give only the information regarding the rate of tiger signs (pugmark/ scat/ scrape) encounters.

Eight transects of 5 km length were chosen in the study area. Five were in Chilla Range (within park boundary) and three fell outside in Shyampur Range, outside Chilla but adjoining the park boundary. Transects were along dry riverbeds or *raus* with loose to compact sand. It ensured that tracks and signs were well noticeable. Eight transects (*raus*) well spread out in the study area were selected. For the analysis, transect "segments" comprised the basic sampling unit. There were 20 segments of 250m length in each transect. Pugmark encounter rate is measured as the total number of pugmark encounters upon the total number of segments.

The exercise was carried out with a team of 2-4 persons and on an average took 3-4 hours per transect (1.25-1.5 km/hr). Standardizing the search effort with two persons proved to be difficult as the width of the riverbeds ranged between 5-50m across. A better design was thought to be to use up to four persons and search in strips along the length of the riverbed. As mentioned earlier the feature of interest was relative occurrence in the area, and not abundance. The pugmarks of tigers, whenever seen were identified and the continuity of the pugmark trails was used to identify them as separate encounters. Care was taken to avoid double counts of trails. Tracks of ungulates were recorded in each segment as present (<5 track trails), common (>5 track trails) and abundant (>10 track trails). The transects were walked in the mornings during winter and summer as the riverbeds were moist and ideal for locating pugmarks. Pellet groups of prey species and cattle and elephant dung depositions were also recorded. A total of 78.75 km was surveyed during the entire period - 38.75 km in winter and 40 km in summer.

Vegetation quantification

The human (*Gujjar)* settlements in Chilla and Shyampur Ranges are known as *dera* and each of its location was the feature of interest for the study. The design included ten points for vegetation quantification on two radiating 2 km transects moving away from the *dera* (the origin) in the north and south direction. Ten *deras* were chosen for this purpose. Five of them were unoccupied and within the evacuated area of Chilla Range and the other five were occupied and were in Shyampur Range, hereon simply called occupied and evacuated areas. With the help of a hip chain and compass vegetation plots were laid at every 200m of the 2 km transect. The GPS location of each plot was recorded.

Sampling was carried out at every 200 m interval along the transect and data on tree density, lopping and canopy cover was recorded from a total of 25 314 m^2 plots (radius 10m) in all four forest types. A tree was defined as any plant with GBH (girth at breast height) $\geq$ 20 cm. Any tree splitting below 1.3m height was considered as two individuals, as the structural contribution of such splitting boles was equivalent to two nearly placed trees. A branch was defined as a part of the tree that either had substantial leaf growth and/ or big enough for firewood use. We enumerated lopping by counting each slash mark on the trees that corresponded to the lopping of branches. The proportion of lopped or cut stems per sampling unit has often been used as a quantitative measure of human induced disturbance by various authors (Barve et al. 2005, Ganeshaiah et al. 1998, Pandey & Shukla 2003, Shaanker et al. 2004), some others use it as an index of disturbance. It has proved to be a fairly accurate measure that reflects the relative anthropogenic pressure at a site.

Other structural attributes were inventoried from nested plots within the 10m radius plots. Shrub and weed density and sapling density were estimated from 5m radius plots. Small individuals of tree species were also included as shrub, as they have a structural contribution to shrub cover and density. Structure of ground vegetation, including tree seedlings was noted from within 1m diameter plots placed at the four cardinal directions of the 10m radius plots. The category of ground vegetation at each 0.1m was recorded as herb cover, litter or barren ground following the point intercept method (Mueller-Dombois & Ellenberg 1974). Canopy readings were taken using a densiometer from the four cardinal directions of the 10m plots. From each plot, the approximate distance to the nearest water source and the nearest *dera* was noted.

Pellet/ dung count of prey species

For recording wild herbivore pellets and livestock dung, counts were carried out using 30X2m plots. Strip transects are widely used for this purpose (Plumptre & Harris 1995). Pellet/dung counts are a widely used and good indicator of habitat use as it tallies the usage over a longer period of time (Neff 1968, Cairns & Telfer 1980, Campbell *et al.* 2004). Campbell *et al.* (2004) refers to this procedure as faecal standing crop (FSC). Each plot was laid with the help of a 30m tape, stretched out at two sample points, one being the centre of the vegetation plot and the other 30m ahead in the direction of the transect. In general, longer rectangular plots are preferred over square ones (Neff 1968) while doing a FSC. Following this, two observers searched a 1m distance on either side of the tape. Only pellet groups with at least 15 pellets were considered as a single count. Identification of pellets was done instantly by observing the shape and size. Faecal matter of looser consistency was recorded if more than half the mass was within the plots.

Data analysis

Basic statistical analysis was done using graphs and scatterplots of MS Excel and SPSS 8.0 (Norussis 1990). Correlation matrices were made to see associations between variables. Further analysis was done using linear regressions to uncover relationships between correlated variables. Principle component analysis (PCA) was performed on the variables collected for the habitat analysis.

4. Results

Tiger occurrence patterns in Evacuated and Occupied area

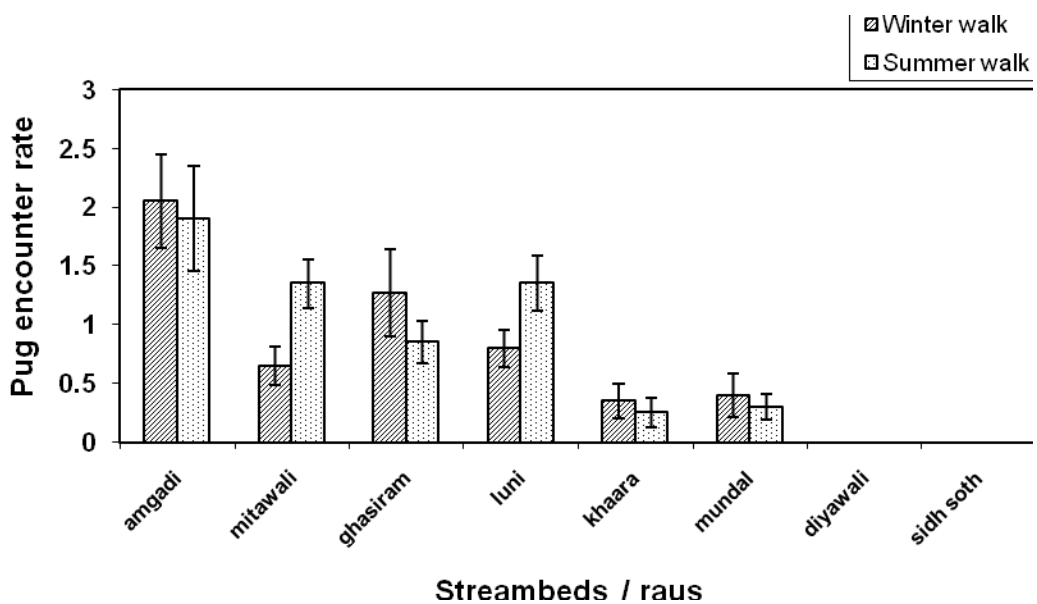


Figure 5.1. Pugmark encounter rate in each streambed during winter and summer

Among the eight streambeds, Amgadi, Ghasiram, Luni and Mitawali (all in evacuated area, within the park) had the maximum pug encounters during both winter and summer (Fig. 5.1). Khaara being more towards the park boundary did not have many encounters. Of the four above mentioned streambeds, Amgadi, Mitawali and Luni were the most promising as far as encounters were concerned. Amgadi had similar encounters in winter and summer, and stream character wise is closer to Mitawali. The results also show that the two 5 km stretches of Diyawali and Sidh soth (occupied areas) did not yield even a single tiger pugmark encounter. The sample size for the streambed transects analysis was low (only one per season) to run any robust analysis for inferences that could be predictive. Incessant rainfall defeated the effort to replicate the transect walks. As a result, only two walks were possible - one during winter, and the other during summer. All analysis is based on this dataset.

Among the streambeds that are within Chilla Range, Amgadi and Luni are the places that had similar proportions of water occurrences throughout the study period (Fig. 5.2). Mundal had a much higher proportion during the winter, but dried up in summer, as it was a fairly plain streambed. So was the case with Mitawali as some parts of the transect were in rocky, dry and plain terrain. Ghasiram and Khaara also fall in similar category of terrain, but some reasonable rainfall made sure that most segments of the stream had running water. Sidh soth also experienced similar summer rainfall. Diyawali was the only stream which had similar proportion of segments with water throughout the study period.

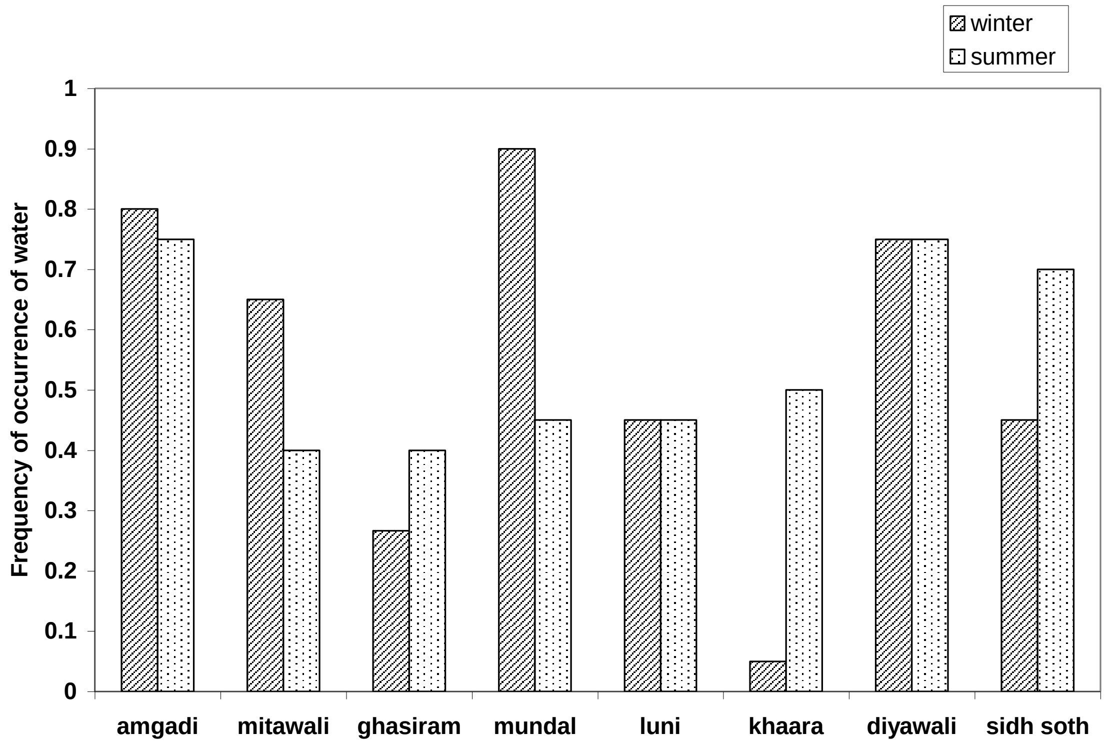


Figure 5.2. Proportion of segments with water presence in all the eight transects. The summer experienced unsually high rainfall in some areas

Table 5.1. Mean encounter rates (no. of pugmarks/ segments) of tiger in evacuated and occupied areas during the two seasons

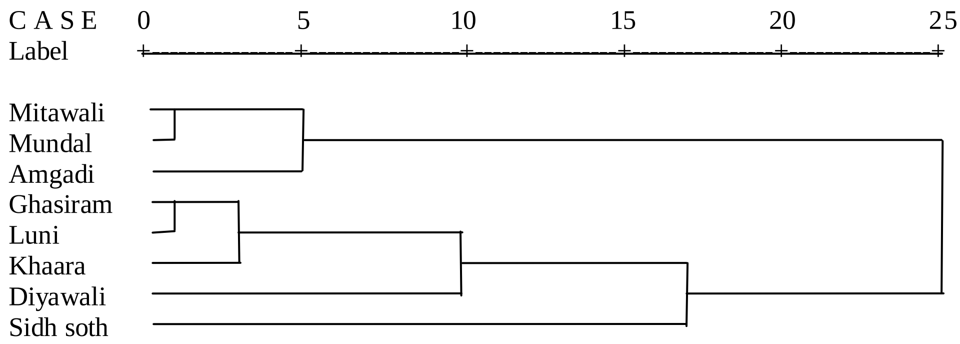
	Mean Encounter rate, ER (no./ segment)	
Sites	Winter (ER ± SE)	Summer (ER ± SE)
Evacuated	1.03 ± 0.29	1.15 ± 0.27
Occupied	0.12 ± 0.11	0.08 ± 0.08

A mean value of the winter and summer data of the pugmark encounter rates of tiger from both the evacuated and the occupied areas reveal a drastic skew of occurrence patterns towards the evacuated area. As mentioned earlier, this could be a result of the availability patterns of water and prey abundance combined with reduced disturbance. Shade cover in the nearby areas, or just a result of its sociobiological behaviour could also mediate it. What is of importance is that areas without human disturbance are clearly being preferred over areas with human disturbance.

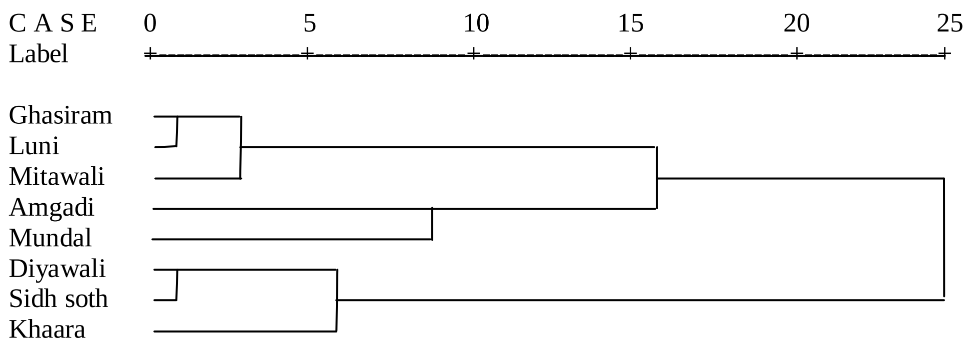
A dendrogram was prepared using hierarchical cluster analysis for the winter; summer and combined track count data (Fig. 5.3). The scaling used was squared Euclidean distance. The length of the straight line indicates dissimilarity between objects. The principle is that the first cluster is formed between objects with least dissimilarity (more similar objects). Further clusters take place which are relatively more dissimilar.

A linkage at fairly close cluster scale distance was observed between Amgadi, Mitawali and Mundal and Ghasiram, Luni and Khaara for the winter track count data indicating the within group similarity. Diyawali and Sidh soth (occupied area) stand out separate and similar. The summer count data speaks of two different clusters - Ghasiram, Luni and Mitawali and Diyawali, Sidh soth and Khaara. Amgadi and Mundal get linked to the former cluster at a further distance. However the analysis on the combined dataset (Fig. 5.3) reveals the clear clustering of three groups, Ghasiram-Luni-Khaara (rau's along park periphery), Diyawali-Sidh soth (occupied area) and Mitawali-Mundal-Amgadi (evacuated area), which is more similar to the winter count clusters.

a)



b)



c)

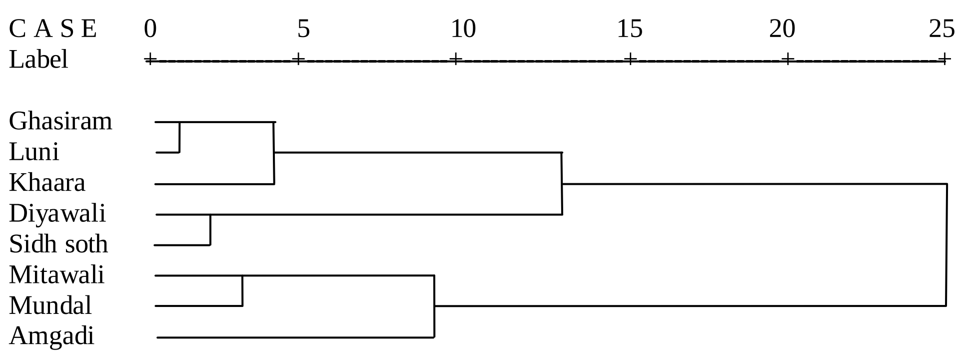


Figure 5.3. Dendrogram using average linkage (between groups) showing similarity between streambeds with respect to animal movement during a) winter, b) summer and c) combined dataset.

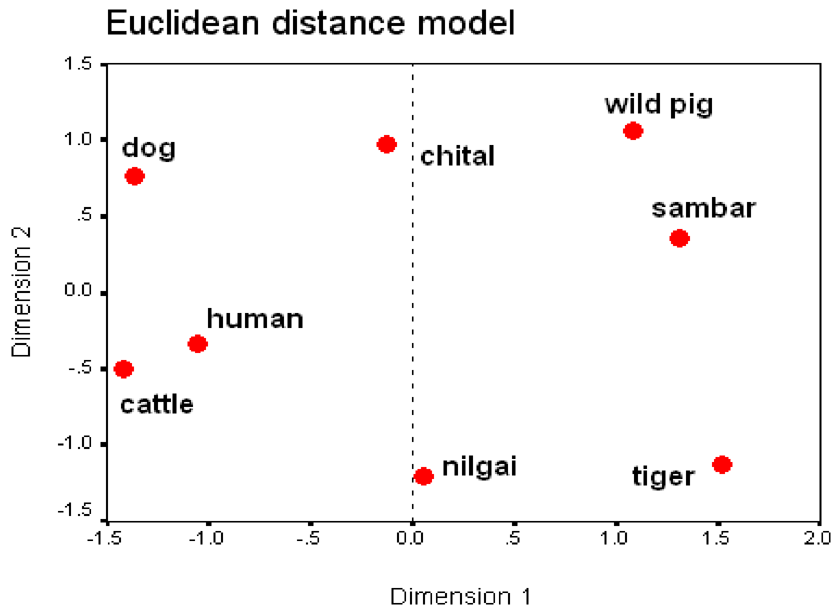


Figure 5.4. NMDS on winter track count data from all the streambeds
Kruskal's Stress = 0.154; R^2 = 0.801

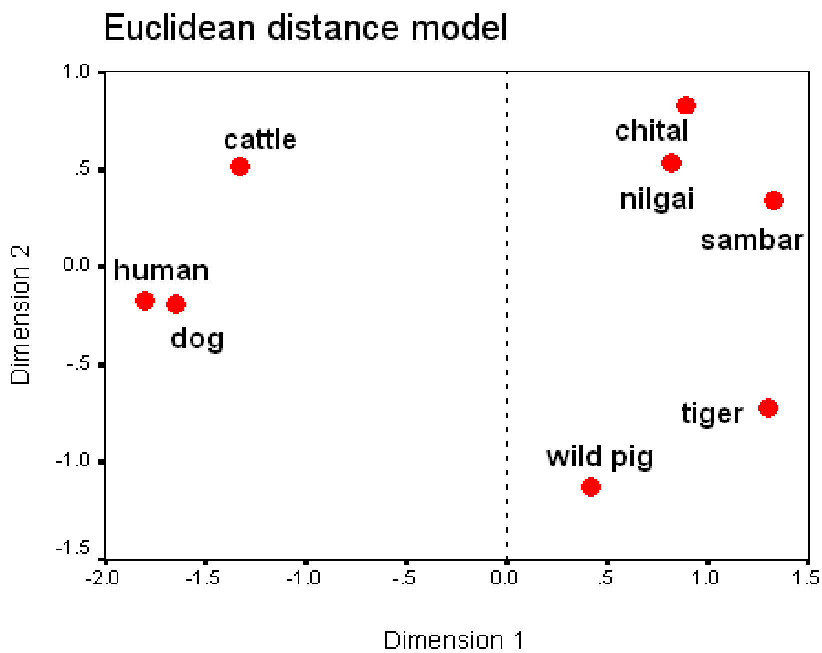


Figure 5.5. NMDS on summer track count data from all the streambeds
Kruskal's Stress = 0.076; R^2 = 0.969

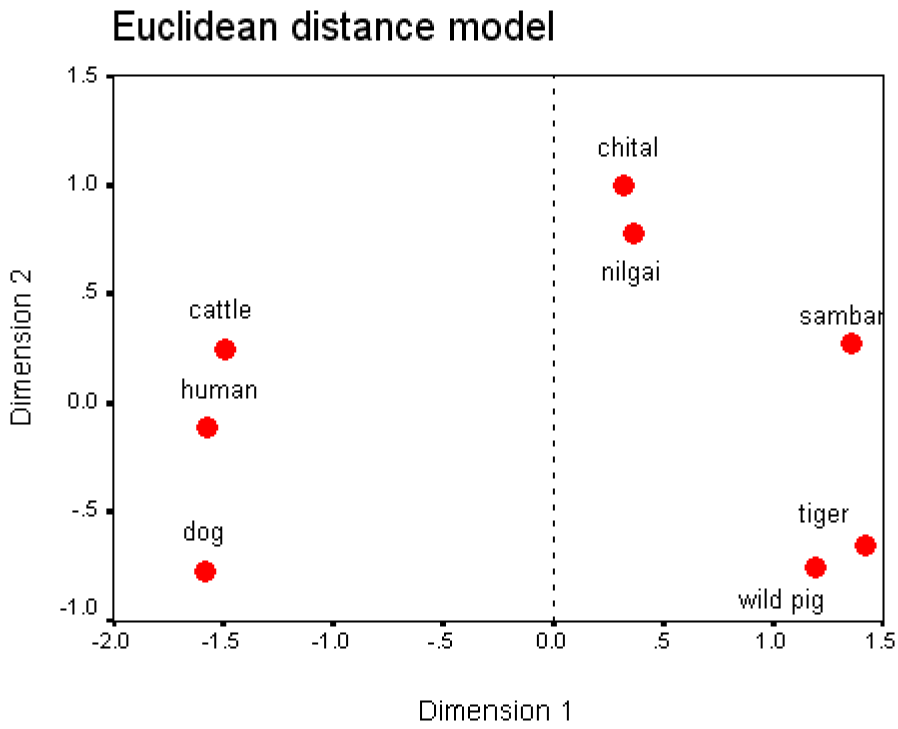


Figure 5.6. NMDS on winter and summer track count data from all the streambeds. Kruskal's Stress = 0.115; R^2 = 0 .930

Non metric multi dimensional scaling (NMDS) was performed on the track data that was obtained from the streambed transects. The winter and summer data was first treated separately before combining and analyzing. NMDS is an ordination method considered robust for most data types and is amenable to transformations. The intention with the non-metric method is to moderate the often violated assumption of linearity (change in value of one variable is directly proportional to the change in value of another) in the data with a weaker and less problematic assumption of monotonicity (paired variables must increase together, or as one increases the other must not decrease). It is also flexible in terms of the type of dissimilarity measure used for describing patterns (Quinn and Keough 2002). It employs ranks of the distances observed and those predicted by dissimilarities.

NMDS was employed after Z transforming the track count data to standardise the dataset. The stress value (Kruskal's stress) that is a measure of departure from monotonicity, presents the fit of the data, along with the R^2 value. The winter track data (Fig. 5.4) shows the similarity of tiger occurrence and sambar occurrence. Although, nilgai and wild pig not as close by, the four variables clearly separate out on the right side of the y- axis. Chital is a little off the axis on the right side. This scattered pattern could be a result of larger movement patterns shown by most animals. The notion that human, dog and cattle tracks indicate disturbance as far as tiger occurrence is concerned is visually proved in the plot. The pattern that emerged from the summer track count data (Fig. 5.5) showed the occurrence of tiger and its prey species towards the far side of the right side of y- axis. The disturbance variables were clearly separated with a very significant stress value (Kruskal's Stress = 0.076; R^2 = 0.969). The combined dataset (Fig. 5.6) was closer to the summer pattern, with wild pig attaining more similarity to tiger occurrence followed by sambar (Kruskal's Stress = 0.115; R^2 = 0 .930).

Relative abundance of Prey species

The pellet densities are significantly different for some species. However, the overall wild herbivore density in evacuated and occupied areas (Fig. 5.7) does not show a statistically significant difference ($t = 0.107$, $p<0.9$).

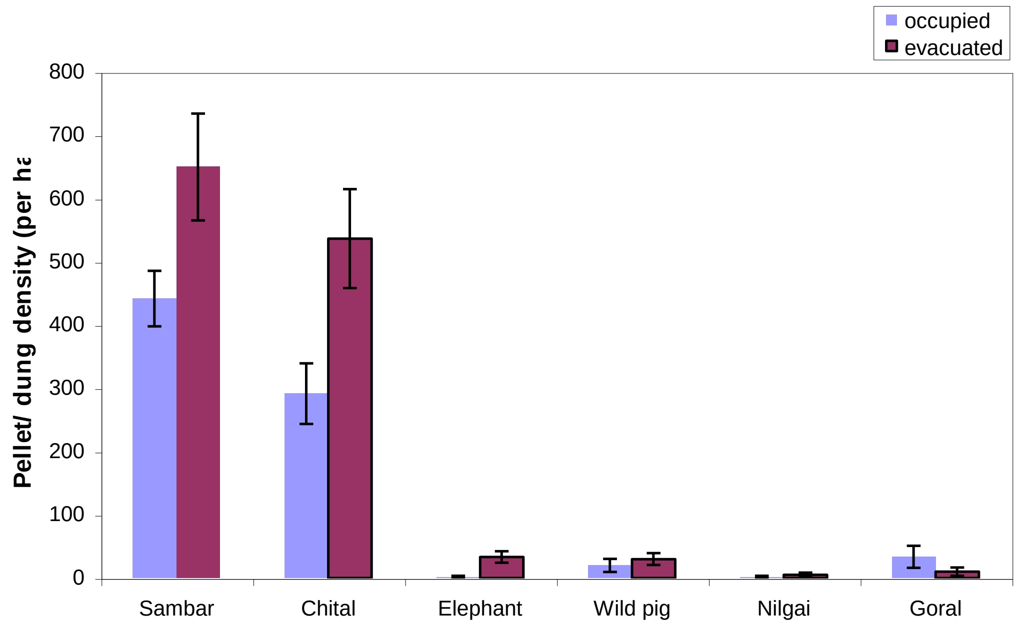


Figure 5.7. Differential pellet density occurrence of the main wild herbivores in the evacuated and the occupied areas.

Table 5.5. Pellet densities (per ha) of wild herbivores in the study area

	Evacuated area	**Occupied area**
	mean ± SE	**mean ± SE**
Sambar	650 ± 57.02	441.66 ± 39.73
Chital	536.66 ± 92.16	291.66 ± 52.18
Elephant	33.33 ± 10.05	1.66 ± 1.66
Wild pig	30 ± 9.29	20 ± 7.59
Nilgai	5 ± 2.85	1.66 ± 1.66
Goral	10 ± 3.97	33.33 ± 15.53

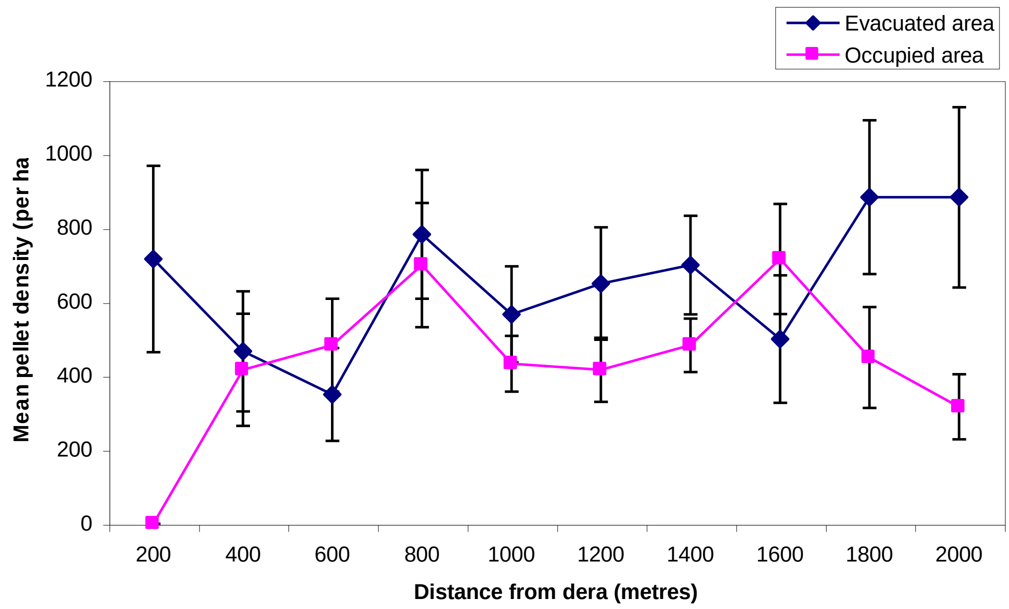


Figure 5.8. Variation in sambar pellet density distribution along the radiating transects from Gujjar deras in the study area

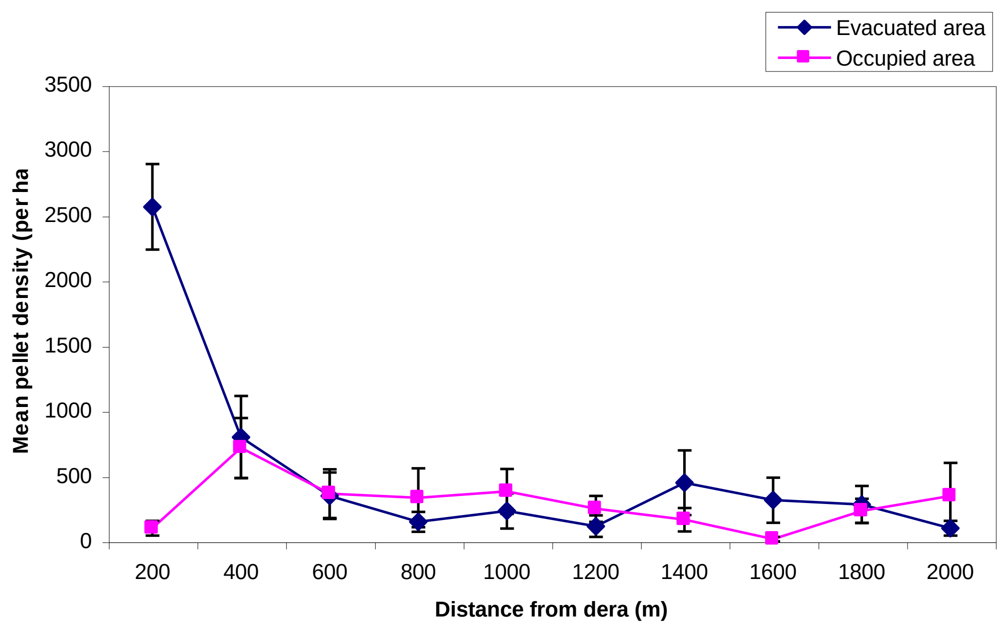


Figure 5.9. Chital pellet density along the radiating transects from deras in the study area

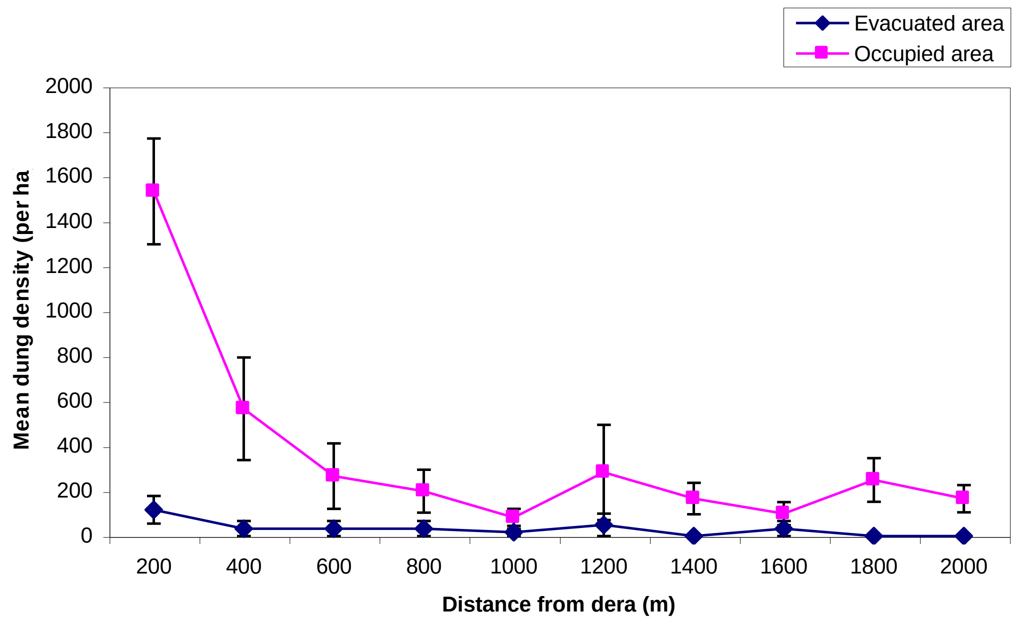


Figure 5.11. Cattle dung density along the radiating transects from deras in the study area

The sambar pellet density distribution (Fig. 5.8) occurring in the evacuated area and the occupied sites were significantly different (t = 2.46, p = 0.02). However, they seem to be at slightly lower densities at 600 and 1600m. Fig. 5.9 shows the pattern of chital pellet distribution in both the evacuated and the occupied areas. An independent sample 't test' was performed and showed no significant statistical difference between the distributions (t = 1.01, p = 0.33) at a significance level of 0.05. There is a large gap at all the plots that are 200 metres away from the deras. Patterns of pellet distribution are similar at all the other points along transect. The areas closer than 400m are closer to the deras and therefore are not frequently visited by chital. Humans and buffaloes deter their use of the areas close to the Gujjar deras. Distribution of cattle in Fig. 5.11 shows a pattern that was expected from the occupied area. However, there is noticeable movement of cattle into the protected areas as well.

Variation in habitat characteristics in relation to Evacuated and Occupied areas

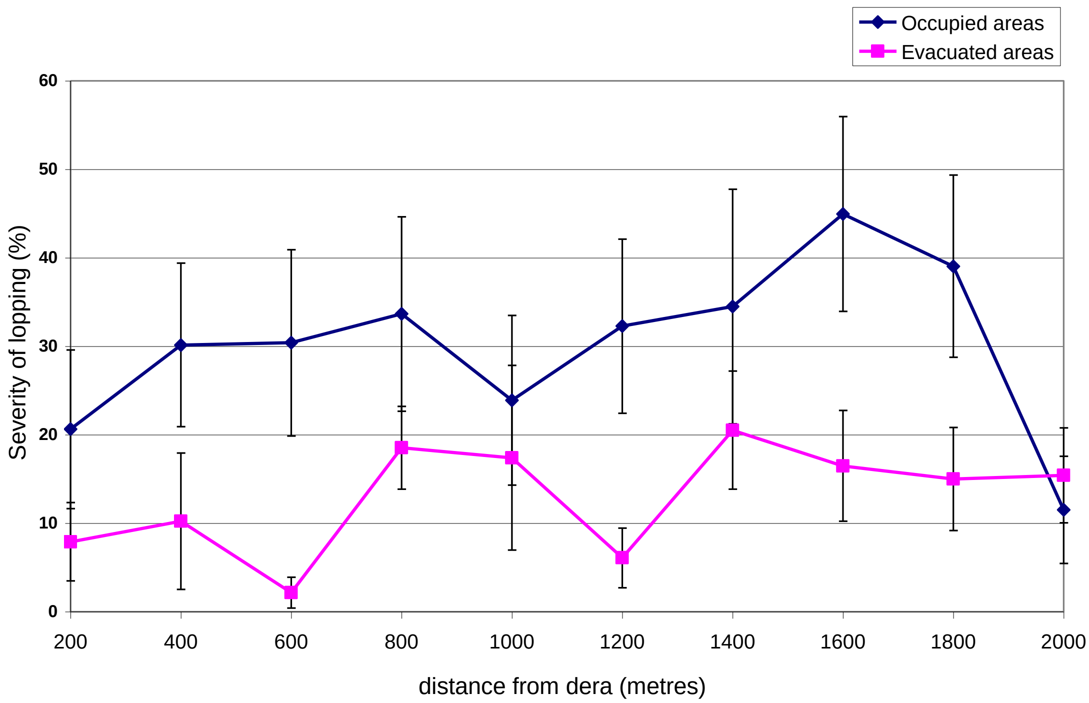


Figure: 5.13. Variation in severity of lopping along the linear distance from Gujjar dera

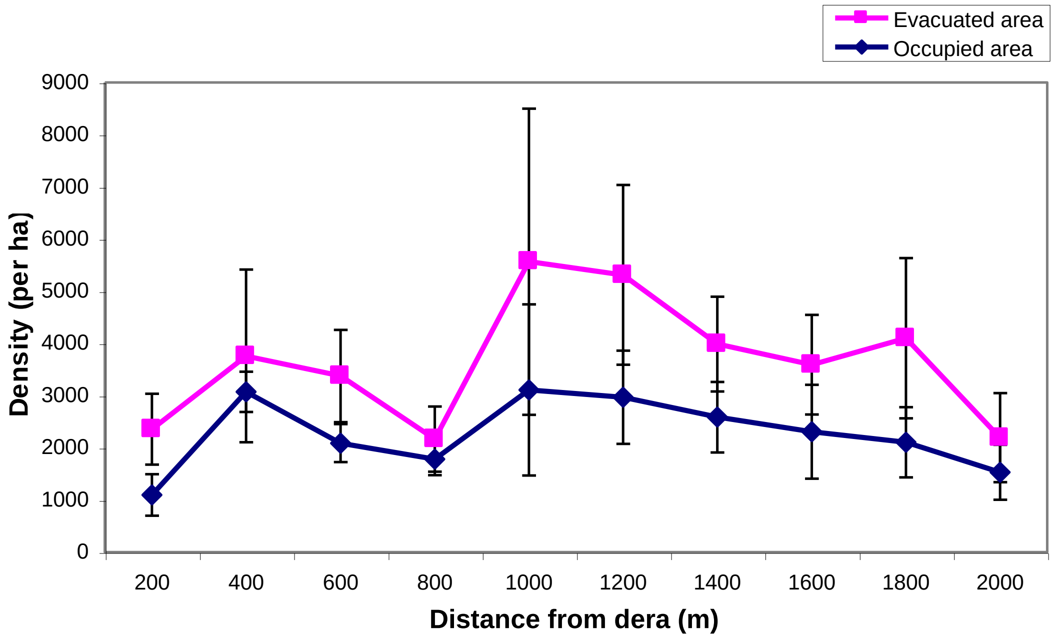


Figure 5.14. Differences in shrub density across evacuated and occupied areas

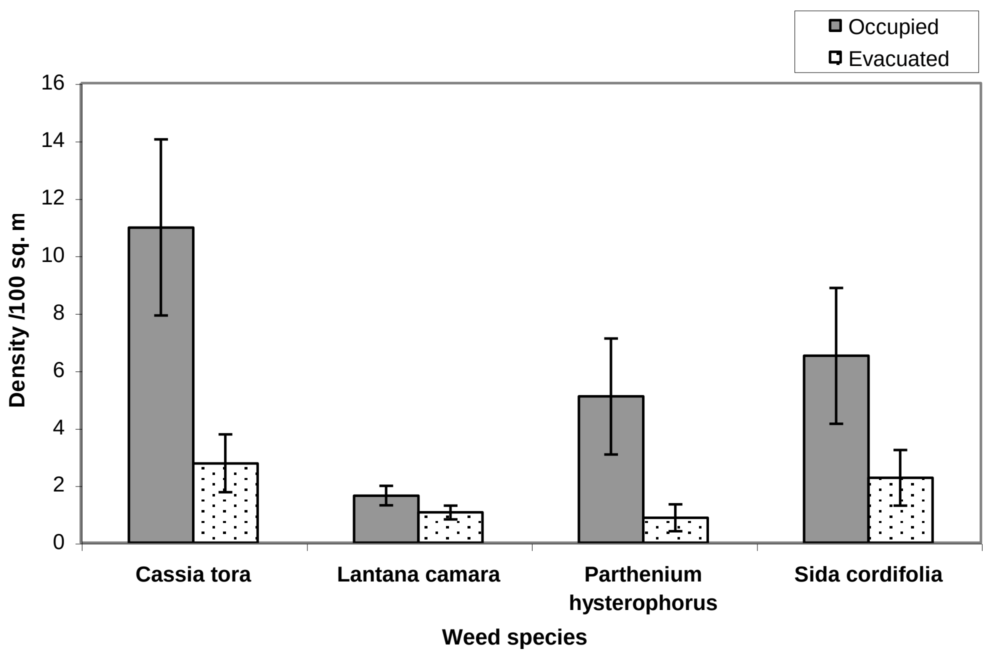


Figure 5.15. Difference in weed densities in evacuated and occupied areas.

Fig. 5.13 shows a clear trend that severity of lopping is drastically different in the occupied areas compared to the evacuated areas. Most of the lopping in the evacuated area is just the evidences of lopping found more than thirteen months back, as compared to the occupied sites where lopping is still rampant. A two sample 't test' shows that in terms of severity of lopping, the two areas are significantly different ($t = 3.37$, $p<0.003$). It indicates that as a result of ongoing pressure on the forests in Shyampur lopping is much more as compared to Chilla where people are no more lopping trees. Although one cannot be confident, it could also be indirectly revealing details about the livestock population densities that are present in Shyampur that is being supported by the forest resources.

Shrub density in the evacuated and occupied area is significantly different ($t = 3.16$, $p<0.005$) and is shown in Fig. 5.14. Much of the shrub cover being contributed in the occupied sites is by *Lantana camara.* The evacuated areas have many species that contribute to shrub cover, and many of them although shrub sized are actually tree species. Among weed species *Cassia tora, Parthenium hysterophorus,* and *Sida cordifolia* show a difference in densities in the two areas (Fig. 5.15). *Lantana camara* does not seem very different in terms of abundance in either area. The reason could be because of its perennial nature and invasibility.

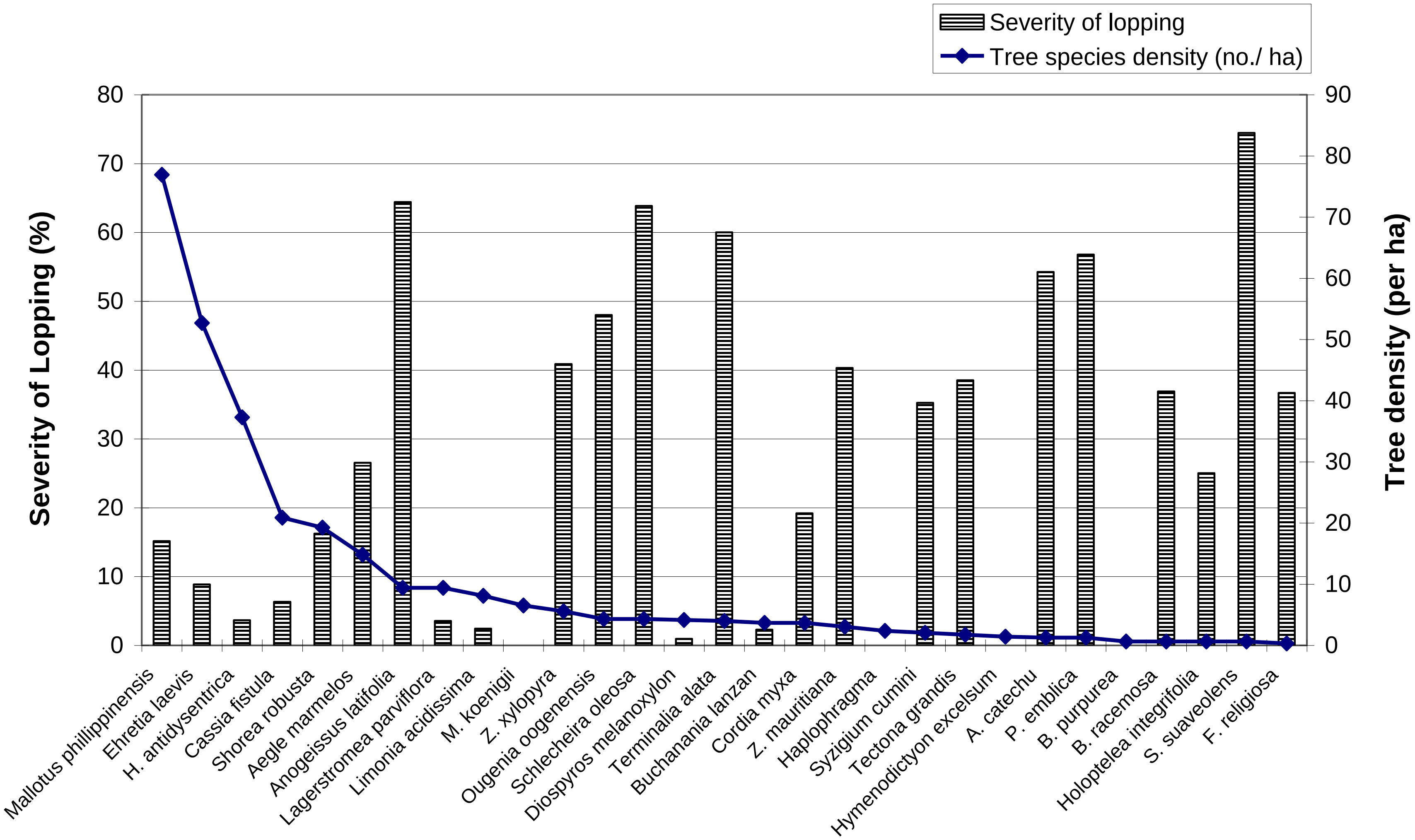


Figure 5.16. Tree species density and corresponding lopping percent

Table 5.6. Lopping percentage (total no. lopped/ total no. found) observed on tree species from study area

	Occupied area		Evacuated area	
Species	**Total no. of trees**	**Lopping %**	**Total no. of trees**	**Lopping %**
Acacia catechu	5	100	3	66.6
Anogeissus latifolia	29	79.3	30	90
Aegle marmelos	43	48.8	23	4.3
Buchanania lanzan	20	5	3	0
Bauhinia purpurea	1	0	3	0
B. racemosa	3	33.3	3	33.3
B. vahlii	15	100	4	0
Carissa opaca			2	0
Casaeria tomentosa	1	0	1	0
Cordia myxa	8	50	8	0
Cassia fistula	40	20	89	88.7
Diospyros melanoxylon			26	3.8
Ehretia laevis	141	19.1	170	10.5
Ficus benghalensis	1	0	1	0
Ficus religiosa	2	50		
Grewia tiliafolia			4	0
Holarrhaena antidysenterica	133	8.2	96	3.125
Haplophragma adenophyllum	14	0	1	0
Haldina cordifolia	1	0	3	0
Hymenodictyon excelsum	2	0	4	0
Holoptelea integrifolia	2	50	2	0
Kydia calycina			2	0
Limonia acidissima	13	0	30	10
Lagerstromea parviflora	50	6	8	0
Milletia auriculata	3	0		
Murraya koenigii	9	0	25	0
Mallotus phillippinensis	152	19.1	308	27.2
Ougenia oogenensis	14	71.4	13	53.8
Phyllanthus emblica	6	66.6	1	0
Pinus roxburghii	1	0		
Phoenix sylvestris	5	0		
Semecarpus anacardium	1	0	1	100
Syzigium cumini	8	25	3	100
Schlecheira oleosa	14	85.7	4	0
Shorea robusta	63	44.4	52	11.5
Stereospermum suaveolens	5	80		
Terminalia alata	12	100	10	60
T. belerica	1	100		
T. chebula			4	0
Tectona grandis	10	50	1	0
Unidentified sp.	1	0	1	0
Ziziphus mauritiana	18	66.6	1	100
Z. xylopyra	24	70.8	8	12.5

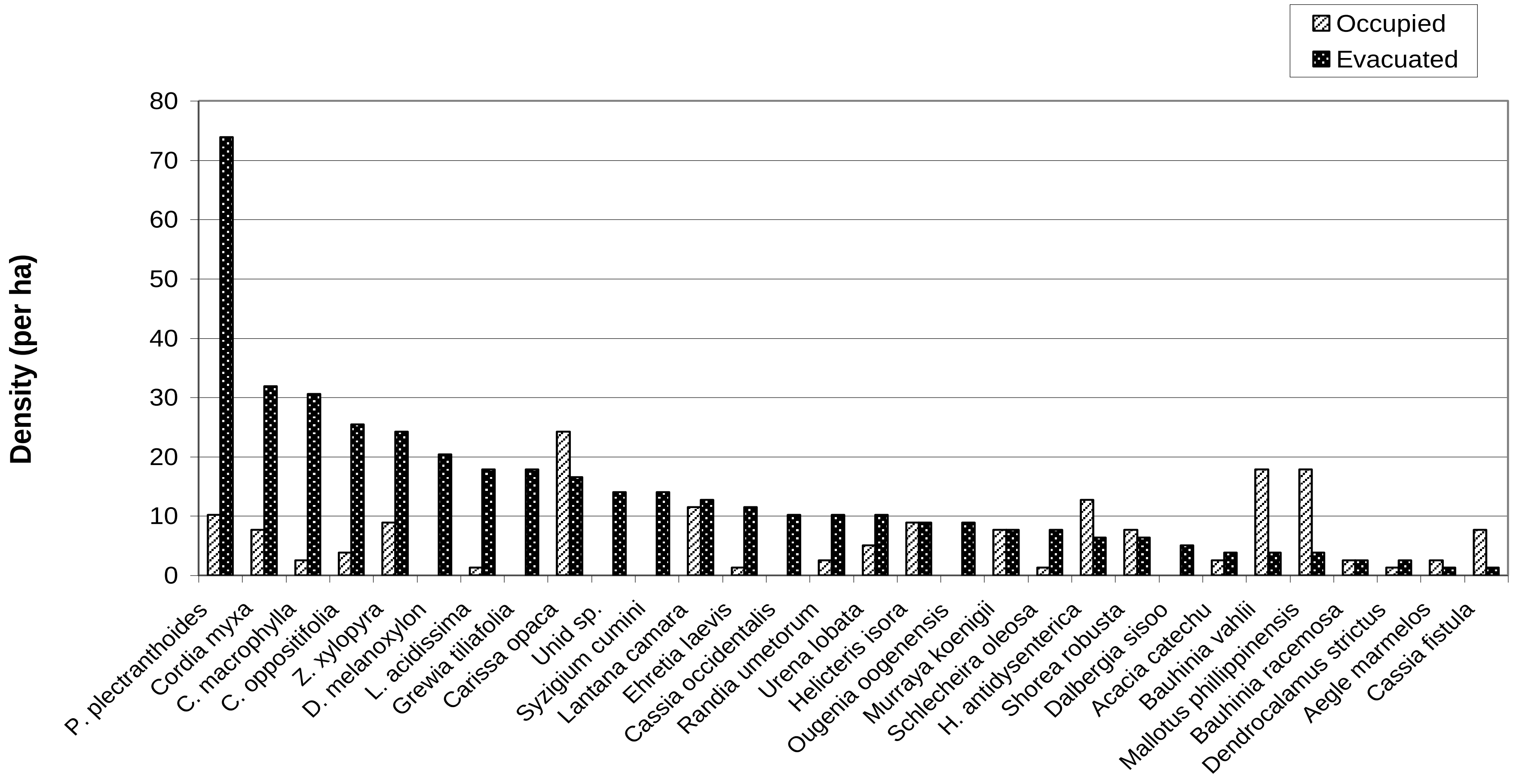


Figure 5.17. Shrub densities in both evacuated and occupied areas

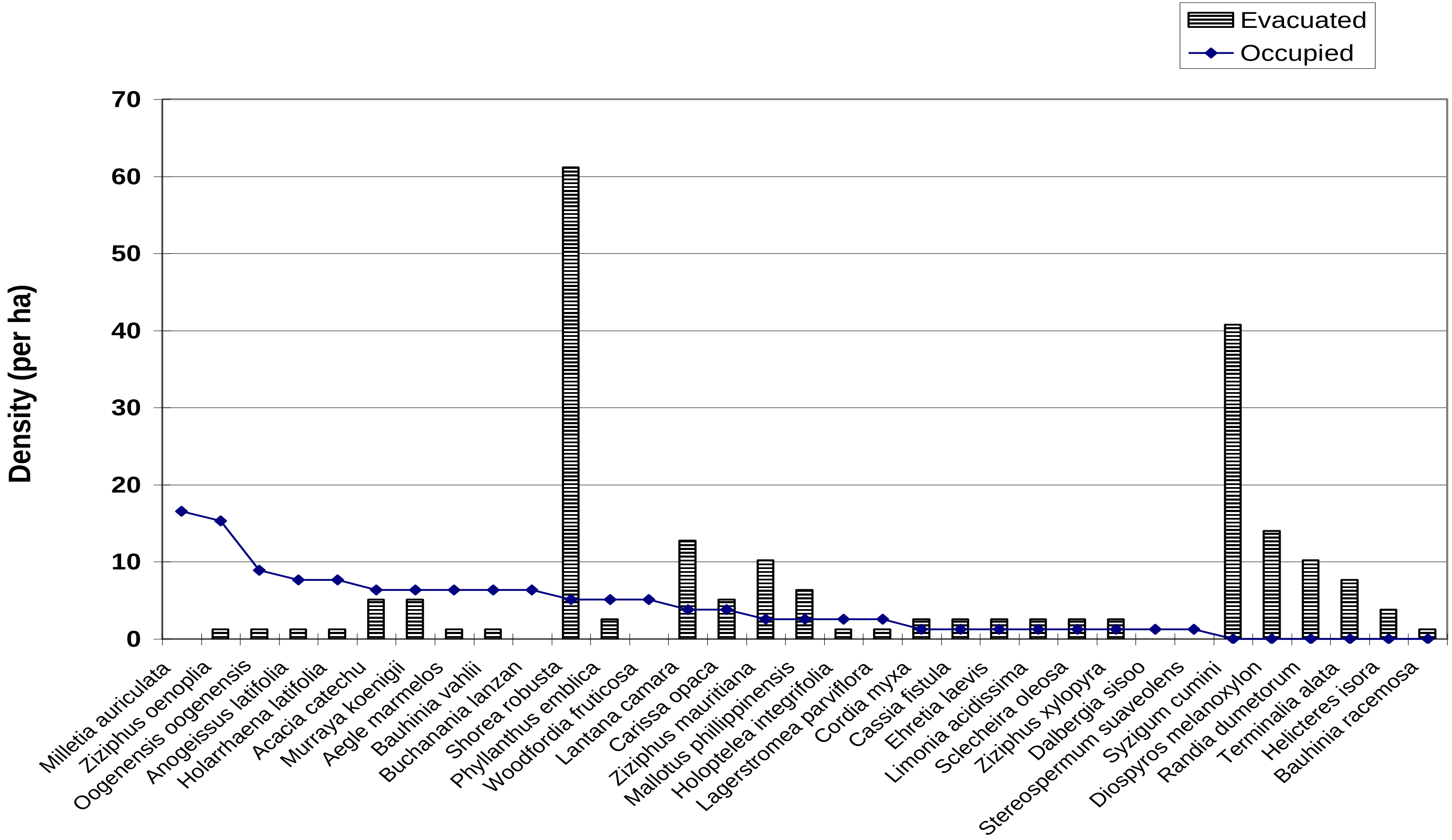


Figure 5.18. Sapling densities across the evacuated and occupied areas

The density per hectare of all tree species lopped (Fig. 5.16) reveals the excessive lopping of many species that are relatively low in density in the study area. Some of the more abundant species like *Mallotus phillippinensis, Ehretia laevis, Cassia fistula, Shorea robusta, Holarrhaena antidysenterica* are not good as fodder or firewood and is therefore not lopped. Areas with these species lopped indicate signs of desperation. The more favoured loppable species (used more than in proportion to availability) are *Anogeissus latifolia, Terminalia alata, Ziziphus mauritiana, Z. xylopyra, Ougenia oogenensis, Acacia catechu, Phyllanthus emblica* and *Bauhinia racemosa*.

Higher densities of shrub are found in the evacuated areas as compared to the occupied areas (Fig. 5.17). Only three species *Carissa opaca, Bauhinia vahlii* and *Mallotus phillippinensis* show higher shrub density in the occupied areas. *Mallotus* is a species that is heavily fed on by elephants in most places of Chilla Range. However, areas of Shyampur Range do not have much elephant use as shown in Fig. 5.10. *Mallotus* trees are in full bloom in all these areas. *Bauhinia vahlii* is a species that is found more in the miscellaneous type forests that is the predominant vegetation type in the sampled areas of Shyampur Range. So is the case with *C. fistula* and *H. antidysenterica*.

Inspite of all the vegetation type differences, saplings of many trees like *Terminalia alata, B. racemosa, Syzigium cumini, Z. mauritiana* and the most dominant of the forest types, *S. robusta* are relatively higher in the evacuated areas (Fig. 5.18), which is a promising sign. However an independent sample ‘t test’ shows that the difference in means of saplings in the two areas are not significantly different at an alpha level of 0.05 ($t = -1.49$, $p<0.15$).

Regression tests revealed that in both occupied and evacuated areas the relation between lopping and distance to the nearest dera was very weak (not shown here)- marginally negative in occupied areas and marginally positive in evacuated areas. As far as disturbance is concerned, the effect of a human settlement has a negative relation if they are residing in the same place, and using the same space for lopping and cattle grazing etc. What is interesting in this perspective is the Fig. 5.19, which shows the locations of gujjar deras (both occupied and evacuated), with an average zone of influence of two km (the 2 km buffer) around every dera. The information about the distance traversed by Gujjars for firewood and lopping was given to me by Gujjars and forest guards. The figure shows high overlap of these zones of influence that could be a reason that may suggest very erratic patterns; neither with a particularly strong positive nor negative skew. Each Gujjar family has a particular permitted area within which they are supposed to lop and graze their buffaloes. However, this is not followed strictly as the Gujjar population density have tremendously increased over the years that a particular zone of usage is utilized by more than the permitted number of families.

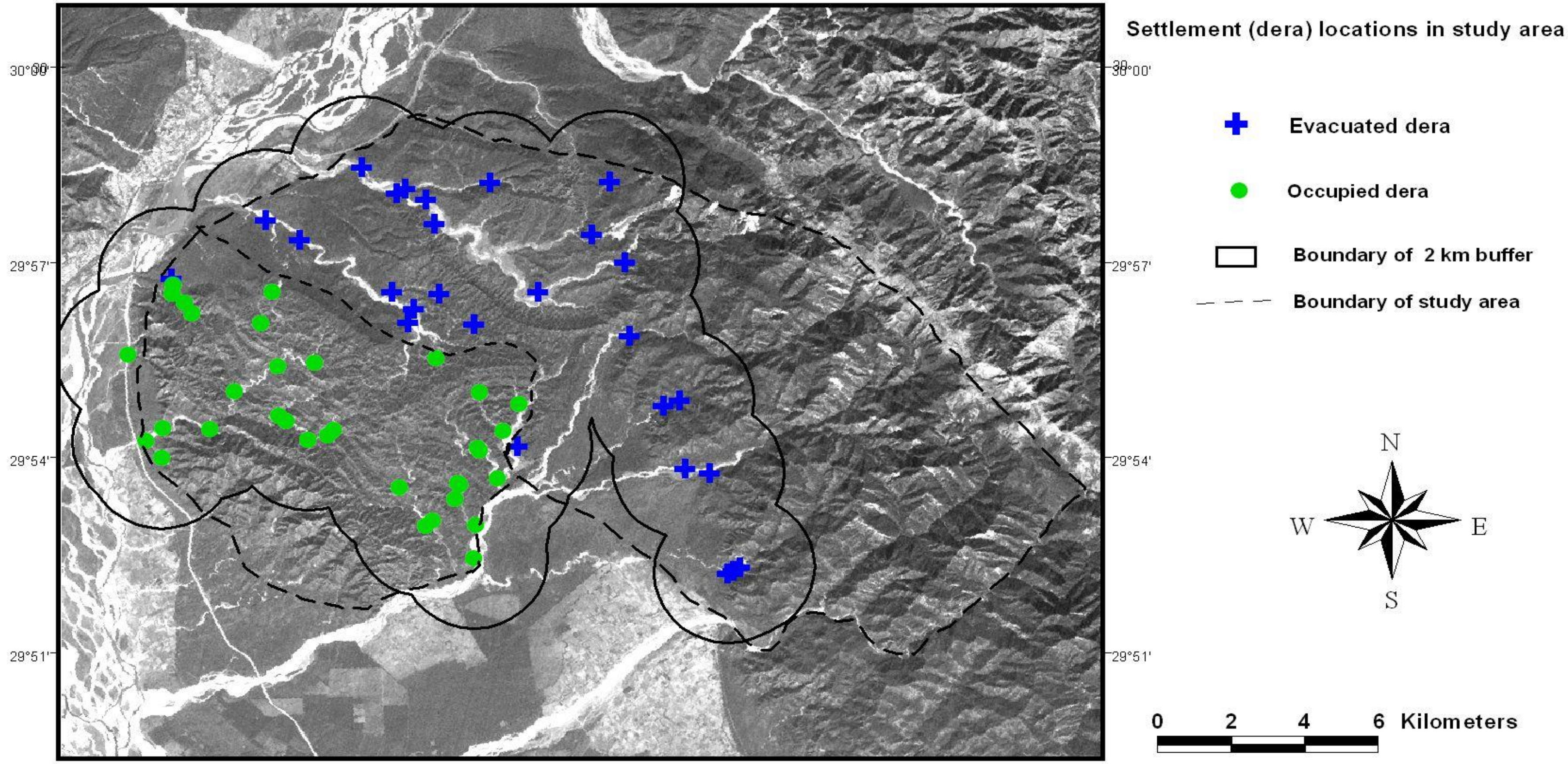


Figure 5.19. Two km buffer around settlement (dera) locations indicating zone of influence

Result of Principal component analysis on the evacuated and occupied sites

Table 5.10. Variation explained by each variable in the selected components for explaining the total variation of the dataset

Rotated Component Matrix

	Component 1	Component 2
DISTDERA	.260	.734
CANOPY	.833	.131
SHRUBDEN	.594	.274
CATTLE	.106	-.831
TREEDEN	.789	-.175

The matrix of component values was rotated before finalizing the results. The rotation method used was varimax orthogonal rotation with Kaiser Normalization. The rotation converged in 3 iterations.

Initial variables used to perform the PCA were DISTDERA – distance to the nearest dera, EXTLOP – extent of lopping, CANOPY – canopy cover, TERRAIN – terrain, SHRUBDEN – shrub density, SAPLINDE – sapling density, CATTLE – cattle dung density, TREEDEN – tree density, LANTANA – weed density (only *Lantana*)

Table 5.11. Results of the PCA with explained variances by each component

	Initial Eigenvalues			Extraction Sums of Squared Loadings			Rotation Sums of Squared Loadings		
Component	Total	% of Variance	Cumulative %	Total	% of Variance	Cumulative %	Total	% of Variance	Cumulative %
1	1.859	37.175	37.175	1.859	37.175	37.175	1.749	34.973	34.973
2	1.242	24.840	62.016	1.242	24.840	62.016	1.352	27.043	62.016
3	.781	15.623	77.639						
4	.669	13.373	91.012						
5	.449	8.988	100.000						

The iterations for the PCA had to be done three times to get the most suitable and acceptable solution. The initial iterations using all variables revealed the high positive loading (>0.5) of the LANTANA variable (weeds) on two components indicating a complex structure of the variable loading. It therefore had to be removed. Keeping variables with complex structure can mislead the interpretation of the PCA. During the second iteration the variable EXTLOP (extent of lopping) was dropped from the analysis as a result of a very low communality (correlation) value of 0.141. Usually a PCA requires the communality of variables to be greater than 0.50, i.e. there should be a high correlation. Also, this variable was not significantly loaded on either of the two extracted components (loadings < 0.4). The variable representing sapling density (SAPLINDE) and TERRAIN was low on communality (0.324) and therefore had to be removed. The subtraction of four variables however did not affect the analysis considerably as the cumulative percent variation explained by the two extracted components was still only 54.3 %. The final iteration extracted two components that were explaining 62 % of the variation in the dataset. The KMO measure of sampling adequacy was not required as the PCA was based on a correlation matrix and not a covariance matrix. The Bartlett's sphericity test was significant. Communality values for the remaining variables were fairly larger than 0.5 (except SHRUBDEN which was less than, but close to 0.5). The requirement of the derived components explaining more than 50% or more of the variance in each of the variable was also achieved.

The components were rotated via varimax orthogonal rotation. It is one of the most common orthogonal rotations and is done merely to interpret the data better, and the process is known to keep the geometry of the constellation of points and the cumulative variance intact (McCune & Grace 2002).

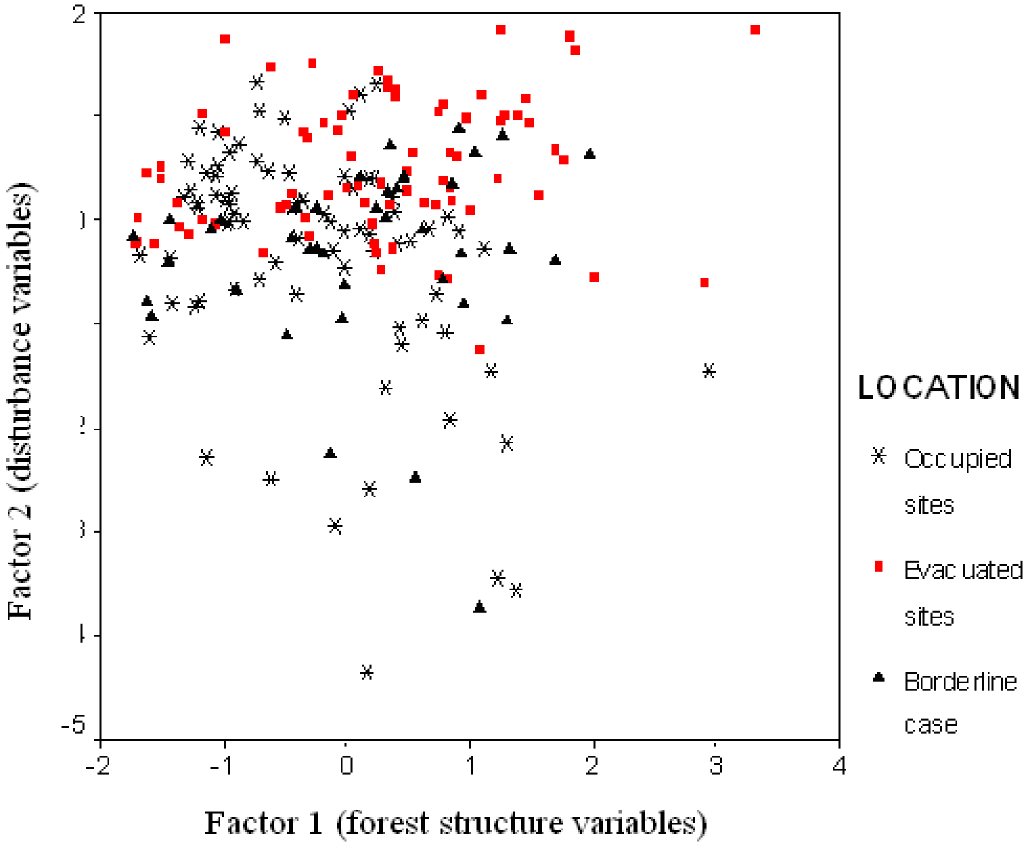


Figure 5.20: Factors plotted against each other with the sampling sites at the periphery of both the Ranges highlighted – the borderline cases

In the Table with explained variances (Table 5.11), 62 % of the variability in the dataset is explained by the first two components. The other three components lack the discriminatory power to explain the rest of the variation. This reason allows the selection of only those components that contribute significantly, i.e. the first two. One of the rules of thumb while performing a PCA is to select only those components that have an eigenvalue greater than one. The reasoning is as follows- all the eigenvalues of the PCA sum up to the number of variables. In this case it is five. The first two components add up to 3.101. It is an indication that the cumulative variance explained by these two components involves the interaction of more than three variables out of the five given. The explanatory power is therefore large enough to employ only these two components for further analysis to reveal the patterns in the dataset.

The component 1 has high positive loadings of variables that are representative of the structure of the forests - canopy, shrub density and tree density. On the other hand, component 2 has high loadings of disturbance related variables like cattle dung density and distance to the nearest dera.

The extracted factors are capable of discriminating between the two areas. But, as is seen in the graph showing the borderline cases (Fig 5.20), there exists certain sites (deras) that was sampled near the park border. The effects of them fall between the totality of the cluster of points, thus indicating that this could only be a gradient that exists between the areas that are occupied and the areas that face lesser human disturbance as a result of the evacuation.

5. Discussion

It is evident that Chilla range that is presently void of human disturbance is clearly in better structural condition than adjoining Shyampur range that is human occupied. Part of the reason is that Chilla is part of a larger protected area, which has good forest cover. It seems that in such a short period a full-fledged recovery cannot be perceived. A comparison with a nearby human dominated (Gujjar communities) area with similar land use patterns was to give an insight into the changes human presence can do to an ecosystem. Prey and predator movement are affected by differing intensities of human disturbance and it was necessary to study two areas to comprehend and distinguish the events in one of them, despite a proper control site not being part of the experiment – simply because such a no-disturbance zone is unavailable in this highly populated landscape.

Tiger occurrence patterns

The encounter rate of pugmarks that were found on the eight different streambeds indicates the trend that the maximum occurrence of tiger was in those areas that are devoid of human settlements. Amgadi and Mundal (evacuated areas, within Chilla Range) were reported to have 0.15 and 0.05 as the pugmark encounter rates respectively during the same 5km walk surveyed while the Gujjars were still present in Chilla Range (Johnsingh *et al.* 2004). Comparing this with the now available pugmark encounter rates of 2.05 ± 0.40 and 1.9 ± 0.44 for Amgadi and 0.4 ± 0.18 and 0.3 ± 0.10 for Mundal gives a fair perception about the differences before and after the Gujjar relocation. This just goes on to reiterate the point that at local scales, if not regional, high human population density is associated with large mammal declines (Forester & Machlis 1996, Cellabos & Ehrlich 2002, Parks & Harcourt 2002). Mundal yielded a lower encounter rate than expected (considering the movement patterns of tigers that were noticed during the non sampling period). This could just be a consequence of trying to generalize occurrence patterns vis a vis a single sample. What seemed interesting however, was the high pugmark encounter rates on the streambeds near the peripheries of the park during summer (Fig. 5.1). Although certain parts get closer to the boundary, the water availability is fairly good (Fig. 5.2) and the habitat is more suitable with more shrub cover and canopy. This is of utmost importance as far as park managers are concerned, as fringe areas of Chilla Range seem to be very conducive for tiger movement.

Most area within Chilla Range shows a higher pellet density of the main ungulate prey species (Fig 5.7). The NMDS results also go on to prove the affinity of the association of tiger with its prey species as compared to disturbance inducing factors such as human and livestock presence separately (Fig 5.6). The two dimensions of the plot are representative of habitat space. Giving ecological meaning to those dimensions, the first dimension is clearly a dimension of a composite disturbance gradient – the tracks of human, cattle and dog falling nearer to high disturbance as against tiger and most prey species falling away from it. The second dimension is a little complex and seems like a compound dimension with presence of water being a likely variable explaining most of the variation of the dissimilarity between the entities. The stretches of Amgadi, Ghasiram, Luni and Mitawali had the maximum pugmark encounters during both winter and summer (Fig. 5.1). What was disturbing is that the two 5 km stretches of Diyawali and Sidh soth (streambeds in occupied area) which is spatially not too far away from Chilla Range did not yield even a single tiger pugmark. Khaara was generally low on tiger pugmark occurrence being on the periphery and more towards Shyampur Range.

Some of the trends that are definite are as follows

- Diyawali and Sidth soth (representing the human occupied area) does not seem to have tiger movement
- There maybe other factors (water, prey availability, territory marking) that may also be an interactive role in the pattern of tiger occurrence in most streams within Chilla.

Habitat and prey relations

One of the clear patterns emerging is the difference in structure of the forests in the two Ranges of Chilla and Shyampur as a result of differing human use patterns (Fig. 5.12 & Fig. 5.13). Compositional change is also evident at certain strata. The patterns of lopping that are seen in both the areas at present gives an indication of the structure of the forest. Significantly different in terms of statistics, the effect of excessive lopping in Shyampur Range (Fig. 5.12 & Fig. 5.13) is also seen in the parched landscape that is present with very little shrub cover and canopy. As far as the vegetation classification for the study is concerned, most of Shyampur Range falls in the Miscellaneous with sal and Miscellaneous category (the more disturbed forest types explained in the following chapter of the report). In terms of the tree species lopped as against its available densities, the resource use patterns are skewed to damage these two forest categories, particularly the miscellaneous type which has tree species of low overall densities like *Terminalia alata, Anogeissus latifolia, Bauhinia racemosa, Schlecheira oleosa* etc (Table 5.6). Fruiting trees like *Ziziphus mauritiana* are also lopped heavily and their regeneration is also very poor in these areas. In this context Chilla Range, from where Gujjar relocation took place about a year and a half back, shows a very different structural development.

One can never be sure whether or not the effects of Gujjar communites in these two Ranges were exactly the same. Ideally speaking, it can never be, as other confounding variables in the form of loppable tree species density, human population density, livestock density etc can be different. But the reasons for a comparative examination is based on the fact that the areas fall in the same landscape, is a 'Bhabar tract', with similar topography and are spatially not too far from each other and is therefore deemed available for tiger and prey species utilization. The extent of utilization as a response to changing human disturbance regimes is what the research hypothesis tests. It is fact that the land use patterns in the two areas are similar and in this context it is important to view the implication of the Fig. 5.19, which infers that the common 2 km stretch that most gujjars travel to lop and gather firewood actually damages the habitat as a result of the overlap of the zone of influence of any two given deras. Although it seems a negligible distance with respect to the size of the landscape, the effect this causes as a result of the density of people living in this area is the root cause for the present state of affairs.

Chilla Range shows a better picture more in terms of floristic structure than compositional development. Although not exactly quantified, the regeneration of the lopped trees is prolific with human disturbance coming to a complete stop. The presence of high densities of pellet groups of sambar, elephant and chital in Chilla, as against Shyampur (Fig 5.7 & Table 5.5) is a clear indication of wider use patterns, if not higher densities of ungulates in Chilla. Competition from livestock over space and shared resources can be a very important governing factor that impact wildlife, in particular wild ungulates (Madhusudan 2004) and this is the scenario in most of the Gujjar occupied areas. Sambar and chital clearly seem to avoid the use of areas that are closer to human settlements (Fig. 5.8 & Fig. 5.9). Although this study does not indicate a clear preference of sambar for more shrub density, an earlier study specifically on the habitat preference of sambar in Rajaji National Park revealed a preference for good forest under cover (Bhatnagar 1991). According to recent scientific literature tiger densities in protected areas are being mediated more by prey abundance than any other single factor (Karanth et. al. 2004). Sambar is a more important contributor as prey species for tiger in Chilla Range (Harihar pers. comm. - results of tiger scat analysis), but the reasons for the higher densities of sambar are still unclear. Chital on the other hand are not known to have any particular preference to high shrub cover areas (Bhat 1993) but prefers open grassland patches.

The Shrub density is higher in Chilla Range and although species representativeness is similar, densities of many non weed, non palatable species are much higher. Sapling densities of many of the browse species like *Terminalia alata, Bauhinia racemosa* and fruit plants like *Ziziphus mauritiana* are also higher in Chilla. Saplings and shrubs of other important herbivore species like *Ehretia laevis* and *Mallotus phillippinensis* are also higher in Chilla Range. Weed densities are also relatively higher in human dominated areas as against forested patches where competition from native species prevents their proliferation. Annual exotic weeds like *Cassia tora,* and *Parthenium hysterophorus* are relatively higher in density in the Shyampur Range. *Sida cordifolia* which is a weed of degraded grazed lands (Mishra & Rawat 1998) is also common in the Gujjar occupied areas. Growth and regeneration of native vegetation in Chilla Range seems to have lowered the weed densities. The matter of concern with weedy species is that given their inherent properties of invasibility, what matters is the degree of disturbance a region faces that makes it succumb to weed invasion (Lonsdale 1999). A case in point is the abundance of annual plants in the recovering Chilla range especially of poaceae family like *Imperata cylindrica, Vetiveria zizanoides, Saccharum spontaneum, Phragmites karka* that can completely obstruct the growth of most weeds. In the presence of livestock the grasses are selectively grazed upon, which exclude the use of these areas by wild ungulates and also adds to weed proliferation. However, *Ageratum conyzoides* is present in a carpet form in most areas of Chilla Range where moisture levels are high (mostly the northern aspects). Shyampur Range owing to the large canopy openings has relatively less growth of *A. conyzoides.*

In terms of overall differences between the two areas the PCA factors (Factor 1 – representing effects of forest structure and Factor 2 representing effects of disturbance variables) discriminated between the evacuated and occupied sites (Fig. 5.20). However, there is a moderate mixing of sites indicating that there maybe a similarity of certain nature between some of the occupied and evacuated deras. One can give reasons for this apparent coalescing of sites. The two Ranges encompassing the study area are spatially non exclusive. There exists a border (the park boundary) that acts as an ecological continuum between both the Ranges. Resource use by animals and humans are therefore apparent in a certain 'illegal' buffer zone that remains open to effects both natural and anthropogenic. What it indicates is that the time period of just a thirteen months since relocation may not be enough to show clear ecological segregation of a site being disturbed and the other being relatively undisturbed. Structural aspects of the forest like canopy cover, tree density are variables that will take time to change considerably to show remarkable contrasts between habitats. In all likelihood it could be that Chilla is undergoing a successional phase, structurally at least, if not compositionally. Our suggestion to field managers would be to conduct similar habitat monitoring studies to identify such successional trends, mostly at the level of forest composition. This associated with ungulate use would give a clear picture about the recovery phase and the dynamics it follows.

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Impact of human disturbance on woody species of four different forest communities in the dry tropical forest of Siwaliks, North India

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***Abstract*:** We studied the effects of human disturbance on four different forest types in the forest of Chilla Range within the Rajaji National park. The occurrence of these forest types was mostly based on the differing densities of Sal, *Shorea robusta*. It was found that most of the lopping of trees was mostly focused on only one forest type – that which contained the least abundance of Sal trees. The selectivity of tree species analysis showed that Sal was among the least selected for woody species. Most species found or associated with more open forest types like the Miscellaneous forest type, that also are the least dense in occurrence were most favoured for lopping. The high diversity of this forest type as compared to low diversity of leats disturbed Sal forests indicate a change in structural compostition of the forests of the region that may have important consequences for diversity of woody species in the long run.

Impact of human disturbance on woody species of four different forest communities in the dry tropical forest of Siwaliks, North India.

Amit Kurien, S.P. Goyal and Bivash Pandav

1. Introduction

The dry tropical forests constitute 42% of all forests in the tropics and are widely exploited and threatened (Maass 1995, Murphy & Lugo 1996). They constitute habitats most susceptible to disturbance as they are densely populated because of favourable conditions for agriculture and more suitably for livestock (Murphy & Lugo 1986). Widely considered as the most endangered of all tropical ecosystems (Janzen 1988), much of its landscapes have had a complex history of human land use and natural disturbance (Aragón & Morales 2003). Ecosystem characters of such areas are therefore largely determined by past land use (Noble & Dirzo 1997, Ogden *et al.* 1998). Despite such pressure ecological studies on degradation and restoration in tropical dry forests are few (McLaren & McDonald 2003, Sánchez-Azofeifa *et al*. 2005) and the response of these ecosystems to human exploitation is not well understood and requires investigation (Maass 1995).

In India, about 40% of the forested land is dry tropical forests (Singh & Singh 1988), of which the Siwaliks of north India forms an integral part. The Siwaliks have faced anthropogenic pressure from migrating and resident human populations in this area for alteast the last one century. Resident nomadic pastoralists called *Gujjars* have been part of this ecosystem for the last 60-70 years (Kumar 1995). Substantial extraction of forest resources as a result of widespread lopping for firewood and fodder and grazing by their livestock has maintained a sustained disturbance regime within this fragile ecosystem (Edgaonkar 1995). Such demanding pressure can alter the vegetation structure and composition (Borgmann & Rodewald 2005) and may even induce land degradation and weed invasion. The forest communities of Siwaliks have also faced forestry management for timber extraction and select species were promoted for this purpose. However, the impact of these various human disturbances on the forest community remains unidentified. Though there have been studies focusing on Central Himalayan belt (Kumar & Ram 2005) and other regions of Sal dominated areas including plantations (Pandey & Shukla 2003), there have not been many studies within protected areas in North India that analyzed the effect of anthropogenic disturbance in shaping the forest communities, particularly its structure as a result of chronic disturbance. It is however crucial to identify this because of the increasing population of people and livestock within the Siwalik forests (Rajvanshi & Dasgupta 2004) that can potentially intensify lopping and related disturbance factors. Being one of the primary studies on direct anthropogenic impacts on forest communities in this region we wanted to address two key issues;

1) To assess whether anthropogenic disturbance is different in the four forest types – Sal dominated, Sal mixed, Miscellaneous with Sal and Miscellaneous - within Chilla range of Rajaji National Park. Specifically, we predicted that since forest exploitation is for fodder and firewood, lopping will not be random in nature, but in conformity with the selectivity (i.e. palatability) for tree species that constitute the respective forest communities.

2) To assess the structural attributes of these forest types.

Forest types

Forest communities of the Siwalik ecosystem are principally classified based on the association of species with the varying densities of the most dominant tree species of the region - Sal or *Shorea robusta* Gaertn.f. (Dipterocarpaceae) (Champion & Seth 1968). Most of the Siwaliks are undulating to hilly and the composite effect of aspect, moisture and slope have created four broad categories of visually identifiable forest communities. They are, Sal dominated forest community (mostly a pure Sal forest of high canopy with low light penetration), Sal mixed forest community (found largely on the northern and western slopes with higher moisture), Miscellaneous with Sal forest community (open forests with lower abundance of Sal, higher light penetration and considerably drier) and Miscellaneous forest community (almost devoid of Sal with very dry soil). These communities correspond to the major forest types Moist siwalik sal (3C/2a), Moist mixed deciduous forest (3C/C3a), Dry siwalik sal (5B/C1/1a) and the Northern dry mixed deciduous forest (5B/C2) respectively that extend to areas even outside Rajaji National Park (Champion & Seth 1968). The study area was open to anthropogenic disturbance from resident *Gujjar* communities until thirteen months before sampling. Although other forms of disturbance like grazing and weed proliferation is present, we focused on lopping as the major disturbance factor as it directly affects the canopy structure that has had a commanding influence on the moisture regimes that shaped the Sal forest communities (Joshi 1980) within this largely dry forest environment.

2. Methods

After reconnaissance, at least 6-8 sufficiently large patches of the different forest types were identified in the study area. From March to April 2005, we sampled the study area in a stratified random manner with five transects of one km each laid in five different patches of the four forest types. Sampling was carried out at every 200 m interval along the transect and data on tree density, lopping and canopy cover was recorded from a total of 25 314 m^2 plots (radius 10m) in each of the four forest types. A tree was defined as any plant with GBH (girth at breast height) ≥ 20 cm. We enumerated lopping by counting each slash mark on the trees that corresponded to the lopping of branches. The proportion of lopped or cut stems per sampling unit has often been used as a quantitative measure of human induced disturbance by various authors (Barve et al. 2005, Ganeshaiah et al. 1998, Pandey & Shukla 2003, Shaanker et al. 2004), some others use it as an index of disturbance. It has proved to be a fairly accurate measure that reflects the relative anthropogenic pressure at a site. We however, calculated two measures of lopping; 1) severity and 2) extent of lopping – the former being the ratio of branches lopped against the total number of branches (number of branches cut and number of branches remaining) and latter being the ratio of individuals of a species lopped to the total number of individuals. Both measures were expressed as a percentage. Since the data on severity of lopping from the four forest communities was not normally distributed (Kolmogorov-Smirnov test, $P < 0.05$) we compared them using a Kruskal-Wallis test. Canopy cover was estimated from the four cardinal directions of the 10m radius plots.

Other structural attributes were inventoried from nested plots within the 10m radius plots. Shrub and weed density and sapling density were estimated from 5m radius plots. Structure of ground vegetation, including tree seedlings was noted from within 1m diameter plots placed at the four cardinal directions of the 10m radius plots. The category of ground vegetation at each 0.1m was recorded as herb cover, litter or barren ground.

To determine the selectivity of tree species for lopping in the study area we used the Ivlev's electivity index, calculated using the formula

$$E = (r_i - p_i) / (r_i + p_i)$$

where r_i is the proportion of lopped trees of the ith species and p_i is the proportion of the ith species in the sampling universe (Krebs 1989). The index compares the proportion of resource used to the proportion available in a specified area. It ranges from -1 to +1, where avoidance was expressed by values between -1 and 0, and selection indicated by values between 0 and +1. We chose a cut off mark of density for the calculation of the index as it was found that low sample sizes increased the index value unrealistically. Hence the electivity index of only those species with overall density ≥ 1.6 trees ha^{-1} (≥ 5 individuals). Tree species diversity was calculated using Shannon-Wiener index of diversity (Krebs 1989).

3. Results

A total of 760 trees of 37 species were recorded in the total 100 plots sampled. Of these, 22 species were lopped in the ISA mostly as fodder for livestock. There was significant difference in severity of lopping in the four forest communities ($H = 29.8$, $P < 0.001$). The Miscellaneous forest community faced maximum percent severity (mean 14.6 ± 2.2 SE) and extent (mean 26.3 ± 6.7 SE) of lopping (Fig. 1). For calculating electivity index for the tree species in the ISA only 17 species were used for the analysis as five tree species had density below the minimum density of 1.6 individuals ha^{-1}. *Terminalia alata* Heyne ex Roth (Combretaceae), *Acacia catechu* (L.f.) Willd. (Mimosaceae) and *Anogeissus latifolia* Wall. ex Guill. & Perrotet (Combretaceae) were the most utilized species for lopping (Table 1).

They were also the most preferred species among the 10 most selected species by the *Gujjars* ($E > 0.94$; Gujjar pers. comm.) (Table 2). The first two species were in highest densities in Miscellaneous with Sal (7.64 trees ha^{-1}) and Miscellaneous community (8.91

trees ha^{-1}) respectively while the latter was in highest density in Sal mixed community (11.46 trees ha^{-1}). Seven of the 10 preferred species occurred in highest densities in Miscellaneous or Miscellaneous with Sal communities.

Table 1. The total extent of lopping of the preferred tree species in decreasing order and its respective densities in the Chilla range.

Species	Extent lopped (percent individuals) (%)	Tree density (ha^{-1})
Anogeissus latifolia	94.5	6.1
Terminalia alata	55.5	5.1
Acacia catechu	52.4	3.8
Terminalia belerica	33.3	3.0
Diospyros melanoxylon	33.3	3.8
Holoptelea integrifolia	26.7	5.1
Cordia dichotoma	25.0	6.1
Ziziphus xylopyra	20.0	3.4
Holarrhena pubescens	15.6	8.9
Cassia fistula	11.5	12.7
Mallotus philippensis	6.5	100.6
Ehretia laevis	1.6	34.1
Shorea robusta	1.2	30.9

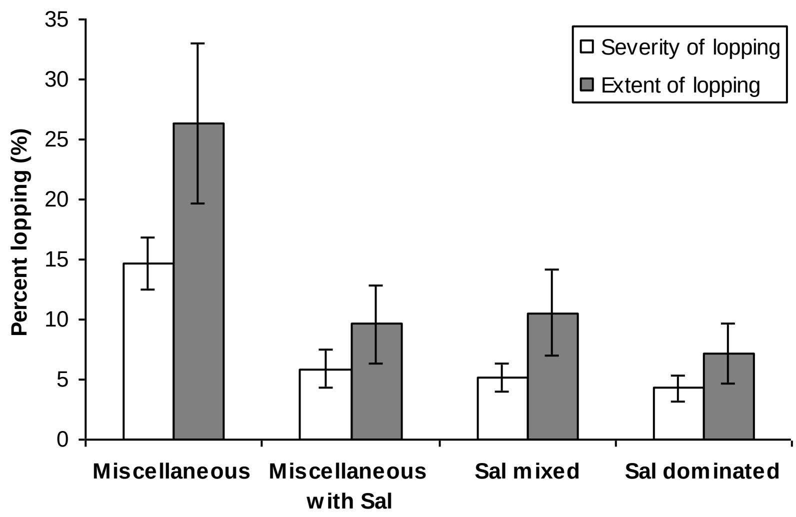

Figure 1. The severity and extent of lopping in the four different forest communities in Chilla range. The error bars represent standard error of the mean.

Table 2. Physiognomy of the four forest types present in Chilla range

	Miscellaneous	Misc. with Sal	Sal Mixed	Sal dominated
No. of Tree Species	28	19	22	12
No. of Shrub Species	20	22	26	15
No. of Herb Species	59	54	52	50
Trees				
- Mature (per ha)	222.9	179.6	243.3	324.8
- Saplings (per 100 sq.m)	22.6	24.9	42.5	73.4
- Seedlings (per 100 sq.m)	73.5	103.1	118.4	150.0
Shrub density (per ha)	3271.3	3307.0	4254.8	3755.4
Canopy (%)	51.0	41.1	58.0	67.5
Shannon-Wiener diversity index (*H*')	2.1	2.3	2.2	1.5

Among the 22 species that were lopped, 14 (64%) were present in the Miscellaneous forest type. Although Miscellaneous community had maximum number of species, it was seen that forest communities associated with species of the Miscellaneous community showed highest diversity, as indicated by the Shannon-Wiener diversity index (Table 2). Sal dominated community that was lowest in species richness and diversity (12 species) contained very high densities of relatively avoided and less selected species like *Shorea robusta*, *Ehretia laevis* Roxb. (Boraginaceae) and *Mallotus philippensis* (Lamk.)

Muell.-Arg. (Euphorbiaceae) (E < -0.6). Species like *Hymenodictyon excelsum*, *Lagerstroemia parviflora*, *Limonia acidissima* and *Syzygium cumini* were not lopped although they were found in adequate densities.

Table 3. Variation in severity of lopping and tree density for each species in decreasing order of electivity among the four forest community in Chilla sanctuary, Rajaji National Park; family is mentioned in parenthesis.

Species*	Electivity (E)	Miscellaneous (M)	Miscellaneous with Sal (MS)	Sal mixed (SM)	Sal dominated (SD)
		Severity of lopping (%) Density (ha^{-1})	Severity of lopping (%) Density (ha^{-1})	Severity of lopping (%) Density (ha^{-1})	Severity of lopping (%) Density (ha^{-1})
Terminalia alata (Combretaceae)	0.96	58.6 (3.8)	60.5 (7.6)	52.2 (3.8)	-
Acacia catechu (Mimosaceae)	0.95	35.9 (8.9)	69.2 (1.3)	0 (1.3)	-
Anogeissus latifolia (Combretaceae)	0.94	76.6 (5.1)	69 (6.4)	45.7 (11.4)	60.9 (1.3)
Ziziphus xylopyra (Rhamnaceae)	0.94	0 (1.3)	0 (2.5)	-	34.8 (6.4)
Holoptelea integrifolia (Ulmaceae)	0.9	43.1 (6.4)	0 (7.6)	-	0 (1.3)
Terminalia belerica (Combretaceae)	0.87	50 (1.3)	0 (6.4)	0 (1.3)	-
Diospyros melanoxylon (Ebenaceae)	0.79	0 (6.4)	0 (3.8)	18.2 (1.3)	-
Holarrhena pubescens (Apocynaceae)	0.46	37.5 (5.1)	0 (12.7)	2.5 (10.2)	0 (7.6)
Cassia fistula (Caesalpiniaceae)	0.38	6.3 (10.2)	0 (10.2)	2.4 (16.6)	13.6 (14.0)

Cordia dichotoma (Boraginaceae)	0.33	76.9 (1.3)	0 (14.0)	0 (5.1)	0 (3.8)
Ehretia laevis (Borginaceae)	-0.68	0 (20.4)	0 (25.5)	2 (59.9)	0 (30.6)
Mallotus philippensis (Euphorbiaceae)	-0.7	7.9 (114.6)	1.1 (58.6)	0.9 (54.8)	4.7 (174.5)
Shorea robusta (Dipterocarpaceae)	-0.73	0 (5.1)	0 (10.2)	0.9 (44.6)	1.1 (65)
Hymenodictyon excelsum (Rubiaceae)	-1	0 (2.5)	0 (2.5)	0 (2.5)	-
Lagerstroemia parviflora (Lythraceae)	-1	0 (1.3)	-	0 (6.4)	0 (16.6)
Limonia acidissima (Rutaceae)	-1	0 (7.6)	-	-	-
Syzygium cumini (Myrtaceae)	-1	-	0 (2.5)	0 (5.1)	0 (1.3)
Aegle marmelos (Rutaceae)	√	32.7 (3.8)	-	-	-
Albizia procera (Mimosaceae)	√	64.3 (1.3)	-	-	-
Emblica officinalis (Euphorbiaceae)	√	20 (2.5)	-	-	-
Ficus benghalensis (Moraceae)	√	9.1 (1.3)	-	-	-
Haldina cordifolia (Rubiaceae)	√	43.1 (3.8)	-	0 (1.3)	-
Bauhinia sp. (Caesalpiniaceae)	√	40 (1.3)	-	-	-
Mitragyna parviflora (Rubiaceae)	√	86.4 (1.3)	-	0 (1.3)	-
Ougeinia oojeinensis (Fabaceae)	√	60 (1.3)	-	16.7 (3.8)	-

Semecarpus anacardium (Anacardiaceae)	√	-	-	37.5 (1.3)	-
Terminalia chebula (Combretaceae)	√	50 (1.3)	-	-	-
Percent severity of lopping (mean ± se):		14.6 ± 2.2	5.9 ± 1.6	5.2 ± 1.2	4.2 ± 1.1
Total number of species:		28	19	22	12

* - Species nomenclature was based on Singh & Prakash (2004).
√ - Lopping observed; electivity not calculated as overall species density < 1.6 trees/ha
No lopping was observed on *Bauhinia purpurea* (MS), *Buchanania lanzan* (MS, SM), *Murraya koenigii* (M, MS), *Schleichera oleosa* (MS, SM), *Stereospermum suaveolens* (SD), *Tectona grandis* (M), *Garuga pinnata* (M), *Lannea coromandelica* (SM) and *Woodfordia fruticosa* (SM). Density < 1.6 trees ha^{-1} for all species.

4. Discussion

Our study indicates that of the four main forest types present in the study area, a disproportionate preference is given to largely one forest type. It is also noticeable that the pattern of lopping seen has a very selective basis to it, with some species highly preferred as against some others that are mostly avoided (Table 3). It can hence be deduced that the pattern of lopping experienced in each forest community is a result of the species composition of the communities. *Shorea robusta* (E = -0.739) although very high in density is not a preferred fodder species. This reiterates the findings of earlier studies from around this area (Edgaonkar 1995). Though there have been some debates in literature regarding the validity of selectivity indices, our choice of using Ivlev’s index was not an arbitrary one. In an important review of the sampling characteristics of electivity indices, Lechowicz (1982) noted that the Vanderploeg and Scavia forage ratio index stood out to be the most appropriate index to be used for analyzing electivities. However when used on our dataset, the Vanderploeg and Scavia index was not representing the lopping preference of species correctly. Its sensitivity to the number of species analyzed affected the symmetry of the index (Lechowicz 1982) and gave an ecologically incorrect representation of lopping preferences. Ivlev's index, E, was used to rank the species as it gave a more correct representation of preferences.

The Miscellaneous and Miscellaneous with Sal communities with higher tree diversity and more preferred palatable species were clearly lopped more (Table 2). It was found that forest types with higher densities of *Shorea robusta* were not lopped much. The reason for this trend is that Sal dominated forests have low species diversity and poor undergrowth (Champion & Seth 1968, Puri 1989). Also, forests with *Shorea robusta*, being commercially important underwent severe thinning in the form of climber cutting and removal of miscellaneous species until few decades back as a result of forestry management practices (Rawat & Bhainsora 1999). Although Sal is one of the dominant species in Siwaliks, some authors (Puri 1989) have expressed doubt regarding the widely held opinion that Sal is always a climax species or whether it has attained its present state as a result of human intervention. Though these are interesting conjectures the fact is that most areas of the Rajaji National Park and surrounding Siwalik region are Sal dominated in basic structure. It is of important conservation significance as the more diverse patches with species of the Miscellaneous forests are lower in density. With selective lopping for species of the Miscellaneous forest type, much of the remaining woody species diversity is at risk of loss. The dual effect of human overutilization of preferred species (Table 1) and the fact that most of the preferred palatable species occur in low densities forced the *Gujjars* to also lop the less palatable species.

Secondly, the lopped species apart from being most preferred by livestock, mainly buffaloes, are also favoured by wild herbivores like primates and ungulates (Bhatnagar

1991, Gupta 1991, pers. obs.) bringing them into direct competition for resource. Herbivore species are known to selectively forage for specific nutrients and plant proteins are among the most significant of them (Robbins 1993, Seagle & McNaughton 1992). Gupta (1991) in a forage value analysis of plants from an area adjoining the ISA assessed that leaves of highly lopped species like *Acacia catechu* is comparatively high in crude protein as against those of species like *Syzygium cumini* that had relatively low crude protein content and were consequently never utilized as fodder. Also, many of the edible fruit trees like *Terminalia belerica*, *Emblica officinalis*, *Ficus benghalensis*, *Aegle marmelos, Limonia acidissima* are species of the drier Miscellaneous forests. A faecal pellet count survey had shown that many wild herbivores also use the Miscellaneous forests to a greater extent as compared to the other three forest types (Kurien *et al.* unpubl. data).

We feel that this result can be suitably extrapolated to large areas in the Siwaliks with similar forest types and presence of *Gujjars*, as their land use patterns are very similar. Conservation measures can therefore be better focused if this disturbance pattern on the physiognomy of these forests is factored into the targets and strategies for wildlife management.

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Density of tiger and prey species in Chilla range, Rajaji National Park, Uttarakhand, India

Abishek Harihar, Bivash Pandav and S.P. Goyal.

Abstract**:** Tigers are large carnivores whose extinction risk is compounded by their need for large ranging areas and their dependence on ungulate prey species. Given that vast tracts of forests that once housed the tiger have now been lost to human habitation, obtaining reliable quantitative information on existing populations, opportunities for dispersal and connectivity between populations could aid metapopulation management thus lowering the risk of local extinction. The terai arc landscape along the foothills of the Himalayas stretches across India and Nepal and supports viable populations of tigers in small forest patches that have low connectivity. This study conducted over three years (2004-2007) in the Chilla range of the Rajaji National Park, Uttarakhand, India along the north western portion of this landscape documents the recovery of prey and tiger populations following the relocation of *Gujjar* (a pastoralist community with large buffalo holdings) settlements. Line transects (102.8 km of walk per survey year) in conjunction with distance sampling was used to estimate the density of the potential prey species while camera trapping (450 trap nights per survey year) in a capture-recapture framework was used to estimate the density of tigers. The intensive study area was found to support ungulate prey species in high individual densities (~66 individuals km^{-2}), with chital and sambar contributing up to 91%. Though density of prey species did not change, minimizing livestock grazing has probably led to a significant increase in the proportion of fawns among chital (ruminant grazers) across the survey years. Though the estimated density ($\hat{D}$) of tigers was low (3-5 individual tigers $100km^{-2}$), the study area could support as many as 13 individuals $100km^{-2}$. Evidences of breeding and a positive rate of change in population ($\bar{\lambda}$ = 1.14) coupled with the strategic location of the study area could help recover populations along the western range limit of tigers in the subcontinent. This study has shown that minimising livestock induced competition could help recover wild herbivore populations and clearly indicates a recovery of tiger population in a prey rich habitat following removal of human settlements.

Density of tiger and prey species in Chilla range, Rajaji National Park, Uttarakhand, India.

Abishek Harihar, Bivash Pandav and S.P. Goyal.

1. Introduction

Conserving large mammals in a human dominated landscape requires reliable quantitative information on existing populations, opportunities for dispersal and connectivity between populations. Using which, the effectiveness of management practices can be assessed and goals set for the future. In India conservation efforts such as Project Tiger have, since 1973, been attempting to save the nations declining populations of tiger, their prey and habitats, yet about 26% of their range has been lost in the recent past (Qureshi et al. 2006). With about 69% of India's protected areas being inhabited by people (Saloni 1996) and the recent crisis of vanishing tiger populations (Project Tiger 2005), the fact that most reserves are faced with severe anthropogenic pressures is increasingly becoming a cause of concern. While the ultimate threats to species survival are anthropogenic, intrinsic ecological and life history traits determine how well populations are able to recover (Cardillo et al. 2004). Tigers (*Panthera tigris*) are highly endangered large carnivores, whose extinction risk is compounded by their need for large ranging areas and their dependence on prey species (Carbone and Gittleman 2002, Karanth et al. 2004) that may themselves be threatened. Despite thirty years of continued conservation efforts, an expanding human population has caused considerable decline in the tiger's habitat, prey and the tiger itself in India (Seidensticker et al. 1999). Though illegal killing of tigers for body parts has contributed greatly to the extinction of local populations (Project Tiger 2005), vast tracts of forested landscape that once housed the tiger have now been lost to human habitation. This has caused a sharp decline in the ungulate populations and confined many of the remaining tiger populations to small, isolated patches of forests (Smith et al. 1998).

One such landscape, the Terai arc landscape, encompassing the Shivalik hills and the Terai flood plains running parallel to the outer Himalayas from Jammu through Nepal to Assam are considered one of the most threatened and fragile ecosystems in the Indian subcontinent. This productive landscape (Wikramanayake et al. 2004) is most prone to human disturbances (Johnsingh et al. 2004). With a human population density of over 500 people km^{-2}, this region is highly populous, surpassing the national average of 300 people km^{-2} (Johnsingh et al. 2004). Viable populations of tigers in this landscape exist only in small patches that have very low connectivity (Johnsingh et al. 2004, Wikramanayake et al. 2004). According to Johnsingh et al., (2004) the north-western portion of this landscape stretching from the Yamuna River in the west to the Sharda River bordering India and Nepal in the east, is fragmented into three distinct tiger habitat blocks (THB). This hilly (*bhabar*) tract covers nearly 6500km^2 that could potentially support a minimum 150 adult tigers, given adequate protection (Johnsingh 2006).

Gujjars, a pastoralist community inhabit many areas of the Shivaliks. With their large holdings of Buffalos (*Bubalis bubalis*), intensive grazing, lopping and firewood extraction, a lack of sustainable regeneration and proliferation in weeds (Edgaonkar 1995) had led to habitat degradation. Until recently they inhabited many parts of Rajaji National Park (RNP). Following a relocation program initiated by the Uttarakhand Forest Department in 2003 much of the occupied areas within RNP were made free of *gujjar* settlements and signs of recovery with respect to utilization of the area by wildlife were noticed. This study carried out since the winter of 2004 to early 2007 assesses the population density of prey and tigers following the relocation of the *gujjar* settlements from the Chilla range of RNP.

Given the nature and history of disturbance, we hypothesised that competition for space and shared resources could have affected wildlife populations. With studies indicating that wild herbivores in particular are adversely affected by populations of livestock (Prins 1992, Mishra and Rawat 1998, Prins 2000), management interventions targeted towards minimizing livestock grazing could help recover wild herbivore populations that have been suppressed by resource competition (Khan 1996, Madhusudan 2004), if left unmanaged it could trigger wild herbivore population declines. Karanth and Stith (1999) show that depletion of prey populations is a determinant of the viability of tiger populations and with recent studies suggesting that abundance of carnivores is closely related to biomass and densities of prey species (Carbone and Gittleman 2002, Karanth et al. 2004), recovery of even small populations would depend on minimizing anthropogenic pressures (Cardillo et al. 2004) given the high reproductive potential of tigers in prey-rich habitats (Karanth and Stith 1999).

2. Study area

This study was conducted from 2004 to 2007 following the relocation of *gujjar* settlements from the Chilla range (148 km^2) of RNP (820 km^2) along the eastern bank of the river Ganges, which forms the western limit of THB II (~3000 km^2; Johnsingh et al. 2004). Narrowly connected to THB I (~1800 km^2) through the Chilla-Motichur corridor (Johnsingh et al. 1990), Chilla also maintains connectivity with Corbett Tiger Reserve (CTR) through the Rajaji-Corbett corridor (Johnsingh and Negi 2003, Johnsingh et al. 2004). The range is characterised by rugged hills ranging from 400m to 1000m in altitude with steep southern slopes and is drained by rivers and streams running north to south, most of which remain dry in late winter and summer. Broadly, the forests of this region can be categorized as Northern Indian Moist Deciduous Forest and Northern Tropical Dry Deciduous Forest (Champion and Seth 1968), with the major associations being miscellaneous forests on the southern slopes and Sal (*Shorea robusta*) mixed and Sal dominated forests on the northern slopes, while the valleys have extensive grasslands. The large carnivores in the area are the tiger and the leopard (*Panthera pardus*). The potential prey species in the study area are sambar (*Cervus unicolor*), chital (*Axis axis*), barking deer (*Muntiacus muntjak*), nilgai (*Boselaphus tragocamelus*), wild pig (*Sus scrofa*), goral (*Nemorhaedus goral*), common langur (*Semnopithecus entellus*), Rhesus macaque (*Macaca mulatta*), porcupine (*Hystrix indica*), Hare (*Lepus nigricollis*) and Indian peafowl (*Pavo cristatus*). Domestic livestock (chiefly cattle and buffalo) found bordering the range are also potential prey species.

3. Methods

Field methods

Owing to logistic constraints and seasonal variations in prey and predator populations (A. Harihar, Wildlife Institute of India, unpublished data) field sampling was carried out only during the winters (December to February) of each of the survey years (2004-05 to 2006-07). Densities of the wild prey species were estimated using line transects in conjunction with conventional distance sampling (Anderson et al. 1979, Burnham et al. 1980, Buckland et al. 1993, 2001). A total of 9 line transects were permanently laid, with lengths varying from 0.91 km to 2.49 km in different parts of the study area covering all vegetation types (Figure 1). The total length of line transects were 12.85 km. Each line transect was walked 4 times each during every survey year, thus the total effort amounted to 102.8 km of walk per year. Line transect data was collected between 0615 hrs and 0930 hrs by two observers. On every walk we recorded, species, group size, age-sex composition, sighting angle measured using a hand held compass (KB 20, Suunto, Vantaa, Finland) and sighting distance measured by a laser range finder (Yardage Pro 400, Bushnell, Overland Park, Kansas USA). Age-sex compositions were recorded for sightings that permitted classification along the line transect surveys, individuals were classified into males, females and fawns.

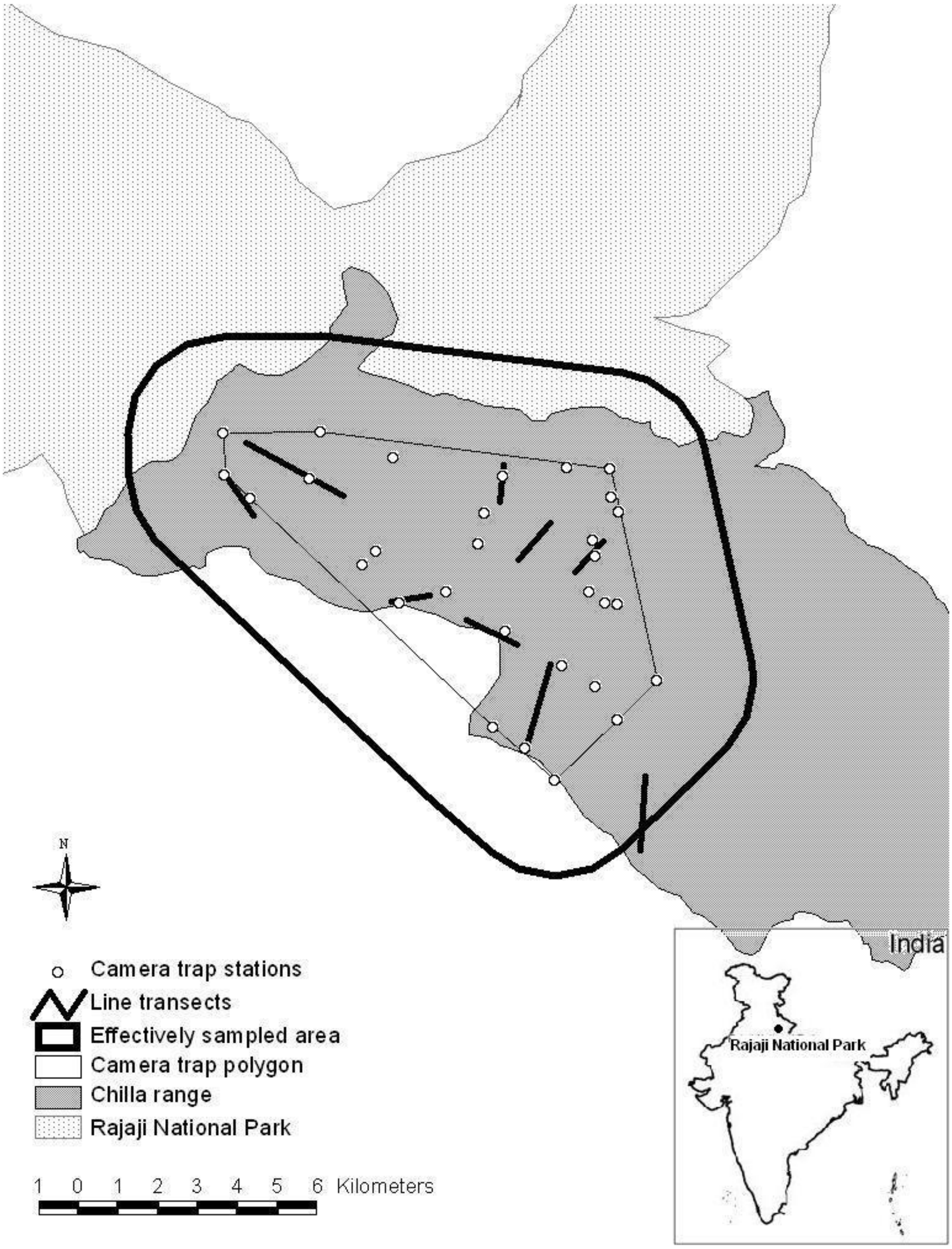

Figure 1. Map of Chilla range showing the camera trap locations, camera-trap polygon, the effective sampled area and the location of the nine line transects over the three survey years (2004-2007).

The population of tiger in the study area over the three years was estimated using photographic capture-recapture analysis (Karanth 1995, Karanth and Nichols 1998, Karanth et al., 2006). Thirty camera-trapping stations were identified (Figure 1), following a reconnaissance (November 2004). These trapping stations were selected based on the presence of secondary evidences that indicated the use of the area by tiger, therefore maximising capture probabilities (Karanth 1995). All trapping stations were maintained through the study period (2004-2007). In order to systematically sample the area, 3 sampling blocks (spatially separated) were identified within the intensive study area and the cameras were deployed in a phased manner. We had a total of 10 TRAILMASTER TM 1550 (Goodson and Associates, Kansas, USA) and 20 cameras, enabling us to photograph both flanks of the tiger at every capture. Each block consisted of 10 trap sites run for 15 consecutive days. Thus, each sampling occasion combined captures from 1 day drawn from each block. One trap-night was a 14-hour period (1700-0700 hrs) during which a camera was functional. The total effort amounted to 450 trap nights per survey year. Owing to a good network of roads all the 10 trapping sites in a block were checked on a daily basis. All rolls of film used during the trapping were given a unique identity (e.g., Block1/Trap1/Roll1) so as to correctly note the date, time and location of the captures. Every tiger captured was given a unique identification number (e.g., RT-002) after examining the stripe pattern on the flanks, limbs and forequarters (Schaller 1967, McDougal 1977, Karanth 1995).

Analytical methods

Population density of principal prey species was estimated using using program DISTANCE 5 Release 2 (Thomas et al. 2006). In order to model detection functions so as to estimate species density, the data for each species per survey year was examined for signs of evasive movement and peaking at great distance from the line of walk. Following this, suitable modifications were made so as to ensure a reliable fit of key functions and adjustment terms to the data in order to arrive at density estimates. Akaike Information Criterion (AIC) and goodness-of-fit (GOF-p) tests were used to judge the fit of the model. Using the selected model, estimates of group density (D_g), group size (GS) and individual density (D_i) were derived. To test for significance between age-sex ratios across the years, 95% bootstrap confidence intervals (Hilborn and Mangel 1997) were computed as the data did not conform to the assumptions of normality.

In order to estimate the population abundance of tigers capture histories (X matrix) were developed for each of the three survey years. Between the survey periods the population was open to gains and losses. However, within each of the survey years it was assumed that the population was both geographically and demographically closed. Since each primary period consisted of 15 trapping occasions (45 sampling days) it was logical to assume that the population was demographically closed, owing to the long life span of tigers, however geographic closure had to be ensured through trap placement. We tested for population closure using the closure test of program CAPTURE (Otis et al. 1978, Rexstad and Burnham 1991) and used the model selection procedure to test for variations in capture probabilities across occasions and estimated the population size ($\hat{N}$) of tiger for each primary period using the best selected model (Otis et al. 1978). The density ($\hat{D}$) of tiger per 100km^2 in the study area was estimated as the population size ($\hat{N}$) divided by the effective sampled area (A ($\hat{W}$)), where A ($\hat{W}$) was estimated by creating a polygon over the trapping stations (A) and a buffer width ($\hat{W}$) estimated as half the mean maximum distance moved (½MMDM) by recaptured tigers added to the camera trap polygon (A) (Karanth and Nichols 1998). Variance was computed following Karanth and Nichols (1998). We then estimated the finite rate of population growth ($\hat{\lambda}_t$) between two consecutive sampling periods as $\hat{\lambda}_t = \hat{N}_{t+1}/\hat{N}_t$, and computed the geometric mean annual rate of increase ($\bar{\lambda}$). Variance was estimated as in Seber (1982).

4. Results

Densities and age-sex composition of principal prey species

Densities were estimated for six of the 13 species that were detected on transects. Estimates of density could not be computed for the remaining seven species due to sample size constraints (Buckland et al. 1993). Though common langur, wild pig and nilgai had fewer than the recommended 60-80 detections, it conformed to the underlying assumptions of model fitting (Buckland et al. 1993, Laake et al. 1994). The estimates of group density (D_g), group size (GS) and individual density (D_i) of six principal prey species (sambar, chital, nilgai, wild pig, peafowl and langur) were derived; from the data we estimated a total prey species density of about 93 individuals km^{-2} (Table 1), with about 71% being contributed by wild ungulates (66 individuals km^{-2}). Of the groups 18.3% were of small bodied animals (peafowl and common langur, <20 kg); 40.6% were of medium sized animals (chital and wild pig, 20-50 kg) and 40.9% were of large bodied animals (sambar and nilgai, above 50 kg). Owing to sample size constraints, age-sex composition of only two major ungulate species (sambar and chital) were assessed over the years (Figure 2). Both species exhibited female biased sex ratios that remained stable across the years. While the proportion of fawns remained fairly stable for sambar across the years, the same increased over the three years among chital ($P < 0.05$).

Table 1. Prey species densities in Chilla range, RNP, India, from 2004-2007. D_g, estimated density of groups (number of groups km^{-2}) with associated standard error [SE]; GS, group size and associated standard error [SE]; D_i, density of individuals (number of individuals km^{-2}) with associated standard error [SE] and GOF-p, the probability of chi-square goodness of fit. Total effort of 102.8 km per survey year.

	2004-05				2005-06				2006-07			
	D_g[SE]	GS[SE]	D_i[SE]	GOF-p[df]	D_g[SE]	GS[SE]	D_i[SE]	GOF-p[df]	D_g[SE]	GS[SE]	D_i[SE]	GOF-p[df]
Sambar	10.5[1.8]	2.1[0.1]	21.3[4.1]	0.949[12]	9.8[1.8]	1.3[0.1]	13.1[2.5]	0.8941[6]	11.8[2.8]	1.2[0.1]	14.6[3.6]	0.981[14]
Chital	6.6[1.5]	6.2[0.7]	41.5[10.7]	0.9661[8]	9.6[2.6]	4.3[0.7]	41.6[13.6]	0.9426[9]	15.6[3.1]	3.1[0.5]	49.9[13]	0.967[14]
Nilgai	0.7[0.3]	2.4[0.7]	1.7[0.9]	0.9919[6]	1.9[0.9]	0.9[0.1]	1.9[0.9]	0.9729[4]	0.9[0.9]	2.5[0.5]	2.4[2.4]	0.9877[4]
Wild Pig	2.4[0.9]	3.3[0.8]	8.1[3.9]	0.9938[7]	0.3[0.2]	3[1]	1.1[0.9]	0.9823[3]	0.9[0.8]	2[1]	1.9[1.3]	0.8575[4]
Ungulate	**20.2**		**72.6**		**21.6**		**57.7**		**29.2**		**68.8**	
Langur	6.4[8.4]	3.9[0.7]	25.3[33.7]	0.9685[3]	1.7[0.7]	12.3[3.4]	21.4[10.6]	0.9861[3]	1.7[0.6]	8.1[1.1]	14.1[5.8]	0.991[12]
Peafowl	2.8[0.9]	4.2[0.8]	11.6[4.6]	0.9598[8]	0.6[0.3]	1.2[0.2]	0.8[0.5]	0.9907[3]	2.8[1.4]	2.3[0.6]	6.5[3.7]	0.8505[7]
Total	**29.4**		**109.5**		**23.9**		**79.9**		**33.7**		**89.4**	

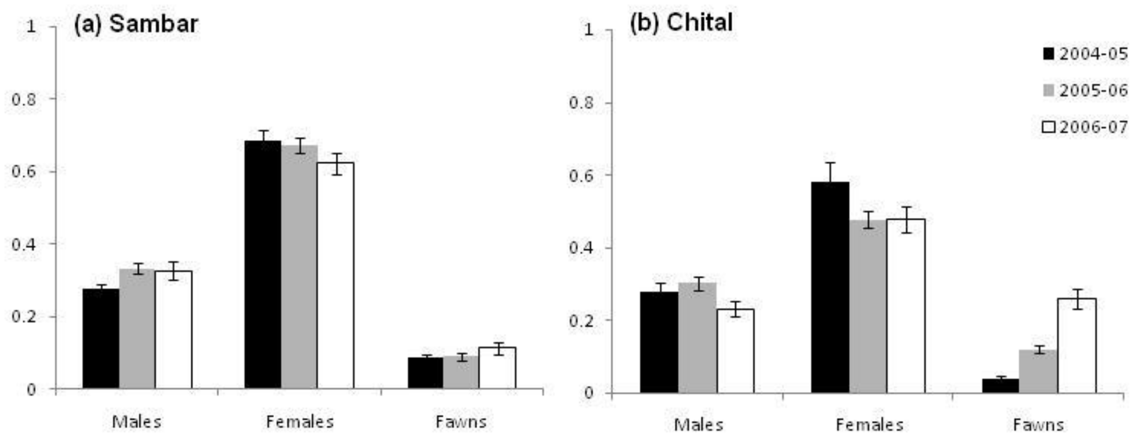

Figure 2. Composition (proportional) of adult males, adult females and fawns and associated 95% bootstrap confidence interval for (a) Sambar and (b) Chital across the three survey years (2004-2007).

Density of tigers in Chilla range

Following an effort of 1350 trap nights a total of 11 adult individual tigers were photographed across the three survey periods. The results of the closure test supported the assumption that the sampled population within each survey year were closed. And model selection procedures indicated no variation in capture probabilities for the first two survey years; however, evidence of individual heterogeneity was noticed during the third survey period (Table 2). The estimates of population size ($\hat{N}$) arrived at (Table 3) showed that the overall probability of capturing a tiger present in the sampled area ($M_{t+1}/\hat{N}$) was 100% for all three survey years. However to denote population trend, the annual rates of increase ($\hat{\lambda}_t$) between the three survey years were computed (Table 3), and the geometric mean growth rate was estimated $\bar{\lambda}$ = 1.14 ± 0.05 (mean ± standard error).

Table 2. Summary statistics for the test of population closure and model selection for photographic capture of tigers in Chilla range, RNP, India from 2004-2007. Total effort of 450 trap nights per survey year.

	Closure test		Model selection							
Survey year	*z*	*P*	M_o	M_h	M_b	M_{bh}	M_t	M_{th}	M_{tb}	M_{tbh}
2004-05	-0.39	0.348	1.00	0.97	0.38	0.59	0.00	0.30	0.47	0.61
2005-06	0.445	0.617	1.00	0.83	0.32	0.59	0.00	0.35	0.30	0.65
2006-07	-0.32	0.371	0.97	1.00	0.41	0.59	0.00	0.25	0.52	0.62

Table 3. Population of tigers in Chilla range, RNP, India, from 2004-2007. M_{t+1}, Number of individuals photographed; cumulative number of individuals across the three survey years; ½MMDM, half mean maximum distance moved by recaptured individual tigers, presented separately for the sexes and overall; $A(\hat{W})$, effectively sampled area; $\hat{N}$, estimated population size and the associated 95% Confidence Interval (CI); $\hat{D}$, estimated density of tigers and $\hat{\lambda}_t$, the annual population growth rate. Given in parenthesis [SE] are the associated standard errors.

		Cumulative No.	½MMDM [SE]							
Survey year	M_{t+1}	of individuals	Female	Male	Overall	$A(\hat{W})$[SE]	$\hat{N}$	CI	$\hat{D}$[SE]	$\hat{\lambda}_t$[SE]
2004-05	4	4	2.15[0.4]	-	2.15[0.4]	132.90[4.4]	4	4-5	3.01[0.7]	...
2005-06	5	8	2.35[0.5]	5.5[5.5]	3.38[1.1]	196.26[6.8]	5	5-6	2.54[0.8]	1.25[0.15]
2006-07	6	11	2.15[0.3]	1.6[0.4]	1.87[0.5]	116.96[5.3]	6	6-9	5.12[0.7]	1.2[0.12]

5. Discussion

Miquelle et al. (1999) and Karanth and Stith (1999) show that prey distribution and density determine first order and second order habitat selection by tiger respectively. The estimates of prey density arrived at in this study show that Chilla range of RNP harbours a high density of prey species (~93 individuals km^{-2}). Tigers being obligate carnivores primarily prey upon ungulates in all the ecosystems in which they occur (Seidensticker 1997). Although they can prey on a wide variety of species it has been noticed that the average prey size is around 60 kg. This is obtained predominantly from ungulate species, which contribute up to 75% of the prey biomass requirement of tiger (Sunquist et al. 1999). Chilla range supports a high density of ungulates (~66 individuals km^{-2}), primarily contributed by chital and sambar (91%). The estimates of individual density derived from this study (Table 1) shows that chital is the most abundant of all ungulate species followed by sambar. Of the animal groups encountered on transects 40.9% were of large prey (sambar and nilgai, above 50 kg) as compared to 40.5% in Ranthambore (Bagchi et al. 2003; sambar & nilgai), 8.5% in Pench Tiger Reserve (Biswas and Sankar 2002; sambar & nilgai) and 7.25% in Nagarhole (Karanth and Sunquist 1995; sambar, gaur *Bos gaurus* & elephant E*lephas maximus*). Recent studies scaling carnivores to the biomass of prey species (Carbone and Gittleman 2002) suggests that high prey biomass correlates to higher chances of persistence of predator populations. Karanth et al. (2004) show that tigers respond numerically to ungulate prey densities, supporting the general trend that higher prey densities correspond to increased tiger numbers. Though Chilla range supports a high ungulate prey density (66 individuals km^{-2}) the tiger density is low (3-5 tigers $100km^{-2}$) as opposed to a predicted 13.2 tigers $100km^{-2}$ given a 10% biomass off take (following the model developed by Karanth et al. 2004), suggesting that such areas will have to be protected so as to ensure the long-term survival of tigers.

Wild herbivores are particularly susceptible to competition with livestock (Sankar 1994, Mathai 1999), monitoring changes in their populations over time, have documented recoveries following the removal of human pressures (Khan 1996, Madhusudan 2004). Across the three survey years it was noticed that chital showed significant variations in proportion of fawns, whereas, sambar showed no significant response (Figure 2a). Chital being ruminant grazers like cattle and buffalo were probably more affected by the presence of livestock within the study area. Madhusudan (2004) documented spatial exclusion and recovery of ruminant grazers (chital and gaur) following a 49% decline of livestock populations, indicating that livestock did cause resource limitation on wild herbivores. Whereas sambar (ruminant forest browsers), showed no response to livestock presence or decline. Though we have limited data to address the issue of resource limitation and competition between ruminant grazers, we feel that the increasing proportion of fawns among chital (Figure 2b) is indicative of population recovery, suggesting improved recruitment and turnover among chital following the relocation of *gujjar* communities. Within Chilla range an overlap in habitat and dietary requirements between the wild herbivores and buffalos could have led to intense resource competition. Given that wild herbivores in the country are increasingly being confined to the protected areas which cover about 5% of land area (Madhusudan and Karanth 2002) and that most of these areas are faced with pressures of livestock grazing (Saloni 1996), recovery of wild ungulate populations following the minimizing of anthropogenic pressures is of conservation importance.

Density of tigers varied across the three survey years (Table 3). It was noticed that the ½MMDM by recaptured individuals varied greatly between the sexes and across the years, therefore influencing the estimates the effectively sampled area ($A(\hat{W})$). Since the estimate of population density ($\hat{D}$) is computed as the population size ($\hat{N}$) divided by the effective sampled area ($A(\hat{W})$), the estimates varied due to sampling uncertainties. Following Wilson and Anderson (1985), Karanth and Nichols (1998) propose the use of estimating a boundary strip width ($\widehat{W}$) based on the mean maximum distance moved (MMDM) between recaptures to calculate the effective sampling area ($A(\widehat{W})$). While it has been demonstrated that movement distances are effectively estimated when most animals are recaptured (Wilson and Anderson 1985), White et al. (1982) shows that using observed movement distances are more useful when animal ranges are small in comparison to the sampling grid. Soisalo and Cavalcanti (2006) recently demonstrated that estimates of MMDM computed from observed recaptures of Jaguars (*Panthera onca*) are under estimates of the true range use (verified using GPS-radiotelemetry), thereby considerably inflating density estimates. Given that movement is influenced by age, sex, reproductive status of tiger, the use of MMDM from camera trap data could be inappropriate while comparing changes in density. Since the trapping area (A) remained constant across the survey years, we compared the estimated population size (N).

This study documented an increase in adult tigers across the three survey periods (Table 3). Only one female was common to all three years while another female and a male were photographed consecutively over the last two survey periods. All other individuals were photographed only for one particular survey year, indicating that the individuals immigrated into the study area during that specific time period. However, photographic evidence (lactating females, females with cubs) is indicative of the fact that tiger populations too are responding to the removal of anthropogenic disturbance caused over many years prior to the study by the *gujjars* in Chilla range of RNP. As mentioned earlier the THB II, covering nearly 3000km^2 stretches from the east of Ganges to the west bank of the Gola river encompassing the Chilla and Ghauri ranges of RNP and CTR. However, these protected areas constitute only about 30% of this landscape. While the surrounding landscape matrix consists of multiple use forests, agricultural land and human habitation thus being less suitable for tiger persistence. Though, documenting the recovery of the tiger population can only be possible through long-term monitoring of the area, the geographic location of Chilla range in perspective of the THBs would help recover tiger populations west of the river Ganges (THB I), which is currently very low (A. Harihar, Wildlife Institute of India, unpublished data).

6. Management implications

This study clearly indicates that even a small population of tiger can recover in a prey rich habitat following removal of anthropogenic disturbance. This makes it imperative to protect breeding populations as source pools and provide dispersal opportunities in the context of managing meta populations. As has been noticed, livestock mediated resource competition probably affected chital populations, thus strongly emphasising the need to reduce such pressures to ensure the persistence wild herbivore populations. However, this area with its high-density prey population has the potential to support a higher density of tigers. As evidence of breeding tigers increase, Chilla range could act as a source population from where tigers can disperse and colonize forests along west bank of Ganges. Securing the connectivity of forest along east and west bank of Ganges through the Chilla-Motichur corridor is absolutely essential to ensure long term persistence of tiger in this landscape.

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Status of tiger and its prey species in Rajaji National Park

Abishek Harihar, Deepika L. Prasad, Chandan Ri, Bivash Pandav and S. P. Goyal

***Abstract*:** The Rajaji National Park (RNP) of 820km^2 forms the north western population limit of tigers in India and is bisected into two (west; 600km^2 and east; 220km^2 RNP) by the river Ganges. Following a gujjar relocation programme carried out in RNP by the Uttarakhand Forest Department, this study aimed to assess the status of tiger and its prey. We used sign surveys (80km) along dry streambeds to assess the distribution of tiger, leopard and their prey. Prey densities were estimated using line transects (100.5km) in conjuncture with distance sampling methods. The density of tiger was estimated using photographic capture-recapture analysis (900 trap nights) and we assessed the recovery of tigers across four years along east RNP using single season occupancy estimation. Our results indicate that the use of the area by tiger differed significantly across the two halves of the national park, as pug mark encounter rates per segment varied from 0.07±0.04 (west RNP) to 1.6±0.3 (east RNP). With estimated prey densities of 158 individuals km^{-2} along west RNP and 68 individuals km^{-2} along east RNP, we captured only one female along west RNP in contrast to six adult tigers and two cubs along east RNP. While recovery of the tiger population along east RNP is evident, the perceived recovery does not seem to have occurred along west RNP. Our results are indicative of an abysmally low tiger use of west RNP. We urge the concerned authorities to carry out a detailed investigation of the matter.

Status of tiger and its prey species in Rajaji National Park.

Abishek Harihar, Deepika L. Prasad, Chandan Ri, Bivash Pandav and S. P. Goyal

1. Introduction

Expanding human populations have in the recent past caused increased decline in the tiger's habitat, prey and the tiger itself (Seidensticker et al. 1999). In addition, the illegal killing of tigers for body parts has greatly accelerated the rates of local extinction (Project Tiger 2005). The terai arc landscape (Johnsingh *et al.,* 2004, Wikramanayake *et al.,* 2004) identified for the long term persistence of tiger (*Panthera tigris*) populations in the Indian subcontinent (TCU I; Wikramanayake *et al.,* 1998) is highly fragmented with viable populations of tigers occurring in forest patches surrounded by a matrix of multiple use forests, agricultural land and human habitation thus offering low connectivity. Bounded by the river Yamuna to west, the north-western portion of this landscape encompasses the Rajaji national park (RNP). With the forests to the west of Ganges (west RNP and other multiple use forests) extending up to the Yamuna river being narrowly connected to the forests on the east bank (east RNP) through the Chilla-Motichur corridor (Johnsingh *et al.,* 1990; Johnsingh & Negi 2003; Johnsingh *et al.,* 2004), RNP forms the population limit of breeding tigers in the Indian subcontinent. As populations at their range limits are more susceptible to local extinctions, ensuring the long term persistence of these wide ranging obligate carnivores would require reliable quantitative information on existing populations, opportunities for dispersal and connectivity between populations. An extensive survey (Johnsingh *et al.,* 2004) aimed to assess the status of tiger and associated species within this landscape indicated that tigers were particularly susceptible to anthropogenic disturbances.

Gujjars, a pastoralist community inhabiting many areas of this landscape, cause habitat degradation owing to their large holdings of Buffalos (*Bubalis bubalis*), intensive grazing, lopping and firewood extraction (Edgaonkar 1995). Following a voluntary relocation program facilitated by the Uttarakhand Forest Department in 2003 much of the *gujjar* inhabited areas within RNP were made free of human infringement and signs of increased animal use were noted (Pandav et al., 2004) within the Chilla range of RNP. Harihar (2005) and Kurien (2005) then established baseline information on prey and tiger densities and occurrence patterns of tigers and leopards in the Chilla range of RNP. Harihar et al., (2006) then documented changes in prey and tiger population and emphasized on the need to continually monitor this recovering population of tigers. This study thus carries forward the monitoring protocols established by Harihar (2005) and Kurien (2005) in Chilla range of RNP and extends the sampling to five other ranges (Chillawali, Dholkhand west, Dholkhand east, Hardwar and Ghauri) covering approximately 400 km^2 of the RNP to document population status of tiger and their prey.

In order to assess the status of tigers and their prey across the RNP the following three aspects were systematically monitored:

1. Occurrence patterns of tigers and leopards based on sign encounter surveys carried out along dry stream beds (*raus*).
2. Estimation of prey densities using line transects in conjunction with distance sampling methods
3. Estimation of tiger density using photographic capture-recapture sampling

And we also assessed the recovery of tigers along east RNP using previously collected data.

2. Methods

Investigation of occurrence patterns of Tiger and Leopard in Rajaji National Park

Based on an intensive survey undertaken for assessing tiger status in the bhabar tracts of Uttarakhand, Johnsingh et al., (2004) suggested that streambed transects (rau walks) are ideal for monitoring tiger pugmark occurrence. Streambeds or raus were therefore chosen as the line of walk for tiger sign surveys. Most of them inadvertently criss cross the pattern of habitats in the park. The objective of the streambed transects/walks was to generate an index that can represent the relative occurrence of tiger and its prey species. Indirect evidences in the form of pugmark, scats and scrapes etc give a good indication of the relative use by tiger of the study area. Since the parameter of interest was animal occurrence, all signs (tracks and pellets) were recorded. It is, by now, widely known that tigers use natural trails for their movement (Smith et al., 1998), and many studies use track plots for identifying tiger occurrence. Tiger occurrence can be estimated as an index that can indicate a rate of occurrence.

Nineteen streambeds ranging from 3.5km to 5km were chosen to conduct sign surveys (Figure 1). Of these eight were on the east of the Ganges (East bank of Ganges, Dogudda, Mundal, Ghasiram, Gara, Amgadi, Khara and Luni) and eleven were in the southern ranges of the park on the west of Ganges (Ranipur, Rawli, Harnol, Beribada, Sindhli, Baam, Malawali, Guleria, Binj, Gaj and Chillawali). Beginning and end points of these rau walks were marked using a GPS for future monitoring (Appendix I). For the analysis, transect "segments" comprised the basic sampling unit. Each segment was of 250m in length. Pugmark encounter rate was measured as the total number of pugmark encounters upon the total number of segments. The exercise was carried out with a team of 2-4 persons and on an average took 3-4 hours per transect (1.25-1.5 km/hr). Since the parameter of interest was relative occurrence in the area, and not abundance, pugmarks of tigers and leopards, whenever seen were identified and the continuity of the pugmark trails was used to identify them as separate encounters. Care was taken to avoid double counts of trails. Tracks of ungulates were recorded in each segment as low (<5 track trails), common (>5 track trails) and abundant (>10 track trails). Pellet groups of prey species and cattle and elephant dung depositions were also recorded.

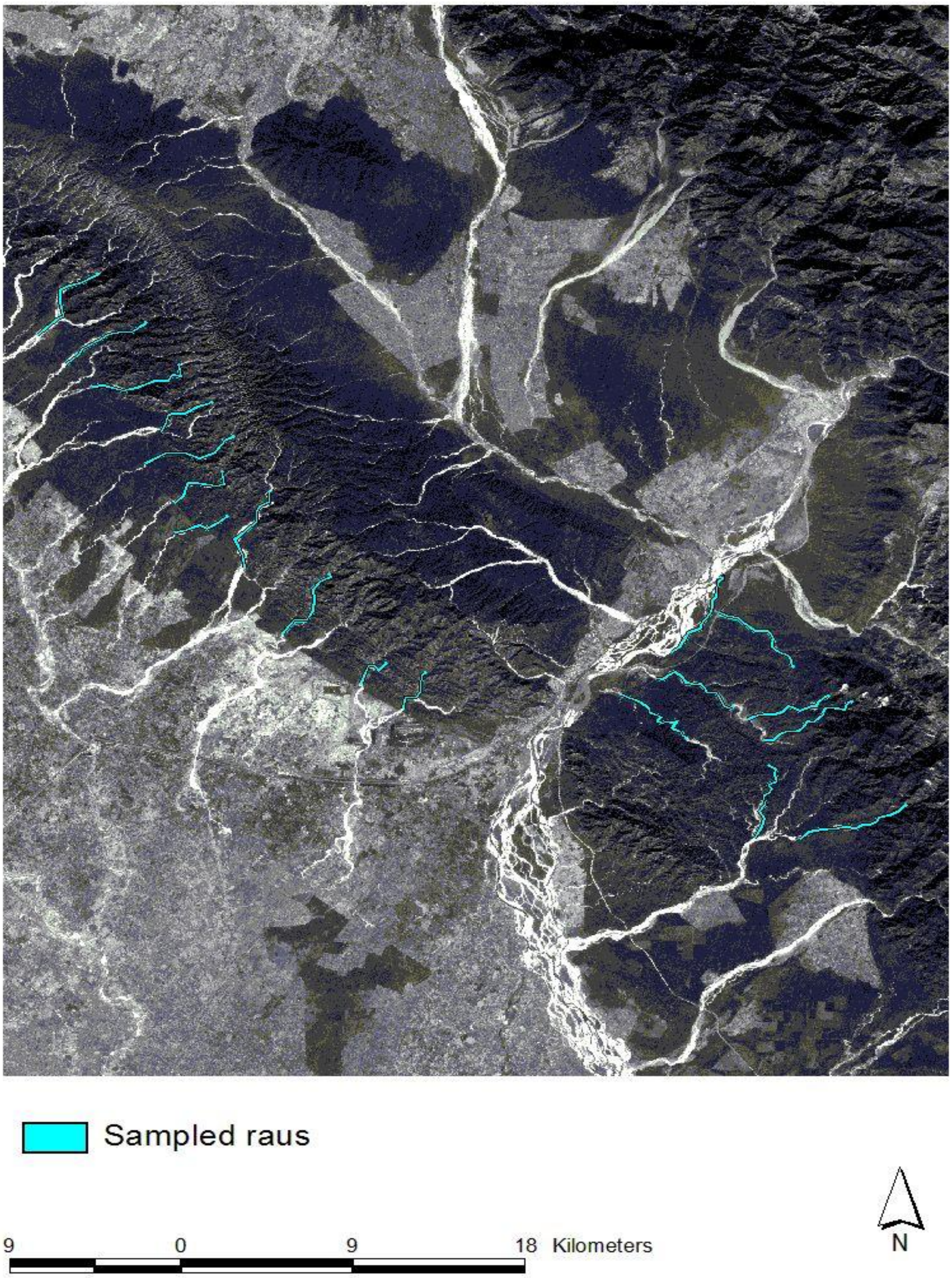

Figure 1. Location of nineteen streambeds (*raus*) sampled during December 2006.

Estimation of prey density in Rajaji National Park

Densities of the wild prey were estimated using line transects in conjunction with distance sampling (Anderson et al. 1979, Burnham et al. 1980, Buckland et al. 1993, 2001). A total of 24 line transects were laid (Figure 2), with lengths varying from 0.91km to 2.5km in different parts of the study area covering all vegetation types across the park. Coordinates of the start and end points of these transect were recorded using a GPS for future monitoring (Appendix II). The total length of line transects was 33.5km, with each line transect being walked 3 times each, the total effort amounted to 100.5km. Line transect data was collected between 0615 hrs and 0930 hrs by two observers. On every walk we recorded, species, group size, age-sex composition, sighting angle measured using a hand held compass (KB 20, Suunto, Vantaa, Finland) and sighting distance measured by a laser range finder (Yardage Pro 400, Bushnell, Overland Park, Kansas USA). Population density of principal prey species was estimated using using program DISTANCE 5 Release 2 (Thomas et al. 2006). In order to model detection functions so as to estimate species density, the data for each species per survey year was examined for signs of evasive movement and peaking at great distance from the line of walk. Following this, suitable modifications were made so as to ensure a reliable fit of key functions and adjustment terms to the data in order to arrive at density estimates. Akaike Information Criterion (AIC) and goodness-of-fit (GOF-p) tests were used to judge the fit of the model. Using the selected model, estimates of group density (D_g), group size (GS) and individual density (D_i) were derived.

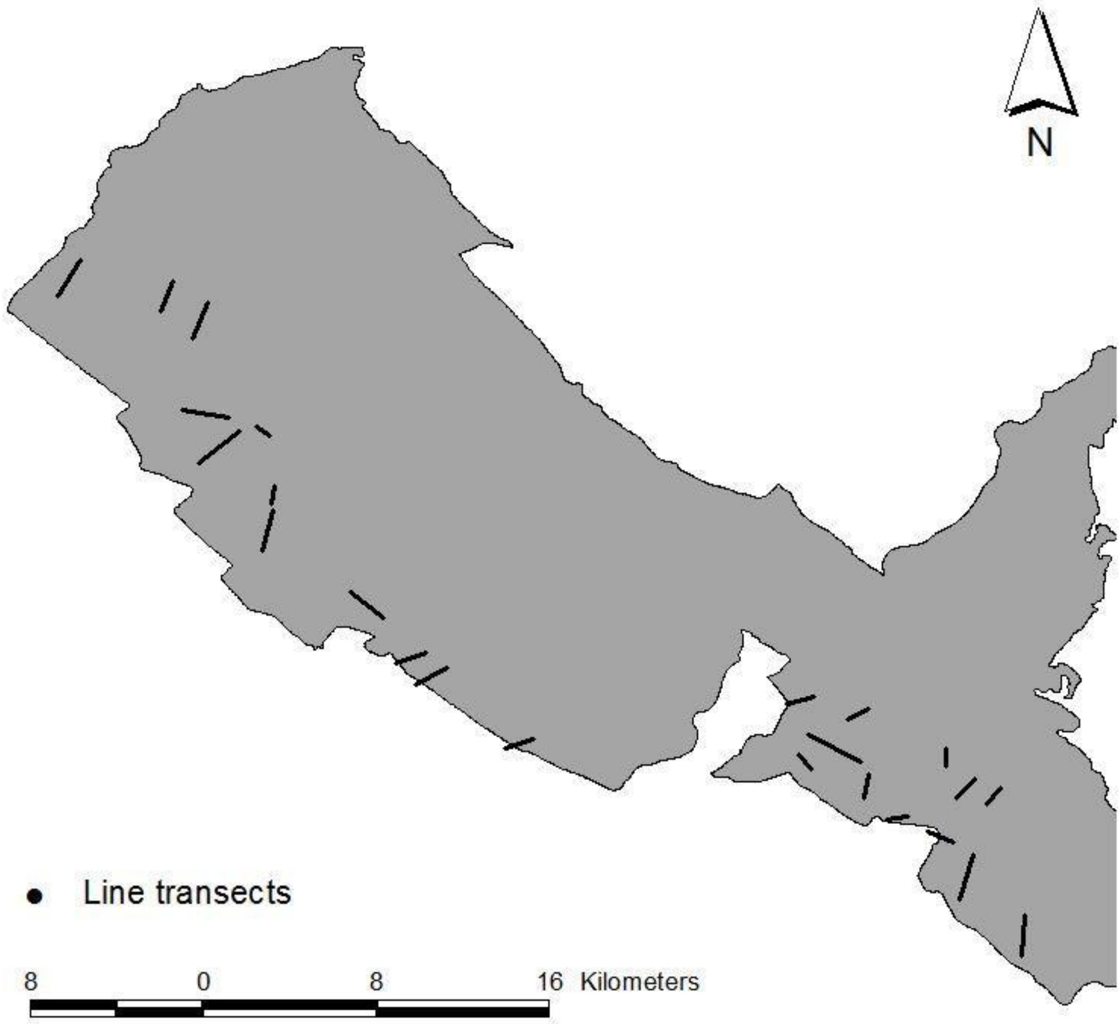

Figure 2. Location of 24 permanent transects laid in Rajaji national park.

Estimation of tiger density in Rajaji National Park

In order to estimate the population density of adult tigers in the study area we used photographic capture-recapture analysis (Karanth 1995, Karanth and Nichols 1998). We had a total of 20 STEALTHCAM IR1. Thirty camera-trapping stations were identified during 2004 in the Chilla range of RNP (Harihar 2005) and an additional 10 were included during the current exercise (Figure 3, Appendix III). An additional twenty camera-trapping stations were identified in the ranges of Dholkhand east and Dholkhand west during a reconnaissance carried out during December 2006 (Figure 4, Appendix IV). These trapping stations were selected so as to maximise the capture probabilities of tigers (Karanth 1995). In order to systematically sample the area, sampling blocks (spatially separated) were identified within the intensive study area and the cameras were deployed in a phased manner. Sampling along the east of Ganges (Chilla and parts of Gohri ranges) was carried out during January to February 2007 in 4 blocks and on the west of the Ganges (Dholkhand West and Dholkhand East ranges) in 2 blocks during March and April 2007. Each block consisted of 10 trap sites run for 15 consecutive days/occasions. Thus, each sampling occasion combined captures from 1 day drawn from each block. One trap-night was a 14-hour period (1700-0700 hrs) during which a camera was functional. Owing to a good network of roads all the 10 trapping sites in a block was checked on a daily basis. All photographs were downloaded at the trap site using a laptop. Every tiger captured was given a unique identification number (e.g., RT-002) after examining the stripe pattern on the flanks, limbs and forequarters (Schaller 1967, McDougal 1977, Karanth 1995). Following the identification of tigers, capture histories (*X* matrix) were developed and analysed using program CAPTURE (Otis et al. 1978, White et al. 1982, Rexstad and Burnham 1991). It was assumed that the sampled population was demographically and geographically closed (Otis et al. 1978, Karanth 1995, Karanth and Nichols 1998) during the short sampling period (45 days). The density (*D*) of tigers in the study area was estimated as the population size (*N*) divided by the effective sampled area (*A(W)*), where *A(W)* was estimated by creating a polygon over the trapping stations (*A*) and a buffer width (*W*) estimated as half the mean maximum distance moved (½ MMDM) by recaptured tigers added to the camera trap polygon (*A*) (Karanth and Nichols 1998).

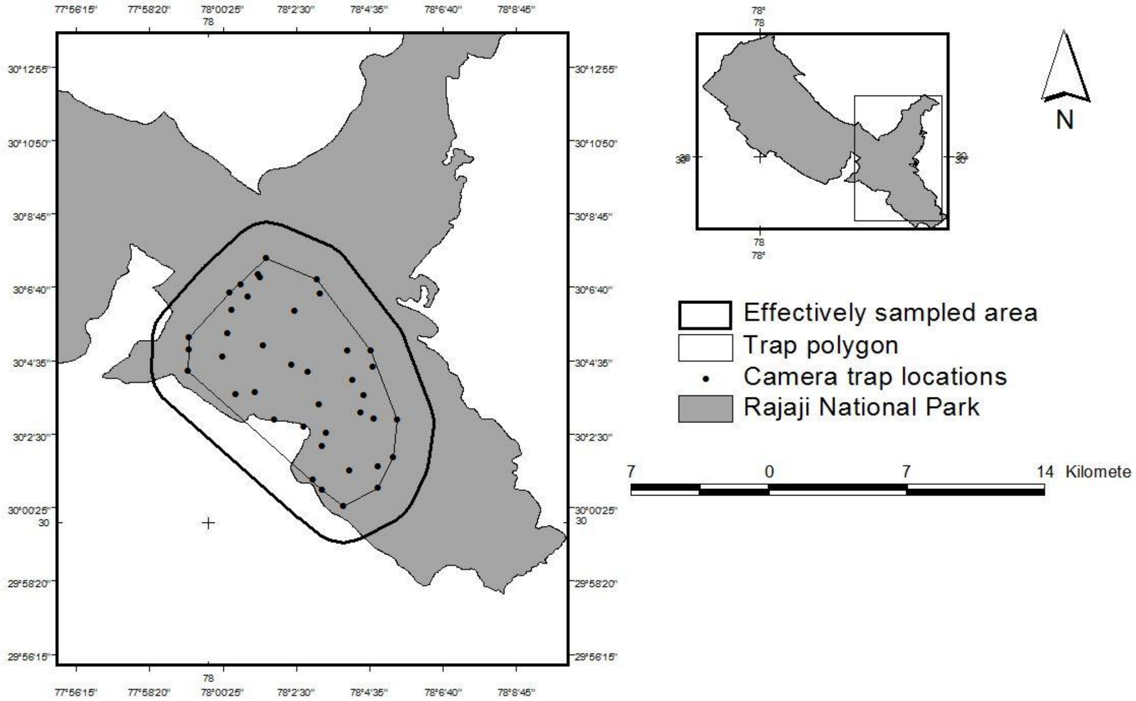

Figure 3. . Location of the camera traps, camera-trap polygon and the effective sampled area in east RNP during January to February 2007.

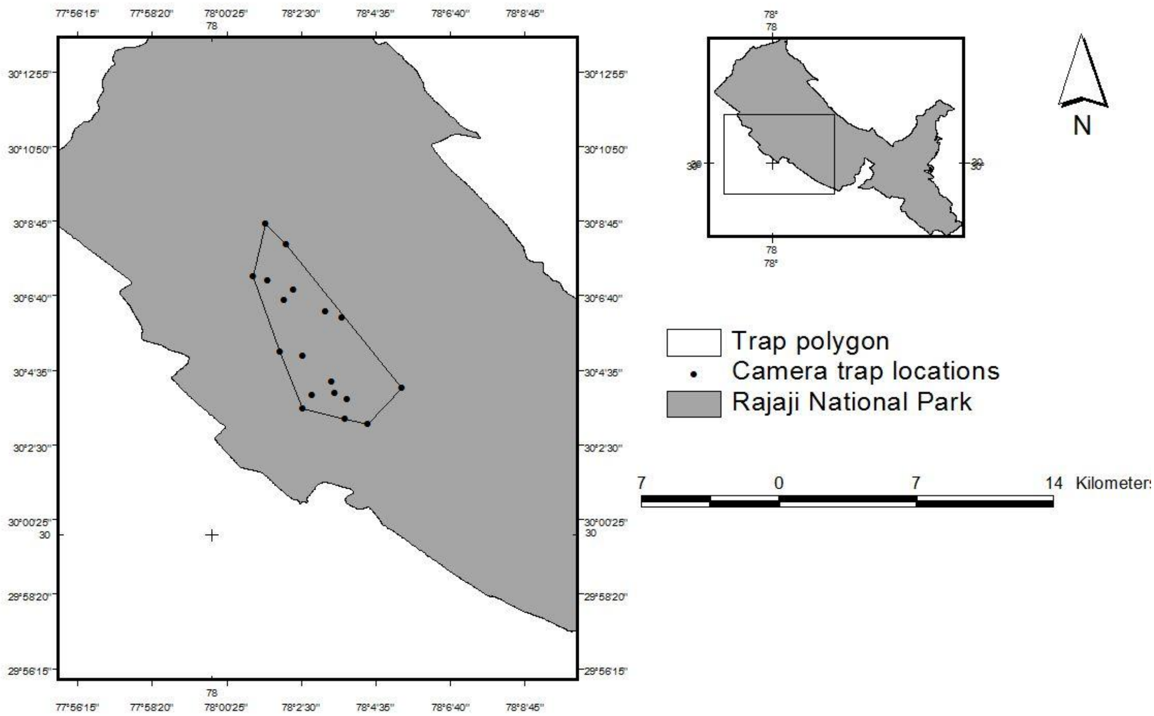

Figure 4. Location of the camera traps and camera-trap polygon in west RNP during March to April 2007.

Assessing the recovery of tigers in Chilla range of RNP using presence absence data

In order to assess the recovery of tigers in the Chilla range of RNP since 2002, detection-nondetection data generated through rau walks over four sampling years were compiled (2002-03 - Johnsingh et al. 2004; 2003-04 - B. Pandav, Wildlife Institute of India, unpublished data; 2004-05 - Kurien 2005; and 2006-07 - this data). The data constituted 10 raus and four sampling occasions typically representing a multi season estimation of occupancy, however we used single season occupancy estimation model of program PRESENCE 2 (Hines 2006), since all sign surveys were only conducted once in a survey year, we were unable to model Ψ and p separately for each year.

3. Results

Tiger and Leopard occurrence patterns in parts of Rajaji National Park

Among the 19 streambeds sampled, Luni, Khara, Amgadi, Gara, Mundal, Ghasiram, Dogudda and Ganga were in the ranges of Chilla and Gohri (East of the river Ganges) and resulted in pugmark encounters of both tigers and leopards. However in West RNP (West of the river Ganges), Beribada, Sindhli, Malawali and Guleria resulted in pugmark encounters of both tigers and leopards. As hypothesized, areas of human occupancy (Baam, Binj, Gaj and Chillawali) resulted in no pugmark encounters of tigers. Though free of human infringement, only pugmarks of leopard were encountered in Ranipur range (Ranipur, Rauli and Harnol; Figure 5a & b).

Hierarchical cluster analysis for prey and predator track count data using squared Euclidean distance resulted in two very distinct clusters (Figure 6). One, the streambeds on the east of Ganges (Chilla and Gohri ranges) and the other, west of the Ganges (Ranipur, Dholkhand west, Dholkhand East and Chillawali). Among the streambeds of the west of the Ganges two further classes are identifiable, primarily being governed by patterns of human occupancy (Chillawali, Baam, Gaj and Binj; *Gujjar* occupied areas and Ranipur, Rauli, Harnol, Beribada, Sindhli, Malawali and Guleria; *Gujjar* evacuated areas). These results (Figure 6) indicate that human occupancy influence animal movement, in particular site usage by tiger. Given the fact that *Gujjars* were evacuated from many parts of West RNP during the year 2005-06, we noticed an increased movement of tiger along the stream beds in west RNP. During a rau walk exercise carried out by the officer trainees of XXVI PG Diploma course of WII during second and third week of November 2005, tiger pugmarks were encountered in 13 of the 15 raus surveyed between Mohand and Ranipur (Andheri, Binj, Dholkhand, Guleria, Sampowali, Malawali, Bam, Beribada, Gholna, Harnol, Chidak, Rawli and Ranipur – S. P. Goyal & A. J. T. Johnsingh, WII, unpublished data). However, our results from the present survey indicate a considerable reduction in movement of tiger in the same area. Of the 11 raus (40km) sampled in west RNP, only four recorded evidence of tiger. In contrary, leopard evidences were recorded on all raus. It is of importance to note that the presence of cattle and humans were also found in all the raus sampled (Table 1). While in East RNP, all eight raus (40km) sampled recorded evidence of tiger and leopard. Evidences of cattle and human were found on all raus except Gara and Amgadi (Table 1).

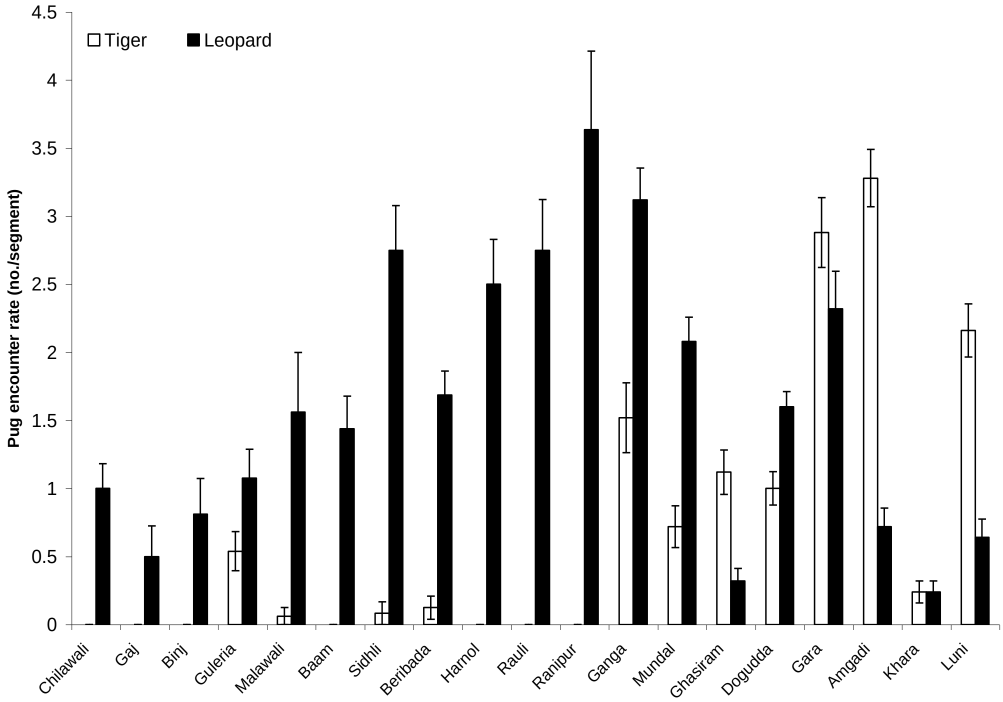

Figure 5a. Pugmark encounter rates of tigers and leopards in each streambed across the Rajaji National Park during December 2006.

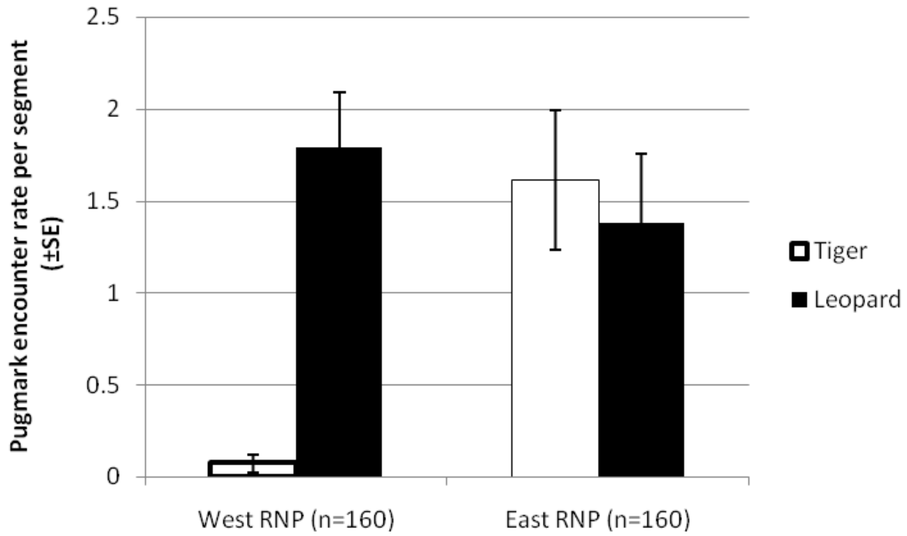

Figure 5b. Mean pugmark encounter rates of tigers and leopards across the Rajaji National Park during December 2006.

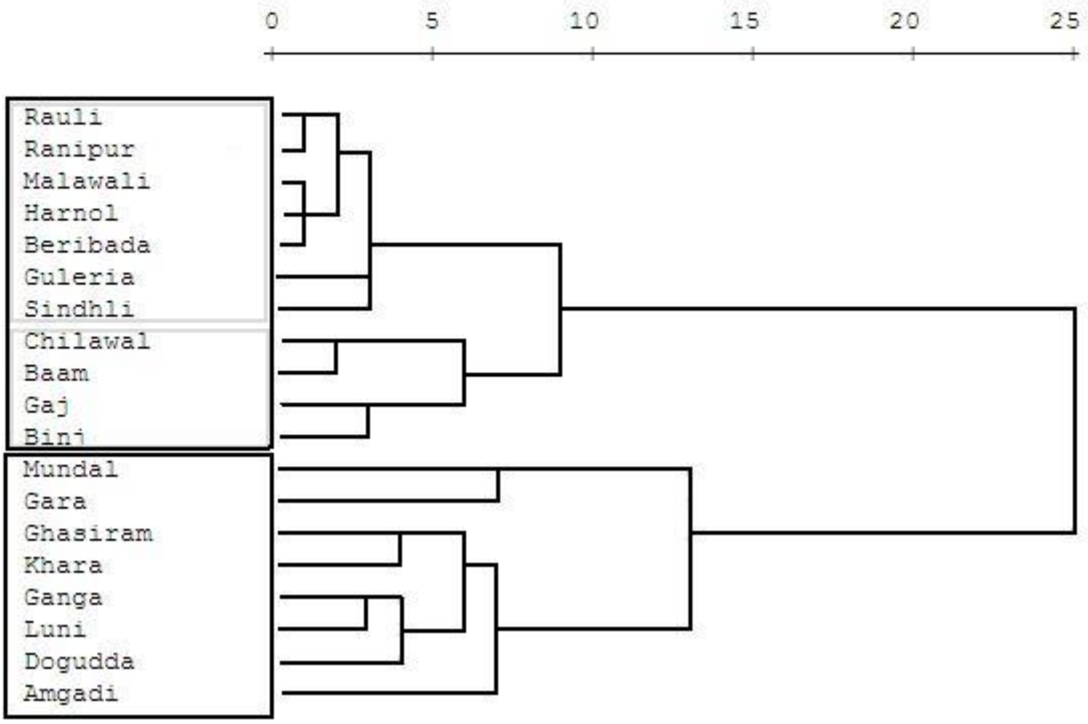

Figure 6. Dendrogram showing similarity between streambeds with respect to prey and predator movement across the Rajaji National Park during the December 2006.

Table 1. Transect wise compilation of animal evidence in the Rajaji National Park, during December 2006.

Strata	Range	Transect name	Effort (km)	Pugmark encounter rate (no./segment)		Frequency of occurance (%)							
				Tiger	*Leopard*	*Tiger*	*Leopard*	*Sambar*	*Chital*	*Nilgai*	*Wildpig*	*Cattle*	*Human*
West RNP	Chillawali	Chilawali	4	0	1.00	0	75	100	81	62	75	100	100
	Chillawali	Gaj	4	0	0.50	0	31	100	63	56	81	100	88
	Chillawali	Binj	4	0	0.81	0	44	100	100	50	63	100	69
	Dholkhand west	Guleria	3.25	0.54	1.08	54	85	100	100	54	62	0	38
	Dholkhand west	Malawali	4	0.06	1.56	6	81	100	100	44	81	88	100
	Dholkhand east	Baam	4	0	1.44	0	81	100	94	38	6	100	69
	Dholkhand east	Sidhli	3	0.08	2.75	8	100	100	100	100	83	67	50
	Dholkhand east	Beribada	4	0.13	1.69	13	94	100	100	100	50	50	31
	Ranipur	Harnol	4	0	2.50	0	94	100	100	81	100	75	75
	Ranipur	Rauli	3	0	2.75	0	100	100	100	100	67	25	25
	Ranipur	Ranipur	2.75	0	3.64	0	91	100	91	91	82	27	27
East RNP	Ghauri	Dogudda	5	1.52	3.12	20	40	95	100	5	45	75	100
	Chilla & Ghauri	Ganga	5	0.72	2.08	50	95	100	100	10	45	90	90
	Chilla	Mundal	5	1.12	0.32	35	85	100	100	40	20	35	20
	Chilla	Ghasiram	5	1.00	1.60	55	20	100	100	10	100	100	20
	Chilla	Gara	5	2.88	2.32	80	80	100	100	0	0	0	0
	Chilla	Amgadi	5	3.28	0.72	95	40	100	100	0	0	0	0
	Chilla	Khara	5	0.24	0.24	15	15	100	100	45	0	95	15
	Chilla	Luni	5	2.16	0.64	85	35	100	100	60	75	95	35

Prey densities in Rajaji National Park

Prey densities in East RNP

Densities were estimated for six (sambar, chital, nilgai, wildpig, langur and peafowl) of the 13 species that were detected on transects (Table 2). Estimates of prey density derived here are high (84.17 individuals km^{-2}), with the estimated ungulate density being 68.98 individuals km^{-2} contributing 81% of the total density. In terms of densities of groups, chital was the most abundant followed by sambar, peafowl, common langur, nilgai and wildpig. Chital was found to be the most abundant in terms of individual densities, followed by sambar, common langur, peafowl, nilgai and wild pig. The estimated ungulate wild prey biomass density was 4817.31 kg km^{-2}. Of the groups, 13.5% were of small bodied animals (peafowl and common langur, <20 kg); 48.7% were of medium sized animals (chital and wild pig, 20-50 kg) and 37.8% were of large bodied animals (sambar and nilgai, above 50 kg).

Prey densities in West RNP

Densities were estimated for six (sambar, chital, nilgai, wildpig, langur and peafowl) of the 11 species that were detected on transects (Table 2). Estimates of prey density derived here are high (158.3 individuals km^{-2}), with the estimated ungulate density being 110.01 individuals km^{-2} contributing 69% of the total density. In terms of densities of groups sambar was the most abundant followed by chital, nilgai, peafowl, common langur and wild pig. Chital was found to be the most abundant in terms of individual densities, followed by common langur, sambar, nilgai, peafowl and wild pig. The estimated ungulate wild prey biomass density was 9579.05 kg km^{-2}. Of the groups, 6.9% were of small bodied animals (peafowl and common langur, <20 kg); 36.4 % were of medium sized animals (chital and wild pig, 20-50 kg) and 56.8% were of large bodied animals (sambar and nilgai, above 50 kg).

Table 2. Prey species densities, body weight (kg) and biomass density (kg km^{-2}) in Rajaji National Park (Winter 2006-07). *n*, total number of groups detected; D_g, density of groups (number of groups km^{-2}); GS±SE, group size and associated standard error; D_i, density of individuals (number of individuals km^{-2}). Total effort, 100.5km.

	Body	***East RNP***					***West RNP***				
Species	**weight***	***n***	D_g	**GS±SE**	D_i	**Biomass**	***n***	D_g	**GS±SE**	D_i	**Biomass**
Sambar	134	47	11.89	1.2±0.08	14.67	1965.78	83	20.115	1.2±0.05	26.142	3503.03
Chital	47	42	15.65	3.1±0.53	49.904	2345.49	70	15.655	4.1±0.4	65.247	3066.61
Nilgai	181	2	0.97977	2.5±0.5	2.4494	443.341	37	4.7475	3.4±0.53	16.197	2931.66
Wildpig	32	2	0.97977	2±1	1.9595	62.704	3	0.27	9±5.6	2.43	77.76
Ungulates	**-**	**-**	**29.4995**	**-**	**68.9829**	**4817.31**	**-**	**40.7875**	**-**	**110.016**	**9579.05**
Langur	9	8	2.17	4.8±1.15	10.58	95.22	4	1.1	39.7±2.50	43.5	391.5
Peafowl	4	9	2.44	1.8±0.45	4.61	18.44	7	1.92	2.5±0.52	4.79	19.18
Total	**-**	**-**	**34.1095**	**-**	**84.1729**	**4930.97**	**-**	**43.8075**	**-**	**158.306**	**9989.73**

* - Schaller (1967) and Karanth and Sunquist (1992)

Density of tigers in Rajaji National Park

Tiger densities in East RNP

The total sampling effort amounted to 600 trap nights. The intensive trapping involving 40 camera traps and 15 nights of trapping at each station resulted in a total of 14 photographs of five adult tigers (Appendix VII). This excludes the two cubs photographed in Sarkada rau of Luni block, Chilla range, during the sampling period. The statistical test for population closure in CAPTURE (Otis et al. 1978, Rexstad and Burnham 1991) supported the assumption that the sampled population was closed for the 45-day study interval (z = -0.390, P = 0.34809). The model selected by program CAPTURE was M_h, the capture probability (p) was estimated at 0.1556 and the estimated population size N(SE[N]) was 6(2.51). The camera trap polygon (A) formed using periphery camera traps measured 52.65 km^2. The boundary strip width W(SE[W]) was estimated at 1.87(0.5) km and the effective sampled area $A(W)$ (SE[$A(W)$]) was 116.96(5.3) km^2. Thus, the estimated tiger density D(SE[D]) for Chilla range of RNP was 5.12(0.7) tiger per 100 km^2.

Tiger densities in West RNP

The intensive trapping involving 20 camera traps covering 42.8 sq km spanning 15 nights of trapping at each station amounted to 300 trap nights and resulted only a photograph of one individual tiger. Therefore no statistical analysis could be performed.

Assessing the recovery of tigers in Chilla range of RNP

Following the estimation of Ψ and p based on the single season occupancy model (Hines 2006), a constant occupancy (Ψ(.)) with survey specific detection probability (p(t)) was selected as the best model based on AIC scores. The model generated estimates of survey specific detection probability (p) that indicated an increase in detecting tiger evidences across the four survey years (Figure 7), thus indicating that the population of tiger in Chilla range along east RNP were recovering following the relocation of *gujjar* settlements.

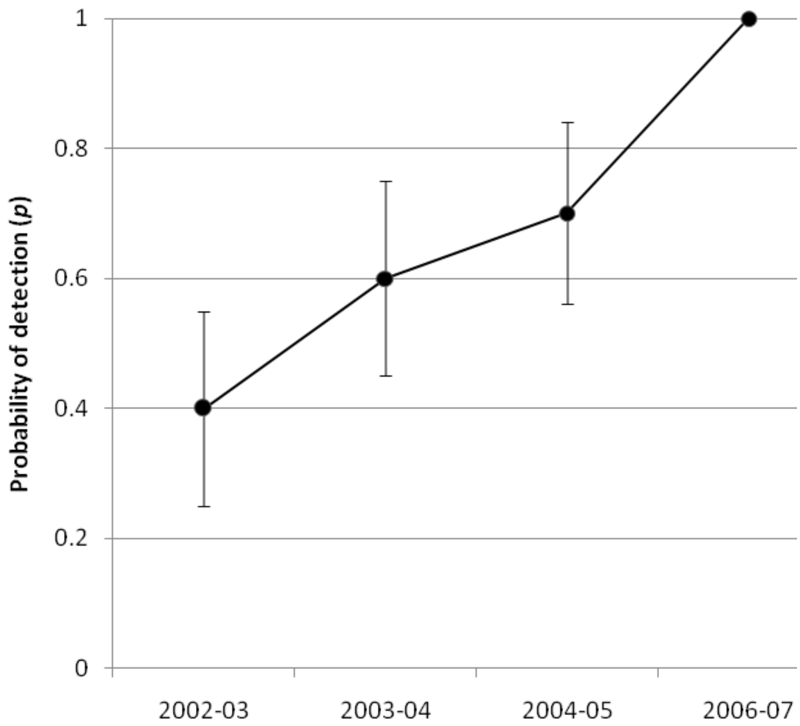

Figure7. Estimates of the probability of detection across four survey years in Chilla range of RNP, indicating an increased use of the sampled raus by tigers. Error bars indicate one standard error.

4. Discussion

Following the range wise relocation programme intiated by the Uttarakhand Forest Department in 2003, Chilla range of RNP has provided us an opportunity to document changes in prey-predator populations following relocation of human settlements (Pandav et al. 2004; Harihar 2005; Kurien 2005; Harihar et al. 2006). The area with its high-density prey population and breeding tigers has the potential to support a higher density of tigers (Harihar 2005; Harihar et al. 2006). Though long term monitoring of prey and predators will help understand their population dynamics (Karanth et al. 2006) and response to management interventions, we feel that this critical information should be available to managers so as to ensure the recovery and long-term persistence of tigers within this landscape. From the present study (Table 1) it is evident that the distribution of major prey species (Chital and

Sambar) across RNP is similar. However, it is of importance to note that there exists a difference in sign encounter rates of tigers, leopards and prey species across the RNP. The results of the hierarchical cluster analysis indicate a difference in patterns of animal movement. Two distinct clusters emerge (Figure 6); one is that of the streambeds in East RNP and the other is that of West RNP. While within the West RNP cluster there exists a distinction between areas still under the influence of the *Gujjars* (Chillawali, Binj, Gaj and Baam) and those evacuated of *Gujjars* (Hardwar, Rawli, Harnol, Beribada, Sindhli, Malawali and Guleria), indicating that areas of human influence adversely affect movement patterns of prey and predator populations. An assessment of the pugmark encounter rates (Figure 5a & b) of tigers and leopards across the streambeds in RNP shows a marked difference in sign encounter rates of tigers in West RNP and East RNP, with East RNP recording at least one pugmark set per segment surveyed as compared to West RNP which recorded a mean pugmark encounter rate of 0.07 per segment surveyed. However, there was no significant difference among sign encounter rates of leopards (Figure 5b) across the RNP. The estimates of ungulate prey density arrived at in this study show that East RNP and West RNP harbour high density of prey species (68.98 individuals km^{-2}; East RNP and 110.01 individuals km^{-2}; West RNP). While Harihar (2005) and Harihar et al., (2006) show that East RNP harbours high prey densities and predict higher tiger densities following the minimising of anthropogenic influence, we feel the same is applicable to West RNP. Though many parts of West RNP have been freed of anthropogenic influence (*Gujjar* influence) since the year 2005, the perceived recovery does not seem to have occurred. Based on the results presented in Harihar et al., (2006) and photographic captures acquired during this study it is evident that the population of tigers in East RNP seems to be on the increase. Photographic evidences of lactating tigress and two different tigress with two and three cubs each captured during

February and April 2007 respectively strengthens the fact that tiger numbers are on the rise in East RNP.

In contrast, neither an expected increase in tiger number nor an increased use of the area by tiger following relocation of *gujjar* settlements in West RNP is occurring. Only one female tiger was photographed following 300 trap nights of effort. Unpublished camera trapping data (S.P. Goyal & A. J. T. Johnsigh, WII, Unpublished data) from Dholkhand west reveals a total of 8 individual tigers used the area during 1996 to 2001 (Appendix VI) indicating fairly good presence of tigers, though the results are incomparable owing to differential efforts, it is to be noted that traps, though placed in the same location did not yield captures of previously identified individuals. The results presented here indicate a marked difference in animal movement across the two halves of the Rajaji National Park.

Despite having a high prey density, tiger use of West RNP at present is abysmally low. As mentioned earlier in this report, during November 2005 we did notice very good tiger use in most parts of west RNP (in miscellaneous forests along the southern slopes of Shivaliks) following relocation of gujjar settlements in Hardwar, Dholkhand east and Dholkhand east ranges. However, this sudden decline in use of area by tiger as evident in this present study is a matter of utmost concern. Although, we have not come across any evidences of tiger poaching, considering the proximity of park boundary to human habitation, possibility of tiger poaching cannot be ruled out. Things are certainly not going in the right direction for tigers in the western part of RNP. This matter needs detailed investigation.

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Separation between two sympatric carnivores in Chilla range of Rajaji National Park

Abishek Harihar, Chandan Ri, Bivash Pandav and S. P. Goyal

***Abstract*:** Prey selection by tiger *Panthera tigris* and leopard *P.pardus* were assessed in the Chilla range of Rajaji National Park following a human resettlement programme that has facilitated the recovery of wild herbivore populations. Densities of wild prey species were estimated using line transects in conjunction with distance sampling methods, while dietary profiles of the two carnivores were assessed based on identifying prey remains from field collected scats. We assessed the dietary niche overlap using the Pianka's overlap measure, selectivity towards individual prey by constructing 95% Bonferronis' simultaneous confidence intervals and selectivity for size classess was evaluated using the Chessons index. With principal prey species densities estimated to be ~93 individuals km^{-2} wild ungulate prey contributed ~71%. While sambar *Cervus unicolor* and chital *Axis axis* contributed greatly to the diets of both the predators (~75% biomass consumed by tiger and ~87% biomass consumed by leopard), livestock living in proximity to the park contributed substantially to the diet of the tiger (~22% biomass consumed by tiger). The dietary niche overlap between the two carnivores was substantial (Piankas' index – 0.63), though tigers positively selected for sambar and leopards positively selected for chital. The mean weights of prey obtained from scats were 98.1kg for tigers and 44.9kg for leopards. The selectivity for prey size classes revealed that tigers selected for large bodied prey (above 50kg) while leopards selected for medium bodied prey (20-50kg). With the study showing that niche separation and co-existence of these predator species were facilitated by prey selectivity patterns that were possibly enabled because of the availability of abundant prey in different size classes, we emphasise the need to continually monitor food habits of both tigers and leopards to evaluate the response of these two predators to the relocation of the *gujjar* settlements.

Separation between two sympatric carnivores in Chilla range of Rajaji National Park

Abishek Harihar, Chandan Ri, Bivash Pandav and S. P. Goyal

1. Introduction

Tigers *Panthera tigris* are obligate carnivores that occur sympatrically with leopards *P. pardus* in most of their range. These wide ranging congeneric mammalian carnivores differing in size (tiger: 120-270kg, leopard: 30-90kg) largely prey on ungulates such as cervids, bovids and suids (Schaller 1967; Seidensticker 1976; Johnsingh 1983; Karanth and Sunquist, 1995, 2000; Andheria et al. 2007). While illegal killing of tigers and leopards for body parts has contributed greatly to the extinction of local populations (Project Tiger 2005), the continual loss of vast tracts of forested landscape that once housed these predators to human habitation has caused a sharp decline in the ungulate populations and confined many of the remaining populations of tigers to small, isolated patches of forests (Smith et al. 1998). Though facing the same threats, leopards are more successful than tigers, largely because of their ability to live in different environments and the flexibility in their diet (Bailey 1993). Previous studies on the food habits of sympatric leopards and tigers have shown that their diets are very similar when prey is abundant (Schaller 1967; Johnsingh 1983; Karanth and Sunquist 1995; Andheria et al. 2007). However, under deteriorating habitat conditions resulting in declining densities of large ungulate prey, leopards may not be as adversely affected as tigers due to their ability to shift towards smaller prey (Ramakrishnan et al. 1999). Therefore, studies assessing the food habits of sympatric carnivores not only aid in our understanding of factors influencing their ecological segregation but may also serve as an indicator of change in habitat quality (Ramakrishnan et al. 1999).

This study carried out since the winter of 2004 to early 2007 assesses the food habits of tigers and leopards following the relocation of the *gujjar* settlements from the Chilla range of Rajaji National Park (RNP). *Gujjars*, a pastoralist community inhabit many areas of the Shivaliks. With their large holdings of Buffalos (*Bubalis bubalis*), intensive grazing, lopping and firewood extraction, a lack of sustainable regeneration and proliferation in weeds (Edgaonkar 1995) has led to habitat degradation. With studies indicating that wild herbivores in particular are adversely affected by populations of livestock (Mishra and Rawat 1998) and given that wild prey populations are a determinant of the viability of tiger populations (Karanth and Stith 1999), the recovery of even small populations of tigers would depend on minimizing anthropogenic pressures (Cardillo et al. 2004). Following a relocation program initiated by the Uttarakhand Forest Department in 2003 much of the occupied areas within RNP were made free of *gujjar* settlements. With evidences of recovery of wild herbivore populations following the minimisation of livestock mediated competition (Harihar et al.), the assessment of the dietary habits of tigers and leopards assumes greater importance.

2. Methods

Study area

This study was conducted from 2004 to 2007 following the relocation of *gujjar* settlements from the Chilla range (148 km^2) of RNP (820 km^2) along the eastern bank of the river Ganges, which forms the western limit of THB II (~3000 km^2; Johnsingh et al. 2004). Narrowly connected to THB I (~1800 km^2) through the Chilla-Motichur corridor (Johnsingh et al. 1990), Chilla also maintains connectivity with Corbett Tiger Reserve (CTR) through the Rajaji-Corbett corridor (Johnsingh and Negi 2003; Johnsingh et al. 2004). The range is characterised by rugged hills ranging from 400m to 1000m in altitude with steep southern slopes and is drained by rivers and streams running north to south, most of which remain dry in late winter and summer. Broadly, the forests of this region can be categorized as Northern Indian Moist Deciduous Forest and Northern Tropical Dry Deciduous Forest (Champion and Seth 1968), with the major associations being miscellaneous forests on the southern slopes and Sal (*Shorea robusta*) mixed and Sal dominated forests on the northern slopes, while the valleys have extensive grasslands. The large carnivores in the area are the tiger and the leopard. The potential prey species in the study area are sambar *Cervus unicolor*, chital *Axis axis*, barking deer *Muntiacus muntjak*, nilgai *Boselaphus tragocamelus*, wild pig *Sus scrofa*, goral *Nemorhaedus goral*, common langur *Semnopithecus entellus*, Rhesus macaque *Macaca mulatta*, porcupine *Hystrix indica*, Hare *Lepus nigricollis* and Indian peafowl *Pavo cristatus*. Domestic livestock (chiefly cattle and buffalo) found bordering the range are also potential prey species.

Estimating the density of prey species

Densities of the wild prey species were estimated using line transects in conjunction with conventional distance sampling procedures (Anderson et al. 1979; Burnham et al. 1980; Buckland et al. 1993; 2001). A total of 9 line transects were permanently laid, with lengths varying from 0.91 km to 2.49 km in different parts of the study area covering all vegetation types (Figure 1). The total length of line transects were 12.85 km. Each line transect was walked 4 times each during every survey year, thus the total effort amounted to 102.8 km of walk per year. Line transect data was collected between 0615 hrs and 0930 hrs by two observers. On every walk we recorded, species, group size, age-sex composition, sighting angle measured using a hand held compass (KB 20, Suunto, Vantaa, Finland) and sighting distance measured by a laser range finder (Yardage Pro 400, Bushnell, Overland Park, Kansas USA).

Population density of principal prey species was estimated using program DISTANCE 5 Release 2 (Thomas et al. 2006). In order to model detection functions so as to estimate species density, the data for each species per survey year was examined for signs of evasive movement and peaking at great distance from the line of walk. Following this, suitable modifications were made so as to ensure a reliable fit of key functions and adjustment terms to the data in order to arrive at density estimates. Akaike Information Criterion (AIC) and goodness-of-fit (GOF-p) tests were used to judge the fit of the model. Using the selected model, estimates of group density (D_g) and individual density (D_i) were derived.

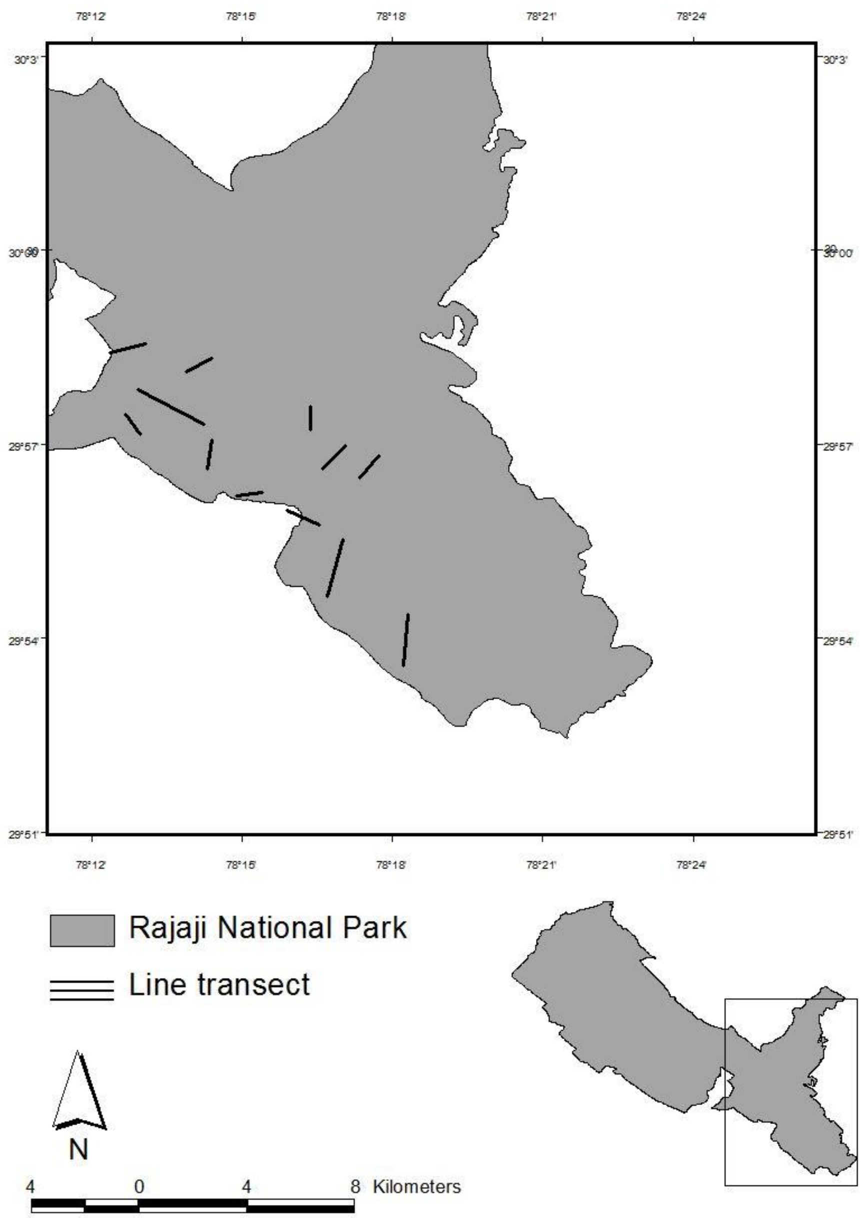

Figure 1. Map of Chilla range showing the location of the nine line transects over the three survey years (2004-2007).

Diet analysis

Scat samples were collected from roads, trails and paths opportunistically. The distinction between leopard and tiger scats was made from the size of scats and signs (pug marks and scrapes) present in the area. During the collection of scats from the field, the GPS locations were also noted. The samples were then sun dried in the field, individually labelled and brought to the laboratory for further analysis. The contents of the scats were sieved and prey remains such as bones, hooves, teeth and hairs separated. By using features such as medullary and cuticular structure (Mukherjee et al. 1994) from the hair in the scats, individual species were identified by comparing it to reference samples from the Wildlife Institute of India. The frequency of occurrence (F%; per cent of a particular species in the total number of prey items found) was calculated, since F% is known to be misleading (Floyd et al. 1978; Ackerman et al. 1984) the relative biomass (D%) and numbers (E%) of individual prey species killed by tigers and leopards using scats was computed using the Akcerman regression equation (Ackerman et al. 1984) developed for puma (*Puma concolor)*. Assuming that the digestive system and degree of utilization of carcass by the tiger and leopard are comparable to that of the puma (Karanth and Sunquist, 1995)

$$Y = 1.98 + 0.035\ (X)$$

where, X represents the live weight of the prey species represented in one collectable scat Y was used. The average number of collectable scats (λ_i) produced by a predator from an individual animal of each prey species ($\lambda_i = X/Y$) and the relative biomass and numbers of each prey killed were computed (Ackerman et al. 1984). The extent of trophic niche overlap was measured using Pianka's index (Pianka 1973) given by

$$O_{tl} = \sum p_{it}p_{il} / \sqrt{\sum p_{it}^2 p_{il}^2}$$

where, p_i is the frequency of occurrence of prey species i in the diet of tiger t and leopard l were computed. Prey selectivity and annual wild prey off take was calculated based on availability of the prey species derived from density estimates using line transects. To estimate the expected proportions of prey in the diets of these predators under non-selective predation of wild prey species, prey species densities (D_i; individual density) derived through line transect sampling were used. Under the null hypothesis that the prey killed is in proportion to availability, a comparison between the observed proportions in scats to the expected proportions was carried out by constructing 95% Bonferronis' simultaneous confidence interval. Further to assess predator selectivity for prey across size classes, we first categorised wild prey species as either being small bodied (peafowl and common langur, <20 kg), medium (chital and wild pig, 20-50 kg) or large (sambar and nilgai, above 50 kg). We then compared the proportion of these categories consumed by predators in proportion to their availability and used selectivity indices (Chesson 1978).

$$\alpha_i = (r_i d_i^{-1}) / \sum (r_i d_i^{-1})$$

where, r_i denotes the proportion of the *i*th size category being represented in the diet of the predator and d_i representing the group density of the prey species of the *i*th size category.

5. Results

Density of prey species

Densities were estimated for six of the 13 species that were detected on transects. Estimates of density could not be computed for the remaining seven species due to sample size constraints (Buckland et al. 1993). Though common langur, wild pig and nilgai had fewer than the recommended 60-80 detections, it conformed to the underlying assumptions of model fitting (Buckland et al. 1993; Laake et al. 1994). The estimates of group density (D_g), group size (GS) and individual density (D_i) of six principal prey species (sambar, chital, nilgai, wild pig, peafowl and langur) were derived; from the data we estimated a total prey species density of about 93 individuals km^{-2} (Table 1), with about 71% being contributed by

Table 1. Prey species densities in Chilla range, RNP, India, from 2004-2007. D_g, estimated density of groups (number of groups km^{-2}) with associated standard error (SE); D_i, density of individuals (number of individuals km^{-2}) with associated standard error (SE) and overall densities of wild prey species D_g and D_i are given as mean across the three survey years with associated standard deviation (SD). Total effort of 102.8 km per survey year.

Species	2004-05		2005-06		2006-07		Overall	
	D_g(SE)	D_i(SE)	D_g(SE)	D_i(SE)	D_g(SE)	D_i(SE)	D_g(SD)	D_i(SD)
Sambar	10.5(1.8)	21.3(4.1)	9.8(1.8)	13.1(2.5)	11.8(2.8)	14.6(3.6)	10.78(1.02)	16.36(4.38)
Chital	6.6(1.5)	41.5(10.7)	9.6(2.6)	41.6(13.6)	15.6(3.1)	49.9(13)	10.65(4.58)	44.35(4.81)
Nilgai	0.7(0.3)	1.7(0.9)	1.9(0.9)	1.9(0.9)	0.9(0.9)	2.4(2.4)	1.21(0.65)	2.02(0.38)
Wild Pig	2.4(0.9)	8.1(3.9)	0.3(0.2)	1.1(0.9)	0.9(0.8)	1.9(1.3)	1.24(1.05)	3.68(3.83)
Ungulate	20.2	**72.6**	21.6	**57.7**	29.2	**68.8**	23.88	**66.43**
Langur	6.4(8.4)	25.3(33.7)	1.7(0.7)	21.4(10.6)	1.7(0.6)	14.1(5.8)	3.29(2.71)	20.30(5.71)
Peafowl	2.8(0.9)	11.6(4.6)	0.6(0.3)	0.8(0.5)	2.8(1.4)	6.5(3.7)	2.11(1.24)	6.36(5.42)
Total	29.4	**109.5**	23.9	**79.9**	33.7	**89.4**	29.29	**93.11**

wild ungulates (66 individuals km^{-2}). Of the groups 18.3% were of small bodied animals (peafowl and common langur, <20 kg); 40.6% were of medium sized animals (chital and wild pig, 20-50 kg) and 40.9% were of large bodied animals (sambar and nilgai, above 50 kg).

Diet

During the entire study, we collected 240 faecal samples; 164 of tigers (45; 2004-05, 53; 2005-06, 66; 2006-07) and 76 of leopards (44; 2004-05, 21; 2005-06, 11; 2006-07). Though a total of 9 prey species were identified (Table 2), sambar and chital contributed up to ~75% biomass consumed by tiger and ~87% biomass consumed by leopard. With a representation of about 5 prey items in the diet of each of the two predators, the Pianka's index of dietary niche overlap was computed as 0.63.

Using the estimates of average body weights of prey species from Scahller (1967) and Karanth and Sunquist (1995), we estimated the relative biomass (D%) and the relative number of individual prey species consumed (E%) from the relative occurrence of prey items (F%) in predator scats (Table 2). During the study period estimates of domestic prey (Cattle and Buffalo) densities could not be attained owing to the poor representation of spatial replicates outside the national park boundary. Therefore we tested predator selectivity using individual densities (D_i) under the assumption that the predators consumed prey in proportion to their availability in the environment. On constructing 95% Bonferroni's confidence intervals on the observed proportion of prey items in the predator's diet, positive selectivity (preference) was observed for sambar in the diet of tiger and chital in the diet of leopard. The mean weights of prey obtained from scats were 98.1kg for tigers and 44.9kg for leopards.

Table 2. The frequency occurrence (F%), relative biomass (D%) and the relative number of individual (E%) prey items consumed by tiger and leopard in Chilla range, RNP, India, from 2004-2007.

Prey species	Body Weight (kg)*	F%		D%		E%	
		Tiger	Leopard	Tiger	Leopard	Tiger	Leopard
Buffalo	150	4.76	-	5.92	-	3.97	-
Sambar	134	53.74	23.08	61.68	37.02	46.24	12.41
Cattle	120	15.65	-	16.64	-	13.93	-
Chital	47	21.77	64.84	13.58	56.52	29.02	54.04
Wild pig	32	4.08	-	2.18	-	6.84	-
Barking deer	20	-	5.49	-	3.54	-	7.96
Langur	9	-	3.3	-	1.82	-	9.08
Hare	3	-	2.2	-	1.1	-	16.51
Unidentified		-	1.1	-	-	-	-

[*] Derived from Schaller (1967), Karanth and Sunquist (1995).

Evaluating the selectivity towards prey size classes we used group densities (D_g) pooled for the size classes (small bodied animals, <20 kg; medium sized animals, 20-50 kg and large bodied animals, above 50 kg) in comparison to the proportion of the size category being represented in the diet of the predator (Figure 2). Our results indicated that tigers selected for large bodied prey (above 50 kg) while leopards selected for medium sized animals (20-50 kg).

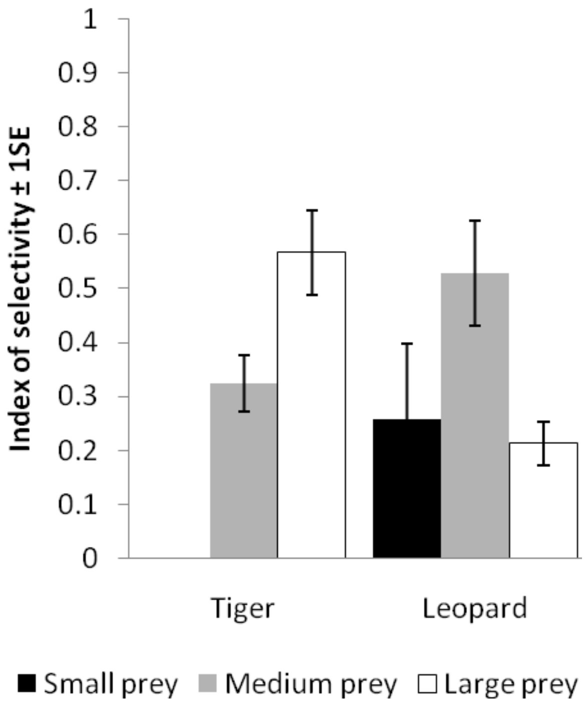

Figure 2. Selectivity (Chesson index ± 1SE) of prey size classes (small bodied animals, <20 kg; medium sized animals, 20-50 kg and large bodied animals, above 50 kg) by tiger and leopard in Chilla range, RNP.

Discussion

Miquelle et al. (1999) and Karanth and Stith (1999) show that prey distribution and density determine first order and second order habitat selection by tiger respectively. The estimates of prey density arrived at in this study show that Chilla range of RNP harbours a high density of prey species (~93 individuals km^{-2}) primarily contributed by chital and sambar (91%; Table 1). Of the animal groups encountered on transects 40.9% were of large prey (sambar and nilgai, above 50 kg) and 40.6% were medium prey (chital and wild pig, 20-

50 kg) as compared to 40.5% and 37.5% in Ranthambore (Bagchi et al. 2003), 14.37% and 60.04% in Nagarhole (Karanth and Sunquist 1995) and 8.5% and 63.2% in Pench Tiger Reserve (Biswas and Sankar 2002), suggesting the availability of large and medium ungulate prey in the region. As high prey biomass correlates to higher chances of persistence of predator populations (Carbone and Gittleman 2002) such areas will have to be protected so as to ensure the long-term survival of such wide ranging mammalian predators.

Despite the differences in body size, both tigers and leopards are obligate carnivores that primarily prey upon ungulates in all the ecosystems in which they occur (Seidensticker 1997), this study documented a substantial dietary overlap (0.63) supporting the observations of Karanth and Sunquist (1995) and Andheria et al. (2007) among tigers and leopards in southern India. Our results also supported the predictions of Griffiths (1975) that vertebrate predators would be 'energy maximisers' in prey rich habitats, with larger predators specialising on larger prey (Rosenzweig 1966; Gittleman 1985). With ~84% relative biomass consumed being contributed by prey larger than 50kg (Table 2) the mean prey weight of tigers was 98.1kg as opposed to 44.9kg for leopards that relied primarily on medium to small bodied prey (~63%; Table 2). Though studies have speculated that tigers and other large predators may not take livestock if wild ungulate prey is abundant (Biswas and Sankar, 2002; Reddy et al. 2004). Livestock from villages adjacent to our study area sometimes grazed illegally along the edges of the park thus contributing up to 22% relative biomass consumption of the tiger, possibly posing a challenge to the park management since large predators often get into serious conflict with humans over livestock depredation (e.g. Saberwal et al. 1994). However, an assessment of the selectivity for prey species indicated a preference towards sambar by tigers while, chital was seen to dominate the diet of leopards. Comparing these observations under the foraging theory (Stephens and Krebs 1987) predators may be selecting for prey that are more profitable by the ratio of energy gain to prey-handling time, thus conforming to the observations made by similar studies (Karanth and Sunquist 1995). A comparison of the Chesson's (1978) index for prey size selectivity indicated that tigers selected for large bodied animals (above 50kg) while leopards preferred medium sized animals (20-50kg). With the study showing that niche separation and co-existence of these predator species are facilitated by prey selectivity patterns that are enabled because of the availability of abundant prey in different size classes, we stress that, additionally, densities of predators, temporal and spatial segregation mechanisms and prey vulnerability could contribute to the ecological segregation of these two sympatric carnivores in the study area. As the prey base is capable of supporting a viable population of tiger following the relocation of the *gujjar* settlements, we emphasise the need to continually monitor food habits of both tigers and leopards to assess modifications to this dynamic habitat.

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Estimating population size of tigers using camera trap based capture-recapture sampling: minimizing closure violation and improving estimate precision

Abishek Harihar, Bivash Pandav and S.P. Goyal

***Abstract*:** Photographic capture-recapture sampling has in the recent past aided in estimating state variables (abundance or density) of cryptic carnivores. However, most studies estimate parameters of interest with ecological and sampling uncertainties and it is thus important to evaluate trap effort in terms of assumption violation and estimate precision. In this study we evaluate the importance of trapping effort (trap area, mean cell area and mean minimum inter trap distance) on the assumption of geographic closure and their influence on capture probability across the sexes among tigers by sub sampling capture histories obtained from 30 trapping stations within the Chilla range of Rajaji National Park, India. We used curve estimation procedures to assess the extent of spatial coverage by traps with respect to the population under consideration and then assumed demographic closure for the study period (45 days) to evaluate the importance of trap area, mean cell area and mean minimum inter trap distance on geographic closure by estimating fidelity ($\hat{F}$) and immigration ($\hat{f}$) under the Pradel model. Estimate precision

was evaluated based on estimates of capture probability. Our results suggested a linear increase in individuals with sampled area therefore indicating that the total trapping grid sampled only a part of a wider ecological population. Owing to large standard errors in parameter estimates, differences among sexes were not evident. However, larger trap area ensured geographic closure and smaller mean minimum inter trap distance positively influenced recapture rates. While, higher estimated capture probability positively enhanced estimate precision. In most situations the goals of managing natural animal populations are expressed in terms of population size. For this purpose unless the right method of estimation is chosen and precise estimates arrived at, trends cannot be detected. Therefore while carrying out live trapping studies using capture-recapture models overcoming sampling uncertainties is of great importance. Our study highlights the need to achieve a tradeoff between trap spacing and spatial coverage while reliably estimating abundance of large ranging elusive mammals such as tigers.

Estimating population size of tigers using camera trap based capture-recapture sampling: minimizing closure violation and improving estimate precision.

Abishek Harihar, Bivash Pandav and S.P. Goyal.

1. Introduction

Estimating population size of wild animals is of prime importance to ecologists and managers. In most situations the goals of managing natural animal populations are expressed in terms of population size. Though a single estimate of population size is of limited value, additional estimates such as those from the same place over years or across habitat types at the same time would better explain the status of the population. For this purpose unless the right method of estimation is chosen and precise estimates arrived at, trends in populations cannot be detected. The use of capture-recapture theory (Otis et al. 1978, Pollock et al. 1990) and remotely triggered cameras to capture individually identifiable animals has resulted in their use for estimating demographic parameters (Karanth 1995, Karanth and Nichols 1998, Karanth et al. 2006). Owing to its +applicability in a wide variety of habitats (Karanth et al. 2004) and ability to provide information on activity pattern, habitat use and reproductive status of cryptic carnivores, camera trapping has in the recent past gained popularity (Griffiths and van Schaik 1993, Karanth 1995, Karanth and Nichols 1998, O’Brien et al. 2003, Trolle and Kery 2003). The basic methodology involves photographing individuals of the target species which are uniquely identifiable by their natural markings. By conducting repeated surveys over a short period of time, individual capture histories are constructed that aid estimate abundance and other parameters of interest. One of the most important assumptions to be met while estimating population size under a closed capture estimator is that of population closure. Implying that, the population under investigation does not undergo change (birth, death, immigration and emigration) during the period of the study. Given the longevity of most cryptic carnivores, sampling over relatively short durations of time (45 days, as is in this study) can aid ensure demographic closure. However, movement of animals in and out of the sampling grid (i.e. immigration and emigration) for wide ranging animals will have to be ensured by spatial coverage. A general trend indicates that estimate precision is influenced by capture probabilities which are in itself a function of trap intensity and spacing (Boulanger and McLallen 2001, Boulanger et al. 2002). Therefore studies wishing to estimate population size of target species will have to ensure closure by sampling large enough areas in relation to the population under investigation as well as intensify trap effort to maximize capture probability and therefore attain desired levels of precision. Though the initial costs involved are high, Silveria et al. (2003) camera trapping proves cost effective, when surveying large areas over long periods of time.

Abundance of a species is a fundamental demographic attribute constrained primarily by energy requirements. Gittleman and Harvey (1982) suggest that obligate carnivores such as the tiger (*Panthera tigris*) would have to range over large areas and therefore naturally occur at low abundances. The social organization of tigers is dependent on breeding females that maintain home ranges within which they raise cubs. While range size of breeding females (13-30 km^2) are primarily a function of age and prey density, males maintain home ranges (40-100 km^2) that usually overlap with an average three females (Sunquist 1981, Smith et al. 1987, Smith 1993, Miquelle et al. 1999, Karanth and Sunquist 2000). Tigers are typically classified into demographic classes (Karanth and Stith 1999) irrespective of the sexes into; cubs (less than 1 year old), juveniles (1-2 years old), breeding adults (3-12 years old), post-dispersal floaters or transients (over 2 years) and evicted breeders. Acknowledging the fact that capture probabilities ($\hat{p}$) vary with respect to age (Karanth and Nichols 1998), sex, range size among other factors it is important to achieve adequate sample sizes (uniquely identified individuals and recapture rates) while estimating population parameters within a capture-recapture framework (White et al. 1982). Though estimates of vital rates (survival, recruitment and movement) over years in populations of tigers have been estimated with reasonable precision given various ecological and sampling uncertainties (Karanth et al. 2006), we feel most studies restricted to single-season estimates of state variables (abundance) would require optimal study design in terms of trap spacing and total area sampled in relation to the population under investigation so as to achieve desired levels of precision.

Since delineating the spatial boundary of a population is difficult (Dasmann 1981), studies estimating abundance or density often are unable to sample the entire area of interest (Smallwood 1999, Williams et al. 2002), therefore investigating only a sample of a wider ecological population. Following Karanth (1995), studies estimating population size of tigers and other cryptic large mammals using photographic capture-recapture sampling (Karanth and Nichols 1998, O'Brien et al. 2003, Karanth et al. 2004, Kawanishi and Sunquist 2004, Wegge et al. 2004) have adopted an approach of trap placement that maximises capture probabilities. Under this design, camera traps are set throughout the sampling area at optimal locations with prior knowledge of the use of the habitat by tigers and minimising the chance that some individual has a near-zero capture probability. It is known that concentrating trap effort over a small area will increase both the capture probability and the probability of animals being absent. Conversely, diluting the effort over a larger area will decrease both temporary emigration and capture probability (Kendall 1999). Estimating population size using closed population estimators requires that the most important assumption of demographic and geographic closure is not violated. Though demographic closure can be assured by sampling over a relatively short period of time, geographic closure will have to be ensured by spatial coverage of traps. Instead of using a hypothesis testing frame work (Pollock et al. 1974, Otis et al. 1978, Stanley and Burnham 1999) we assess spatial coverage of the trapping grid in relation to the population boundary by regressing the estimated population size ($\hat{N}$) to the trap area (TA). We also estimate apparent survival ($\hat{\Phi}$), recruitment ($\hat{f}$) and recapture probability (p) under the Pradel (1996) model incorporated in program MARK to test for geographic closure (Boulanger and McLellan. 2001, Boulanger et al. 2002). The Pradel (1996) model estimates apparent survival ($\hat{\Phi}$) as a product of true survival (S) and fidelity ($\hat{F}$) for the sampled area ($\hat{\Phi} = S\hat{F}$), since we assume demographic closure, true survival is unity (S=1) and apparent survival is thus defined as the probability of presence of an individual within the sampling grid during the period of investigation ($\hat{\Phi} = \hat{F}$). The recruitment rate ($\hat{f}$) estimated as the number of new individuals in the population at time i+1 per individual at time i, is a measure of immigration into the sampling grid under the assumption that no births (demographically closed) occur during the course of the study. Therefore assuming that the population was demographically closed, the Pradel (1996) model helped investigate closure violations in relation to movement in and out of the sampling area.

By constructing capture histories obtained from random subsets of the 30 trapping stations (Figure 1) deployed from December 2005 – February 2006 to estimate the abundance of tigers in the Chilla range of Rajaji National Park (RNP) we examine the individuals-area relationship under the hypothesis that a saturation in individuals with increase in sampling area is indicative of self contained population defined by the topography while a linear increase could indicate that the organisms under investigation are part of a wider ecological population. We then analyse the effect of trapping effort (trap area, mean cell area and mean minimum inter trap distance) on the assumption of geographic closure and their influence on capture probability across the sexes. Given the inherent differences between male and female ranging patterns we hypothesise that males ranging over large areas would exhibit lower fidelity and higher immigration rates into the sampling grid as opposed to females. Given that males range over larger areas we hypothesise that the encounter rate of camera traps would be higher and therefore rates of recapture would be higher than that of females. We also investigate the relationship between capture probabilities on the precision of estimates of population size and provide suggestions on the methodology of photographic capture-recapture sampling.

2. Study area

The Shivaliks running parallel to the outer Himalayas from Jammu through Nepal to Assam are considered one of the most threatened and fragile ecosystems in the Indian subcontinent. Also considered as one of the most productive landscapes along with the Terai flood plains (Wikramanayake et al. 2004), it is most prone to human disturbances (Johnsingh et al. 2004). With a human population density of over 500 people km^{-2}, this region is highly populous, surpassing the national average of 300 people km^{-2} (Johnsingh et al. 2004). Viable populations of tigers in this landscape exist only in small patches that have very low connectivity (Johnsingh et al. 2004, Wikramanayake et al. 2004). The Rajaji National Park forms the north-western limit of the tigers distribution in India. This proposed National Park is bisected by the river Ganges. The forests to the west of the Ganges extend up to the Yamuna river and are narrowly connected to the forests on the east bank through the Chilla-Motichur corridor (Johnsingh et al. 1990, Johnsingh and Negi 2003, Johnsingh et al. 2004). However, the Chilla range of RNP (east of the Ganges) is connected with Corbett Tiger Reserve (CTR) via the Rajaji-Corbett corridor (Johnsingh and Negi 2003, Johnsingh et al. 2004). Known as the Rajaji Corbett Tiger Conservation Unit (RCTCU; Johnsingh and Negi 2003) it is a level I Tiger Conservation Unit (TCU I; Wikramanayake et al. 1998) identified for the long term persistence of the species. The protected areas of RNP and CTR constitute only about 30% of this landscape. The surrounding landscape matrix consists of multiple use forests, agricultural land and human habitation thus being less suitable for tiger persistence.

3. Methods

Field methods

Photographic captures of tigers were obtained by placing active infrared tripping devices (TRAILMASTER TM 1550; Goodson and Associates, Kansas, USA) attached to cameras (Canon A1 sure shot; Canon Inc., NY, USA) at thirty optimal locations based on a pilot survey so as to maximise the capture probability of tigers across the sampling area (Figure 1). In view of resource constraints, we had only ten tripping units and therefore in order to systematically sample the area, three trapping blocks (spatially separated) were identified within the intensive study area and the cameras were deployed in a phased manner. Each trapping block consisted of 10-trap stations run for 15 consecutive days. Thus, each sampling occasion combined captures from one day drawn from each block. One trap-night was a 14-hour period (1700 – 0700 hrs) during which a camera was functional. Owing to a good network of roads, all the 10 trapping stations in each of the three blocks were checked on a daily basis. All rolls of film used during the trapping were given a unique identity (e.g. Block1/Trap1/Roll1) so as to correctly note the date, time and location of the captures. Every tiger captured was given a unique identification number (e.g. t_1) after examining the stripe pattern on the flanks, limbs and fore-quarters (Schaller 1967, McDougal 1977, Karanth 1995). Following the identification of tigers, individual captures at each trapping station were noted and capture histories (X matrix) were developed.

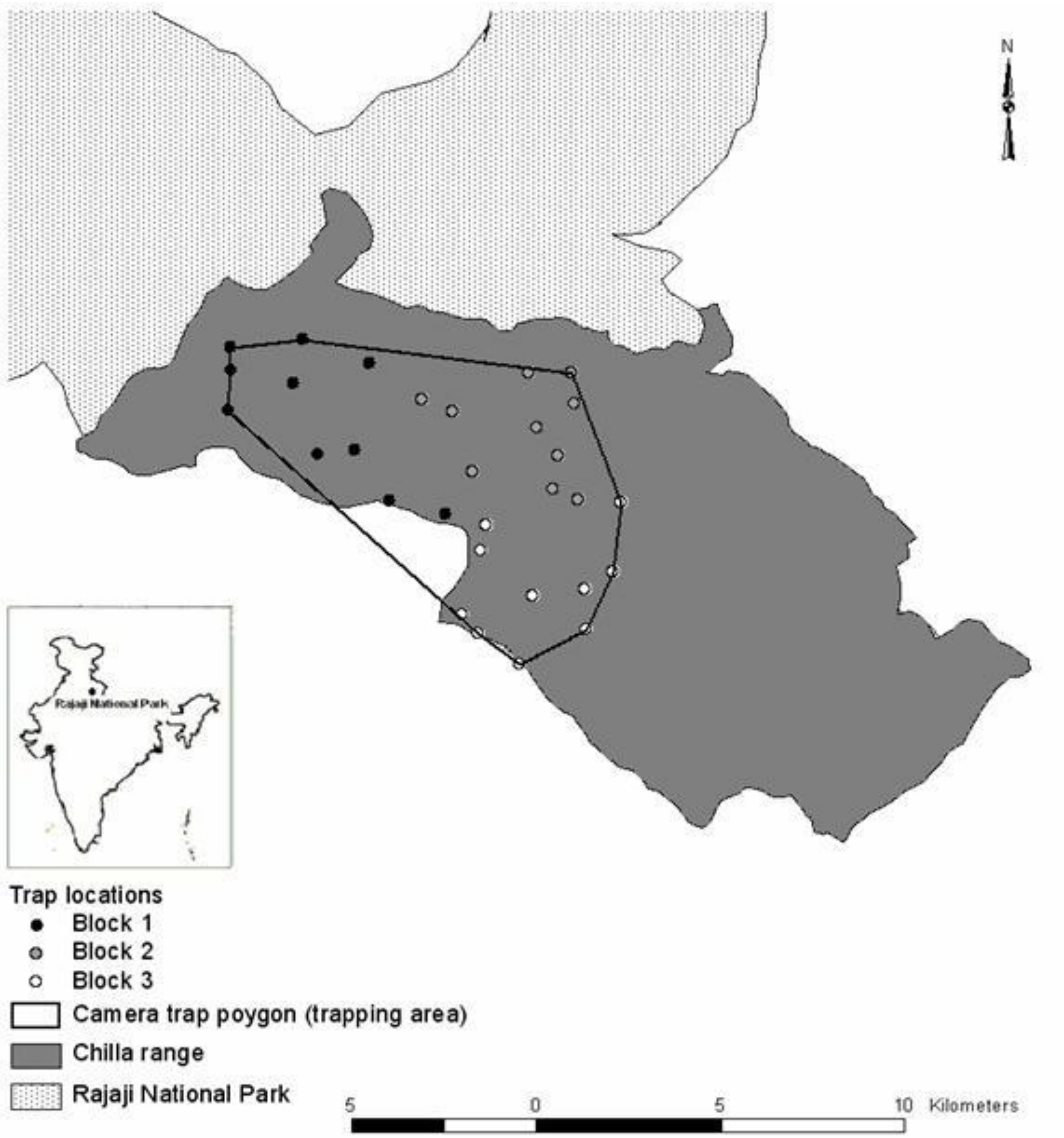

Figure 1. Camera trap locations in three spatially separated blocks within Chilla range of Rajaji National Park placed to maximize probability of capture of tigers during the study (December 2005 – February 2006).

Analysing the effect of trapping effort on the assumption of population closure and recapture rates

After recording tiger captures at each station, 10 random draws each (with replacement) were made by varying the number of traps (5, 10, 15, 20 and 25) within the trapping grid of 48 km^2. X matrices were developed for each of these sub-samples and population size ($\hat{N}$) was estimated using closed capture estimators of program MARK (Otis et., 1978, White and Burnham 1999) by modelling for variations in capture ($\hat{p}$) and recapture ($\hat{c}$) probabilities. Finite mixture models (two mixtures; Pledger 2000) were used to investigate the effect of heterogeneity. A total of eight models were constructed with $\hat{p}$ and $\hat{c}$ estimated as either being constant (.) or varying with time (*t*) across the sampling occasions. Fit of models was evaluated using Akaike Information Criterion (AIC_c). The model with the lowest AIC_c was considered as the most parsimonious (Burnham and Anderson 1998). For each of the random sub-samples the locations of camera trapping stations were plotted using the Geographical Information System environment ArcView 3.2 (ESRI, Redlands, CA, USA). Trapping area (TA), namely the spatial coverage by the trapping grid was derived by developing a minimum convex polygon (MCP) over the outermost trap stations. Cell area, defined as the area enclosed by three adjacent traps, was calculated using the Delaunay triangulation approach (Brassel and Reif 1979). Therefore the mean cell area (MCA) was estimated by averaging individual cell area estimates from the sub sampled area. Inter trap distances were tabulated and the mean minimum inter trap distances (MITD) was estimated. To investigate the nature of the individuals-area relationship, we regressed the estimated abundance ($\hat{\bar{N}}$) to trapping area (TA) and used curve estimation (SPSS, Chicago, IL, USA) procedures. Model significance was assessed at $P<0.05$ and model fit was evaluated based on the coefficient of determination (R^2). We used Pradel (1996) model incorporated in MARK (White and Burnham 1999) and analysed the effect of TA, MCA and MITD as a function of sex on Fidelity ($\hat{\bar{F}}$), immigrationn ($\hat{f}$) and recapture probability (*p*). Competing models were constructed with$\hat{\bar{F}}$, *p* and $\hat{f}$ estimated as either constant (.) or varying with time (*t*). The covariates (TA, MCA and MITD) were then standardised by mean and standard deviation and entered into MARK design matrices to formulate models. Logit link functions were used for modelling. Model fit was evaluated using AIC_c scores (Burnham and Anderson 1998).

4. Results

Population size of tigers using photographic capture-recapture analysis

Total sampling effort amounted to 450 trap nights (December 2005 – February 2006). The intensive trapping involving 30 camera traps and 15 nights of trapping at each station resulted in a total of 16 photographs of five individual tigers (three females and two males). Apparent survival $\hat{F}$ (SE[$\hat{F}$]) was estimated as 0.957(0.02) and the recruitment rate $\hat{f}$ (SE[$\hat{f}$]) was estimated to be 0.04(0.001) using the best model ($\hat{F}$ (.) p(t) $\hat{f}$ (.); K=16), therefore, providing less evidence for the violation of closure. Capture probability ($\hat{p}$ (SE [$\hat{p}$]) was estimated as 0.2133(0.047) and population size $\hat{N}$ (SE [$\hat{N}$]) as 5(0.39) by the model with the lowest AIC_c score ($\hat{p}(.)=\hat{c}(.)\hat{N}(.)$).

Effect of trapping effort on closure violation and recapture rates

Estimated abundance ($\hat{N}$) regressed to trapping area (TA) indicated a significant positive relationship. Curve estimation procedures (SPSS, Chicago, IL, USA) provided greater support to a linear increase (Figure 2) in individuals with trapping area with a non significant intercept ($F_{1,49} = 55.196$, $P<0.001$, $R^2 = 0.5301$):

$$\hat{N} = 0.1236(0.48) + 0.0979(0.013) \times TA, \qquad \text{eqn 1}$$

as opposed to a logarithmic (Figure 2) response ($F_{1,49} = 54.376$, $P<0.001$, $R^2 = 0.526$):

$$\log \hat{N} = -4.4257(1.08) + 2.3022(0.31) \times \log TA. \qquad \text{eqn 2}$$

Therefore, suggesting that, the trap area sampled only a portion of the entire population under investigation.

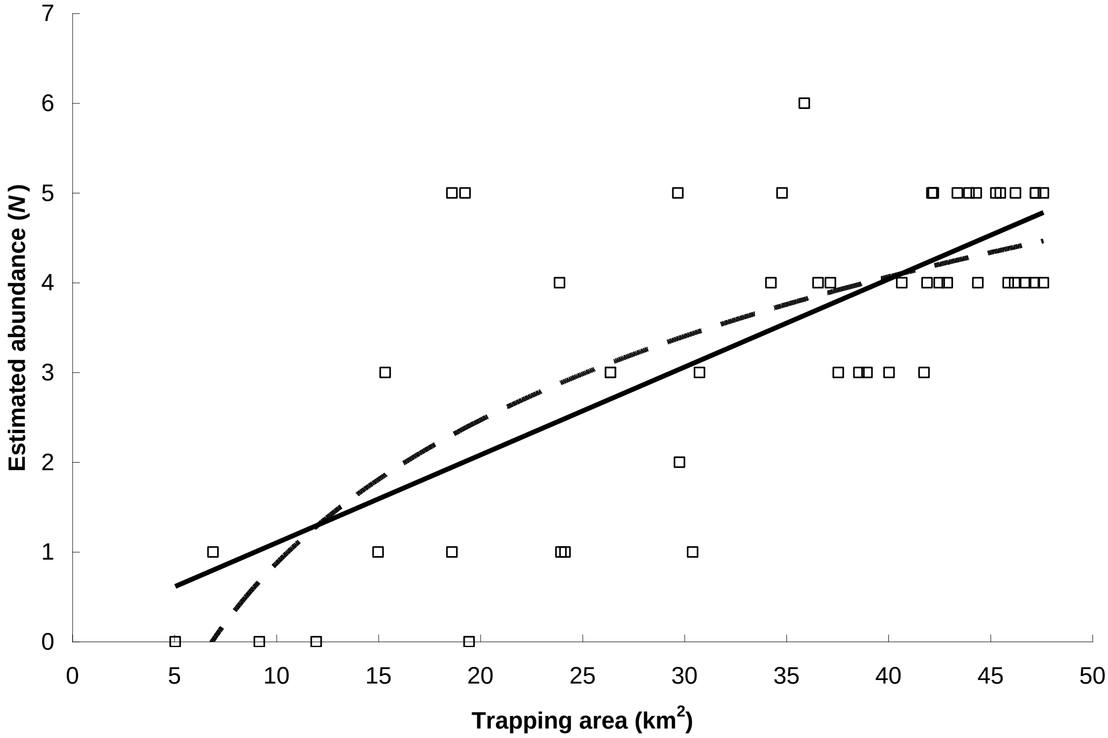

Figure 2. The individuals-area relationship for the trapping area of 48km^2. The solid line indicates a positive linear relationship while the broken line is indicative of a logarithmic function.

Owing to the high error rates associated with parameter estimates in the fully parameterised Cormack-Jolly-Seber (CJS; Cormack 1964, Jolly 1965, Seber 1965) model ($\hat{\Phi}$[sex x time] p[sex x time]) a reduced CJS model ($\hat{\Phi}$ [sex] p[sex]) was used to evaluate the degree of overdispersion. The bootstrap estimate of the dispersion factor suggested moderate overdispersion ($\hat{c}$ = 1.23). However, we decided to use $\hat{c}$ = 1 (i.e. AIC_c scores instead of $QAIC_c$) for the purpose of model selection. The Pradel (1996) model indicated that closure violation with respect to movement in and out of the sampling area was influenced by trapping effort (Table 1). The minimum-parameter model (Model 34) provided less support (Δ_i = 33.95) to variation in the estimates of fidelity, immigration and recapture rates. Though support for differences among sexes was weak, there existed some evidence of the same (Model 3; Δ_i <2). The model with the lowest AIC_c (Model 1) score indicated that TA influenced both fidelity and immigration while MITD influenced recapture rates.

Table 1. Model selection of Pradel analysis of tiger captures from December 2005 – February 2006, Chilla range of Rajaji National Park. Fidelity ($\hat{F}$), recapture probability (p) and immigration ($\hat{f}$) were modeled using covariates (TA; trapping area, MCA; Mean cell area, MITD; Mean minimum intertrap distance) as a function of sex. Presented are the competing models and their corresponding AIC_c scores, the differences in AIC_c scores between the ith model and the model with lowest AIC_c score (Δ_i), model weights (w_i), model likelihood and the number parameters (K). Covariates were modeled as having term specific slope and intercept (x) or term specific slope and common intercept (+).

Model	$\hat{F}$	p	$\hat{f}$	AIC_c	Δ_i	w_i	Model Likelihood	K
Model 1.	sex x TA	sex x MITD	sex x TA	2229.46	0	0.2777	1	8
Model 2.	sex x TA	sex x MCA	sex x TA	2229.47	0.011	0.2762	0.995	8
Model 3.	♀TA ♂TA, MCA	sex x MITD	♀TA ♂MCA, MITD	2231.08	1.619	0.1236	0.445	9
Model 4.	♀TA ♂TA, MCA	sex x MITD	♀TA ♂MCA, TA	2232.04	2.583	0.0764	0.275	10
Model 5.	♀TA ♂TA, MCA	sex x MCA	♀TA ♂MCA, TA	2232.05	2.593	0.076	0.274	10
Model 6.	♀TA ♂TA, MCA	sex x MCA	♀TA ♂TA, MITD	2232.05	2.593	0.076	0.274	10
Model 7.	♀TA ♂TA, MCA	sex x MCA	♀TA ♂ MCA, MITD	2234.16	4.699	0.0265	0.095	11
Model 8.	sex x TA	sex x MITD	(.)	2236.11	6.652	<0.01	0.036	6
Model 9.	sex x TA	sex x MITD	sex	2236.11	6.652	<0.01	0.036	6
Model 10.	sex x TA	sex x MITD	MCA	2236.11	6.652	<0.01	0.036	6
Model 11.	sex x TA	sex x MITD	MITD	2236.11	6.652	<0.01	0.036	6
Model 12.	sex x TA	sex x MCA	sex	2236.12	6.664	<0.01	0.036	6
Model 13.	♀TA ♂TA, MCA	sex x MITD	♀TA ♂TA, MITD	2238.56	9.104	<0.01	0.011	11
Model 14.	♀TA ♂TA, MCA	sex x MITD	♀TA ♂MCA	2238.56	9.106	<0.01	0.011	11
Model 15.	sex x MITD	sex x MITD	sex x TA	2238.70	9.240	<0.01	<0.01	6
Model 16.	sex x TA	sex x MITD	sex x MCA	2238.73	9.277	<0.01	<0.01	8
Model 17.	sex x TA	sex x MITD	sex x MITD	2238.73	9.277	<0.01	<0.01	8
Model 18.	♀TA ♂TA, MCA	sex x MCA	♀TA ♂MCA	2240.69	11.23	<0.01	<0.01	12
Model 19.	sex x TA, MCA	sex x MITD	sex + MCA	2240.74	11.28	<0.01	<0.01	9

Model 20.	sex x TA	sex x MITD	sex x TA	2241.66	12.20	<0.01	<0.01	10
Model 21.	sex x MCA	sex x MCA	sex x MCA	2242.29	12.83	<0.01	<0.01	8
Model 22.	sex x MCA	sex x MCA	sex x TA	2242.85	13.39	<0.01	<0.01	8
Model 23.	sex x TA, MCA	sex x MITD	sex x MCA	2243.40	13.94	<0.01	<0.01	10
Model 24.	sex + TA, MCA	sex x MITD	♀TA ♂MCA, MITD	2245.57	16.11	<0.01	<0.01	11
Model 25.	sex + TA	sex x MITD	sex	2246.70	17.24	<0.01	<0.01	7
Model 26.	sex + TA	sex x MITD	sex x MCA	2247.67	18.21	<0.01	<0.01	8
Model 27.	TA	sex x MITD	MCA	2248.19	18.73	<0.01	<0.01	6
Model 28.	sex + TA	sex x MITD	sex + TA	2250.33	20.86	<0.01	<0.01	10
Model 29.	sex + TA, MCA	sex x MITD	sex x MCA	2250.49	21.03	<0.01	<0.01	9
Model 30.	sex + TA	sex x MITD	sex + MCA	2250.86	21.40	<0.01	<0.01	9
Model 31.	sex + TA, MCA	sex x MITD	sex + MCA	2252.59	23.13	<0.01	<0.01	10
Model 32.	sex x TA	sex	sex x TA	2254.42	24.96	<0.01	<0.01	8
Model 33.	sex	sex	(.)	2263.42	33.95	<0.01	<0.01	4
Model 34.	sex	sex	sex	2263.42	33.95	<0.01	<0.01	4

As was expected, lower spatial coverage returned lower estimates of fidelity (Figure 3) and higher rates of immigration (Figure 4) into the sampling grid, therefore suggesting that geographical closure could be violated by small sampling grids, given the wide ranging behaviour of tigers. Owing to the large error rates over the estimates, no definitive evidence is apparent for differences in fidelity and immigration among the sexes; as hypothesised, males seem to exhibit lower fidelity, and however contrary to our hypothesis immigration rates of males were lower than that of females. As was hypothesised recapture rates of males seem to be more strongly influenced by MITD (Figure 5), with lower MITD positively influencing recapture rates.

The relationship between capture probabilities ($\hat{p}$) and the coefficient of variation (CV%[$\hat{N}$]) of the population estimate suggested increased estimate precision with increasing $\hat{p}$ (Figure 6). Results indicated that for capture probabilities of ≥ 0.2 resulted in estimate precision $\leq 20\%$. Thus indicating that by maximizing capture probability the precision of the estimates of population size can be significantly enhanced.

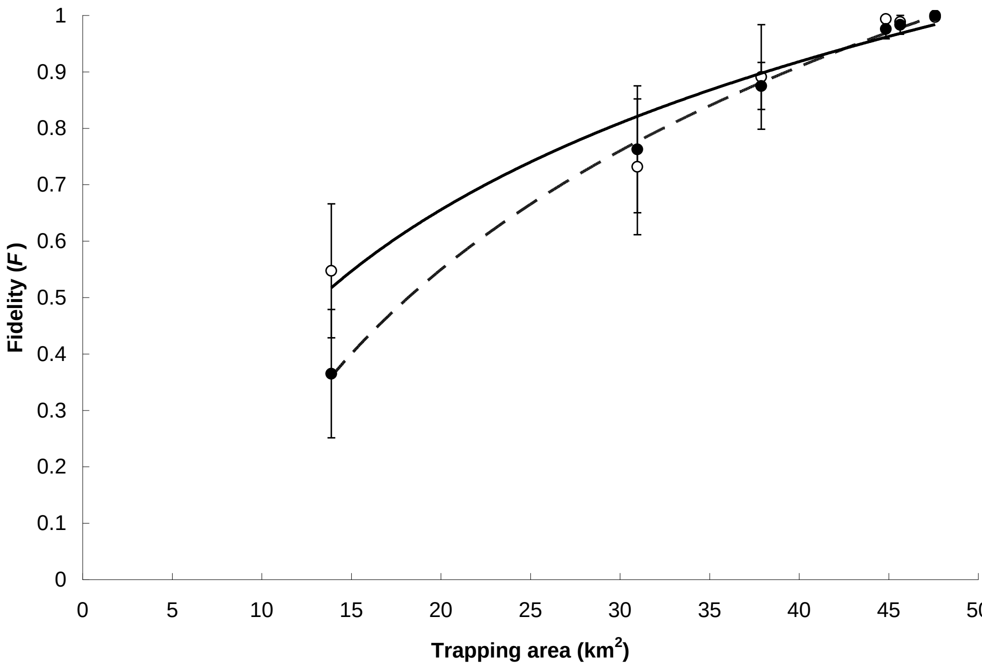

Figure 3. Response of model averaged fidelity estimates from Pradel analysis for male (broken line) and female (solid line) tigers to varying trapping area. With error bars indicating associated one standard error.

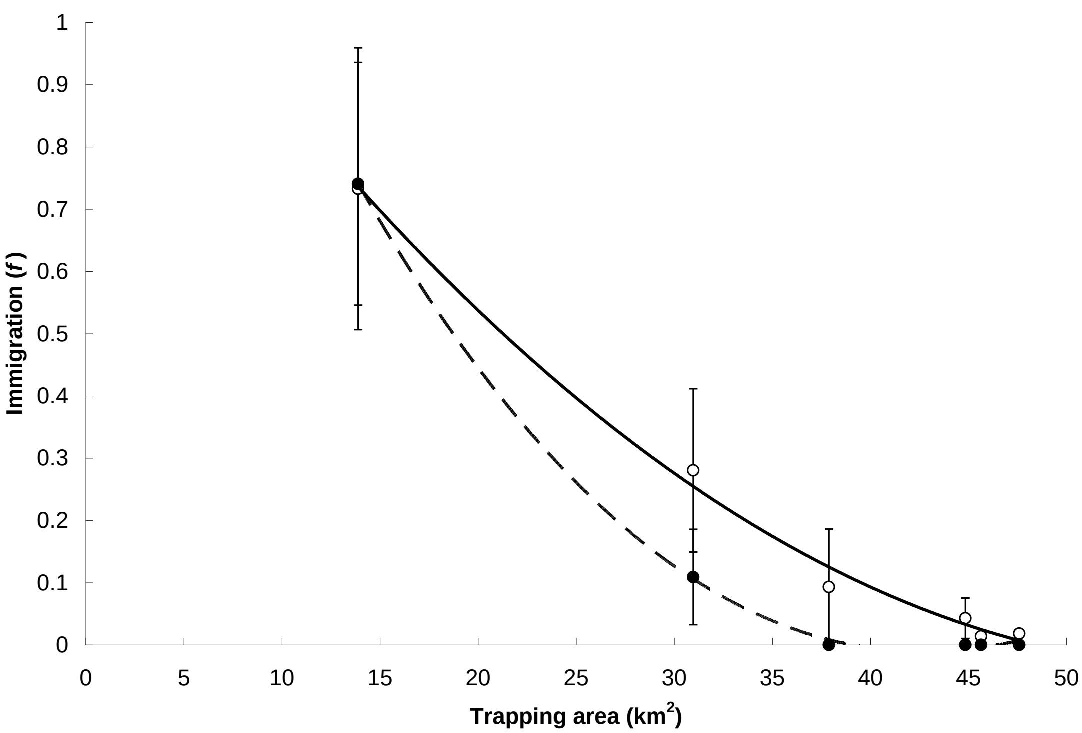

Figure 4. Response of model averaged immigration estimates from Pradel analysis for male (broken line) and female (solid line) tigers to varying trapping area. With error bars indicating associated one standard error.

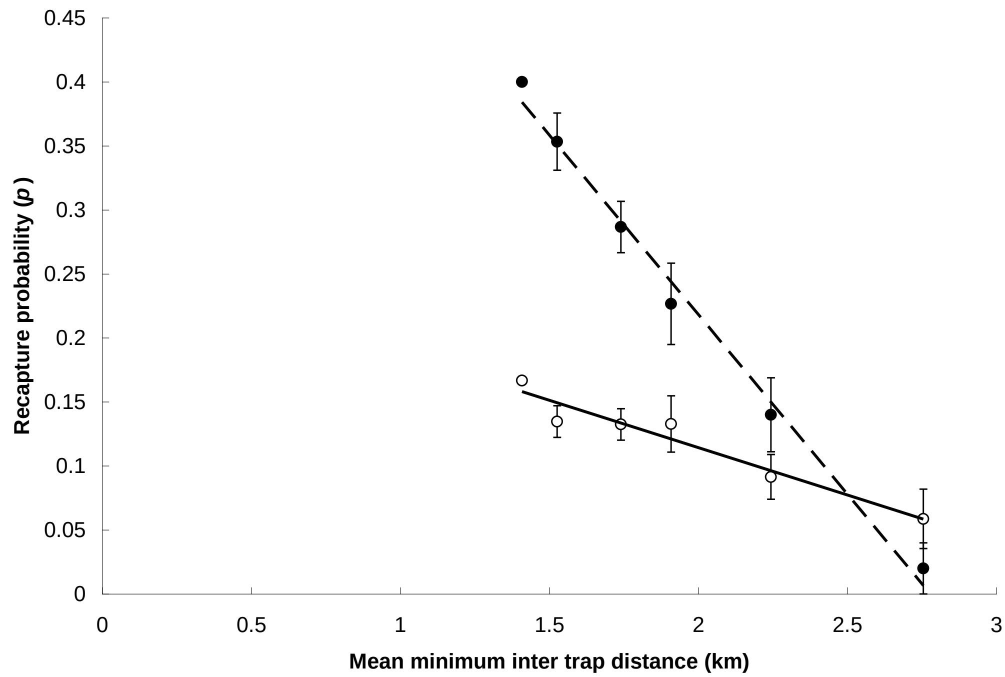

Figure 5. Response of model averaged recapture probability estimates from Pradel analysis for male (broken line) and female (solid line) tigers to varying mean minimum inter trap distance. With error bars indicating associated one standard error.

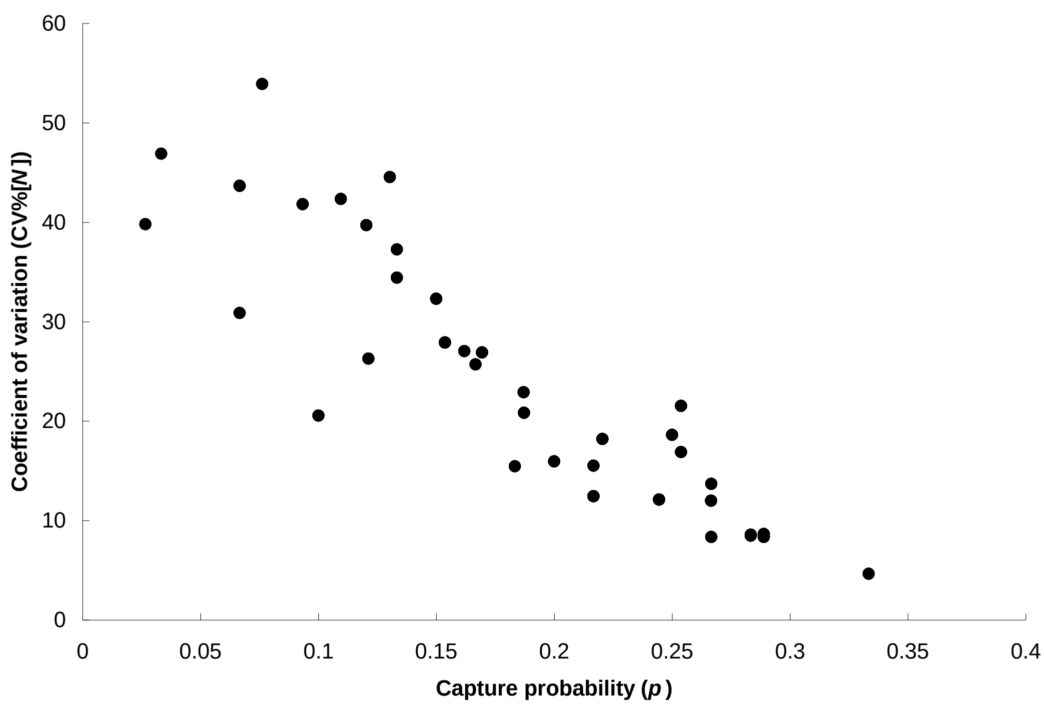

Figure 6. Response of estimate precision to capture probability estimates for tigers using closed capture estimators.

5. Discussion

Relatively homogenous landscapes are essentially always biotically saturated with individuals, implying a linear increase in individuals to sampled area (MacArthur and Wilson 1967). However, species abundance to environment relationships result in the aggregation of conspecifics, a common spatial pattern owing to limited dispersal or positive spatial auto correlation of conditions and resources in the environment (Mayor and Schaefer 2005). Our study estimated a small population ($\hat{N}$ = 5) of tiger using the Chilla range of RNP. Following a management intervention that a reduced anthropogenic pressure within this part of the park, a recovery of the tiger population is being noticed (A. Harihar, Wildlife Institute of India, unpublished data). Since the Chilla range of RNP is bounded by the river Ganges to the west and is contiguous with the Corbett Tiger Reserve through unprotected forest land to the east, the population is not strictly topographically closed. However, we feel that the landscape matrix around the Chilla range is less suitable for the persistence of tigers. The fit of the linear model to our data (eqn. 1 and Figure 2) supported that the individuals sampled form a part of a wider ecological population. However, the close fit to a logarithmic function (eqn. 2 and Figure 2) is suggestive of the fact that the spatial coverage of traps sampled a fairly large number of individuals of the population under investigation. While regression based methods such as this provide a useful tool to assess spatial boundaries, delineation of population units following Mayor and Schaefer (2005) could also be employed using secondary data. As testing for closure is difficult (Otis et al. 1978), we suggest that an objective delineation of population units be first carried out and an attempt be made to sample the entire area of interest.

Variations in capture probability among tigers is potentially due to differences in trap encounter rates (Efford 2004), which are influenced by age, sex, size and position of ranges in relation to traps among other ecological factors. These sampling uncertainties are often the cause of assumption violations. Pollock et al. (1974) assumed no variation in capture probability and tested for closure under four hypotheses: (1) no mortality and no recruitment, (2) mortality but no recruitment, (3) recruitment but no mortality, and (4) both recruitment and mortality. Later, Stanley and Burnham (1999) developed a test for closure under time specific capture-recapture data using the above four hypothesis and suggested that it be used in conjunction with the test for closure by Otis et al. (1978), as variations in capture probability due to behavioural response and heterogeneity were not accounted for. However, the test for closure by Otis et al. (1978) is based on the observed times between the first and the last capture for all animals captured at least twice, being sensitive to behavioural and temporal variations in capture probabilities this test is not suitable for detecting closure when temporary emigration occurs during the course of the study (Williams et al. 2002). Boulanger and MacLallen (2001) and Boulanger et al. (2002) demonstrated that the Pradel (1996) model can be used to test for grid closure by estimating model parameters under the assumption that the sampling design accounts for demographic closure. In this study the Pradel (1996) model estimates of apparent survival, recruitment and recapture probability in relation to sampling covariates (TA, MCA and MITD) demonstrated the effect of trap effort on geographic closure. However, trapping area (TA) and inter trap distance (MITD) are auto correlated and confounded in this study, so it is not possible to separate their combined effects. Since female home ranges are smaller than that of males it was noticed that females exhibited higher rates of fidelity, while the wide ranging movement of males in relation to the trap area resulted in lowered estimates (Figure 3). Though we hypothesised that the rates of immigration by females would be lower than that of males (Figure 4), trap encounter rates, influenced by range size and position had a marked effect on the recapture probabilities (Figure 5). We feel that the number of traps accessed by females could be the reason as to why immigration rates of females were higher than that of males. However, it is apparent that by increasing the trapping area, closure violation caused by the random movement of individuals (irrespective of the sexes) across the trapping area can be minimized. Kendall (1999) shows estimates of population size obtained as a result of such random movement in and out of the sampling grid correspond to the superpopulation (N^* (White (1996)), that cannot be used in the estimation of density as there remains an undefined sampled area. Following Wilson and Anderson (1985), Karanth and Nichols (1998) propose the use of estimating a boundary strip width ($\hat{W}$) based on the mean maximum distance moved (MMDM) between recaptures to calculate the effective sampling area ($A(\hat{W})$). While it has been demonstrated that movement distances are effectively estimated when most animals are recaptured (Wilson and Anderson 1985), White et al. (1982) shows that using observed movement distances are more useful when animal ranges are small in comparison to the sampling grid. Soisalo and Cavalcanti (2006) recently demonstrated that estimates of MMDM computed from observed recaptures of Jaguars (*Panthera onca*) are underestimates of the true range use (verified using GPS-radiotelemetry), thereby considerably inflating density estimates.

Though photographic capture-recapture estimates of population sizes of elusive mammals have been obtained from a wide range of studies, estimate precision has been relatively poor. Pollock et al. (1990) states that a CV%[$\hat{N}$] of less than 20% be attained for management purposes, as highlighted by our results TA and MITD have a strong influence in the estimation of population size. While TA should be governed by the spatial boundary of the population under investigation, the number of trapping stations and therefore the MITD will have to be optimized so as to ensure that model assumptions are not violated. Since reliable estimates of population size are arrived at when capture probability of the target species is maximised (Figure 6), trap encounter rates will have to be enhanced to obtain higher capture probabilities. This implies that studies aiming to estimate population sizes of target species will have to make a trade off between the TA and MITD given the logistic (cost of equipment and personal time spent in field to maintain the sets) constraints posed by photographic capture-recapture sampling in order to achieve desired levels of precision.

6. Management implications

The use of capture-recapture theory (Otis et al. 1978, Pollock et al. 1990) and remotely triggered cameras to capture individually identifiable animals has resulted in their use for estimating demographic parameters (Karanth 1995, Karanth et al. 2006). A major challenge however is to design a study so as to arrive at precise estimates of population size using capture-recapture models. Closed population estimators require that the population under investigation be both demographically and geographically closed. While demographic closure can be assured by sampling for a relative short duration in relation to the life span of the animal under investigation, geographic closure would have to be ensured by trap placement. The results of the regression analysis performed here provide a logical approach to assessing spatial boundary; however we suggest that the spatial limits of trapping be initially delineated by natural topographic boundaries wherever possible keeping in mind that increased trapping area aids in minimizing closure violation. As trap spacing governs recapture probabilities, we also recommend that investigators have an apriori knowledge the biology of the animal in concern since placement of traps is aimed at maximising (in time and space) the probability of capture. Though this study is based on a small population of tigers, we feel that the results highlight the need to design live capture-recapture with apriori knowledge of species biology to attain desired levels of precision.

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Appendix I

Location (GPS) of start and end points of the sampled streambed (*raus*) in Rajaji National Park.

	Range	Rau	Distance (km)	Start		End	
West RNP	Chillawali	Chillawali	4	30.145758	N	30.175075	N
				77.928106	E	77.960222	E
		Gaaj	4	30.132858	N	30.149975	N
				77.942117	E	77.980153	E
		Binj	4	30.120306	N	30.127181	N
				77.953728	E	77.994375	E
	Dholkhand west	Guleria	3.25	30.099228	N	30.110356	N
				77.986222	E	78.007933	E
		Malowali	4	30.082331	N	30.094683	N
				77.981039	E	78.018853	E
	Dholkhand east	Baam	4	30.062114	N	30.081072	N
				77.992869	E	78.016700	E
		Sindhli	3	30.049106	N	30.053969	N
				78.003172	E	78.021744	E
		Beribada	4	30.028033	N	30.072103	N
				78.025003	E	78.036192	E
	Ranipur	Harnol	4	29.995100	N	30.020308	N
				78.045558	E	78.059844	E
		Rauli	3	29.967789	N	29.984369	N
				78.081417	E	78.092869	E
		Ranipur	2.75	29.955869	N	29.972311	N
				78.099722	E	78.108303	E
East RNP	Ghauri	Dogudda	5	30.004508	N	29.981883	N
				78.249844	E	78.282336	E
		Ganga	5	29.983033	N	30.025200	N
				78.224622	E	78.253625	E
	Chilla	Ghasiram	5	29.962492	N	29.933994	N
				78.204200	E	78.237375	E
		Mundal	5	29.975006	N	29.957008	N
				78.214844	E	78.252431	E
		Gara	5	29.951319	N	29.961928	N
				78.263344	E	78.299397	E
		Amgadi	5	29.941139	N	29.955728	N
				78.274647	E	78.305686	E
		Khara	5	29.886003	N	29.924050	N
				78.262483	E	78.274711	E
		Luni	5	29.890175	N	29.908267	N
				78.284419	E	78.345083	E

Appendix II

Location (GPS) of start and end points of the 24 permanent transects laid in Rajaji National Park.

Range	Transect ID	Length (m)	Longitude	Latitude
Chilla	T1 (Start)		78.21518	29.96775
	T1 (End)	2490	78.23721	29.95588
	T2 (Start)		78.21106	29.95957
	T2 (End)	930	78.21668	29.95258
	T3 (Start)		78.25647	29.93346
	T3 (End)	1000	78.24807	29.93200
	T4 (Start)		78.27250	29.95410
	T4 (End)	910	78.27327	29.96246
	T5 (Start)		78.27672	29.94141
	T5 (End)	1160	78.28403	29.94946
	T6 (Start)		78.28940	29.93841
	T6 (End)	960	78.29557	29.94537
	T7 (Start)		78.27571	29.92266
	T7 (End)	1230	78.26480	29.92740
	T8 (Start)		78.27821	29.89867
	T8 (End)	2190	78.28380	29.91775
	T9 (Start)		78.30395	29.87567
	T9 (End)	1980	78.30518	29.89257
	T10 (Start)		78.23848	29.94141
	T10 (End)	1000	78.24033	29.95081
	T11 (Start)		78.23146	29.97414
	T11 (End)	1250	78.24019	29.97792

	T12 (Start)		78.21808	29.98324
	T12 (End)	1200	78.20601	29.98023
Chillawali	T13 (Start)		77.91297	30.16439
	T13 (End)	2000	77.90336	30.14948
	T14 (Start)		77.94614	30.14270
	T14 (End)	1700	77.95153	30.15566
	T15 (Start)		77.15315	30.12504
	T15 (End)	1000	77.95912	30.13152
Dholkhand west	T16 (Start)		77.97421	30.09923
	T16 (End)	2000	77.95528	30.10244
	T17 (Start)		77.97828	30.09379
	T17 (End)	1900	77.96217	30.07999
	T18 (Start)		77.99105	30.09164
	T18 (End)	1100	77.98595	30.09588
Dholkhand east	T19 (Start)		77.99183	30.06363
	T19 (End)	1000	77.99355	30.07080
	T20 (Start)		77.99265	30.06034
	T20 (End)	2000	77.98816	30.04364
	T21 (Start)		78.02613	30.02632
	T21 (End)	1900	78.03877	30.01596
Ranipur	T22 (Start)		78.04435	29.99718
	T22 (End)	1400	78.05674	30.00275
	T23 (Start)		78.05232	29.98770
	T23 (End)	1700	78.06454	29.99523
	T24 (Start)		78.09070	29.96179
	T24 (End)	1300	78.10096	29.96637

Appendix III

Location (GPS) of camera traps placed (2006-2007) in east RNP.

Camera trap ID	Latitude	Longitude	Sampling block
B1CT01	29.964620	78.248220	1
B1CT02	29.970290	78.231860	1
B1CT03	29.968590	78.214540	1
B1CT04	29.963010	78.214280	1
B1CT05	29.953310	78.213860	1
B1CT06	29.959760	78.229630	1
B1CT07	29.942590	78.235600	1
B1CT08	29.943460	78.244620	1
B1CT09	29.931100	78.253250	1
B1CT10	29.927870	78.266800	1
B2CT01	29.955910	78.261320	2
B2CT02	29.952740	78.268720	2
B2CT03	29.962330	78.297460	2
B2CT04	29.962330	78.286940	2
B2CT05	29.954880	78.298170	2
B2CT06	29.938180	78.273620	2
B2CT07	29.948880	78.288920	2
B2CT08	29.942150	78.294140	2
B2CT09	29.934050	78.292710	2
B2CT10	29.931480	78.298910	2
B3CT01	29.925180	78.276840	3
B3CT02	29.918930	78.275410	3
B3CT03	29.903600	78.271150	3
B3CT04	29.899010	78.275070	3
B3CT05	29.891570	78.284820	3
B3CT06	29.907980	78.287800	3
B3CT07	29.899940	78.300840	3
B3CT08	29.909780	78.300750	3
B3CT09	29.913820	78.307480	3
B3CT10	29.930830	78.309520	3
B4CT01	29.989130	78.233020	4
B4CT02	30.004800	78.249840	4
B4CT03	29.981190	78.234010	4
B4CT04	29.997350	78.246080	4
B4CT05	29.992890	78.238320	4
B4CT06	29.994894	78.272714	4
B4CT07	29.988367	78.274289	4
B4CT08	29.995769	78.246889	4
B4CT09	29.986914	78.241233	4
B4CT10	29.980578	78.262808	4

Appendix IV

Location (GPS) of camera traps placed (2007) in west RNP.

Camera trap ID	Latitude	Longitude	Sampling block
WB1CT01	30.046411	78.011311	1
WB1CT02	30.105086	77.995072	1
WB1CT03	30.100406	78.007042	1
WB1CT04	30.130903	77.994358	1
WB1CT05	30.121269	78.003914	1
WB1CT06	30.070264	78.011469	1
WB1CT07	30.095750	78.002975	1
WB1CT08	30.087978	78.029408	1
WB1CT09	30.090553	78.021581	1
WB1CT10	30.072489	78.000792	1
WB2CT01	30.087532	78.028908	2
WB2CT02	30.039333	78.040839	2
WB2CT03	30.053397	78.025969	2
WB2CT04	30.055864	78.056586	2
WB2CT05	30.050436	78.031422	2
WB2CT06	30.058650	78.024347	2
WB2CT07	30.041597	78.030389	2
WB2CT08	30.106986	77.988608	2
WB2CT09	30.052389	78.015489	2
WB2CT10	30.035223	78.041239	2

Photographs of tigers captured during 2004 - 2005 in Chilla Range, Rajaji National Park

RT-001 - 27.05.2004 (B2/T01)

RT-002 - 13.06.2004 (B2/T01)

RT-002 - 01.07.2004 (B2/T4)

06.07.2004 (B2/T4)

RT-002 - 05.01.2005 (B3/T01)

RT-004 - 06.01.2005 (B3/T09)

RT-004 - 06.01.2005 (B3/T09)

RT-003 - 06.01.2005 (B3/T03)

RT-002 - 13.01.2005 (B1/T10)

RT-004 - 13.01.2005 (B1/T09)

RT-004 - 13.01.2005 (B1/T09)

RT-002 - 31.01.2005 (B2/T07)

Photographs of tigers captured during 2004 - 2005 in Chilla Range, Rajaji National Park

RT-002 - 02.02.2005 (B2/T01)

RT-002 - 09.02.2005 (B3/T10)

RT-005 - 13.02.2005 (B3/T04)

RT-005 - 16.02.2005 (B3/T07)

RT-004 - 22.02.2005 (B1/T10)

RT-002 - 22.02.2005 (B1/T06)

RT-004 - 23.02.2005 (B1/T07)

RT-004 - 03.03.2005 (B1/T05)

RT-003 - 03.03.2005 (B1/T03)

RT-005 - 08.03.2005 (B1/T04)

RT-005 - 08.03.2005 (B1/T04)

RT-002 - 25.03.2005 (B2/T03)

Photographs of tigers captured during 2004 - 2005 in Chilla Range, Rajaji National Park


RT-002 - 27.03.2005 (B2/T05)


RT-002 - 13.04.2005 (B3/T10)


RT-005 - 09.04.2005 (B3/T03)

Photographs of tigers captured during 2005 - 2006 in Chilla Range, Rajaji National Park


RT-006 - 27.12.2005 (B1/T4)


RT-006 - 27.12.2005 (B1/T3)


RT-008 - 29.12.2005 (B1/T5)


RT-009 - 04.01.2006 (B1/T7)


RT-005 - 08.01.2006 (B1/T10)


RT-005 - 09.01.2006 (B2/T8)


RT-006 - 14.01.2006 (B2/T4)


RT-005 - 18.01.2006 (B2/T9)


RT-006 - 25.01.2006 (B3/T1)


RT-006 - 29.01.2006 (B3/T10)


RT-005 - 30.01.2006 (B3/T10)


RT-005 - 31.01.2006 (B3/T5)

Appendix VI

Photographs of tigers captured during 2005 - 2006 in Chilla Range, Rajaji National Park


RT-006 - 01.02.2006 (B3/T10)


RT-007 - 01.02.2006 (B3/T7)


RT-007 - 01.02.2006 (B3/T7)


RT-007 - 03.02.2006 (B3/T10)


RT-007 - 03.02.2006 (B3/T10)


RT-006 - 04.02.2006 (B3/T7)


RT-006 - 07.02.2006 (B3/T7)


RT-007 - 09.02.2006 (B3/T7)


RT-010 - 18.02.2006 (B1/T3)


RT-005 - 28.02.2006 (B2/T4)


RT-008 - 02.03.2006 (B2/T4)


RT-005 - 19.03.2006 (B3/T7)

Photographs of tigers captured during 2005 - 2006 in Chilla Range, Rajaji National Park

21.03.2006 (B3/T7)

RT-011 - 06.04.2006 (Dogudda)

RT-011 - 06.04.2006 (Dogudda)

RT-012 - 10.04.2006 (4 no. nullah)

RT-012 - 10.04.2006 (4 no. nullah)

RT-012 - 10.04.2006 (4 no. nullah)

RT-012 - 10.04.2006 (4 no. nullah)

RT-012 - 10.04.2006 (4 no. nullah)

Photographs of tigers captured during January & February 2007 in Chilla Range, Rajaji National Park

RT-006

RT-006

RT-012

RT-006

RT-006

RT-006

RT-006

RT-006

RT-006

RT-005

RT-013

RT-013

Photographs of tigers captured during January & February 2007 in Chilla Range, Rajaji National Park


RT-013


RT-013


RT-013


RT-013


RT-014


RT-014


RT-014

RT-007


RT-007

RT-007


RT-015


West RNP female

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