ECOLOGY OF SYMPATRIC SMALL CARNIVORES IN MUDUMALAI TIGER RESERVE, TAMIL NADU

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DOCTOR OF PHILOSOPHY

IN

WILDLIFE SCIENCE

By

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CERTIFICATE

This is to certify that the thesis titled **"Ecology of sympatric small carnivores in Mudumalai Tiger Reserve, Tamil Nadu"** submitted for the award of degree of Doctor of Philosophy in Wildlife Science to Saurashtra University, Rajkot is a record of original and independent research work carried out by **Ms. Riddhika Kalle** under our guidance. No part of this thesis has been submitted in part or full to any other University/Institution for the award of any degree and it fulfils all the requirements laid down by Saurashtra University.

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I also certify that the research work was appreciated by all who remain present and there was no comments made for this research work/comments made are incorporated in the thesis.

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(Riddhika Kalle)

EXECUTIVE SUMMARY

Habitat features may determine spatial distribution of small carnivores in an area since they exploit a variety of habitats and resources where resource partitioning is the outcome of species coexistence especially for sympatric species. This study aimed to examine the abundance and occupancy of small carnivores with the application of photographic captures using camera traps and diet composition of small carnivores through faecal analysis, investigate seasonal species-habitat relationships from microhabitat (non-spatial) and remotely sensed features (spatial) to understand how these variables govern their occurrence pattern and predict the distribution of small carnivores in Mudumalai Tiger Reserve, Western Ghats. Without careful consideration of population dynamics and habitat associations across species and habitats, conservation efforts may be poorly applied, thus delaying species recovery. This study will critically improve our state of knowledge to protect lesser carnivores and aid the development of theory on their social structure, behavior and habitat preferences. The study will help in suggesting localized monitoring mechanisms wherever necessary as this is important in minimizing surprise cases of extinction. My study species include jungle cat (*Felis chaus*), leopard cat (*Prionailurus bengalensis*), rusty spotted cat (*Prionailurus rubiginosus*), common palm civet (*Paradoxurus hermaphroditus*), small Indian civet (*Viverricula indica*), brown palm civet (*Paradoxurus jerdoni*), grey mongoose (*Herpestes erdwardsii*), ruddy mongoose (*Herpestes smithii*) and stripe-necked mongoose (*Herpestes vitticollis*). The objectives of this study area as follows:-

1. To determine the abundance and occupancy of small carnivores in Mudumalai Tiger Reserve.
2. To study the dietary pattern of small carnivores.
3. To determine spatial and non-spatial variables governing the occupancy of small carnivores and
4. To predict the distribution pattern of small carnivores through habitat suitability modeling in Mudumalai Tiger Reserve.

The study was conducted from 2009-2011 in Mudumalai Tiger Reserve (11°32'-11°43'N; 76°22'-76°45'E), Western Ghats. I selected 114 km^2 of intensive study area which was divided into three intensive sampling zones; deciduous (35 km^2), semi-evergreen (40 km^2) and the dry thorn forest (39 km^2) to conduct camera trap surveys for two years (2009-2010 and 2010-2011). I deployed passive-infrared camera traps in a systematic grid 1 x 1 km^2 using Deercam DC300 (DeerCam, Park Falls, USA) and Stealthcam (Bedford, Texas, USA). The mean inter-camera trap distance was 1.31 km. Each year I deployed 25 pairs of camera traps in the deciduous forest, 21 in the semi-evergreen forest and 25 in the dry thorn forest. Each camera trap survey was sampled for 30 days.

I calculated relative abundance index (here after called RAI) for each camera trap location by dividing the total number of independent records from the total trap nights x 100 (Carbone et al., 2001, O'Brien et al., 2003). Hence the RAI for each species is presented as the number of photo captures per 100 trap nights. To compare RAI of a species between habitats and seasons, I used Student's t-test in program SPSS 16 (SPSS 2007). I used Royle and Nichols (2003) heterogeneity model and repeated-count method (Royle 2004). Both models were analyzed in program Presence 4.1 (http://www.mbrpwrc.usgs.gov/software/presence.html).

I looked at differences in detection probability, occupancy rates and average abundance between seasons in each habitat. Mean abundance estimates were compared between habitats and seasons using Student's t-tests in program SPSS 16 (SPSS 2007). I performed a Pearson correlation test between mean abundance (λ) as obtained from occupancy approach and relative abundance index (captures/100 trap nights). Density of jungle cat was estimated only from camera trap captures in the dry thorn forest due to insufficient captures in the other sampling zones. Due to the difference in right and left flank pictures, I used the flank which yielded maximum unique individuals for abundance estimation. The sampling effort for each survey was 690 trap nights (23 locations × 30 days). Individuals were distinguished in photographs by their distinctive pelage pattern, body shape or sex after which I constructed capture histories necessary for closed capture-recapture analyses. I estimated the population size using software CAPTURE (Rexstad and Burnham 1991), considering the results of model M_h, and assuming heterogeneity among individuals (White et al., 1982). I also tested the closure assumption by applying a closure test implemented in CAPTURE. The density was also estimated using maximum likelihood-based spatially explicit capture–recapture model (ML-SECR) in program DENSITY 4.45 (Efford 2009).

Totally, 7380 trap-nights yielded 448 photographs of nine small carnivore species of which 16.07 % (n = 72) were of jungle cat, 1.34 % (n = 6) leopard cat, 2.46 % (n = 11) rusty-spotted cat, 19.87 % (n = 89) small Indian civet, 8.26 % (n = 37) common palm civet, 4.46 % (n = 20) brown palm civet, 14.73 % (n = 66) stripe-necked mongoose, 21.43 % (n = 96) ruddy mongoose and 11.38 % (n = 51) grey mongoose. Average capture frequencies of small carnivores across both years ranged from 0.13 captures/100 trap-nights to 2.56 captures/100 trap-nights. Capture frequencies for jungle cat ranged from 0.13-3.33 captures/100 trap-nights, leopard cat 0.13-0.4 captures/100 trap-nights, rusty-spotted cat 0.13-0.51 captures/100 trap-nights, small Indian civet 0.63-2.38 captures/100 trap-nights, common palm civet 0.4-1.15 captures/100 trap-nights, brown palm civet 0.13-1.9 captures/100 trap-nights, stripe-necked mongoose 0.13-3.47 captures/100 trap-nights, ruddy mongoose 0.53-1.87 captures/100 trap-nights and grey mongoose 0.13-2.18 captures/100 trap-nights. Grey mongoose RAI was significantly higher in dry thorn than deciduous forest (t test, t = -12.76, df = 1, P = 0.02) in the dry season. Detection probability of small Indian civet in the dry season was higher in dry thorn than deciduous forest (t test, t = -3.06, df = 1, P = 0.04) and higher in semi-evergreen forest than deciduous (t test, t = -6.70, df = 1, P = 0.05) however it was significantly higher in semi-evergreen than dry thorn forest (t test, t = -4.69, df = 1, P = 0.04). Common palm civet detection probability was higher in the deciduous than dry thorn forest (t test, t = -5.92, df = 1, P = 0.05) in the dry season. Habitat-wise estimates of occupancy varied from 0.04 to 0.65, with standard errors from 0.04 to 0.3 and average abundance ranged from 0.04 to 1.13 with standard errors from 0.04 to 0.7. Modeled site occupancies were greater than naïve occupancy. Jungle cat occupancy was significantly higher in the wet season than dry season (t test, t = 27.01, df =1, P =< 0.001) in the thorn forest. Occupancy of stripe-necked mongoose was higher in the dry season than wet season (t test, t = -5.73, df = 1, P = 0.01) in deciduous forest.

The sampling effort of 690 trap-nights for 2010 and 2011 yielded 66 pictures of jungle cat (16 right flanked and 19 left flanked in 2010 and 13 right flanked and 7 left flanked in 2011) from which, I obtained encounter histories for 6 individuals from left flank in 2010 and 7 from right flank in 2011. The closure test conducted in program CAPTURE was consistent with the assumption that the jungle cat population for 2010

(z = -0.382, P = 0.35) and 2011 (z = -1.217, P = 0.11) was closed. The mean maximum distances moved (MMDM) by individual jungle cats detected at multiple sampling stations in 2010 (n = 2) was 2.87 km and 3.23 km (n = 2) in 2011. Effectively sampled areas from traditional approaches under different buffer ranged from 35.5 to 150 km^2. In 2010 the capture probability (*p-hat*) of jungle cat was 0.22 for left flank; population size *(N)* under model M_h was 6.6 (SE = ± 2.5) and in 2011 capture probability was 0.14 for right flank; population size *(N)* was 9.3 (SE = ± 3.3) in the sampled area. Spatially explicit maximum likelihood density estimate for 2010 was 0.08 (SE = ± 0.04) individuals/km^2 and 0.09 (SE = ± 0.05) individuals/km^2 in 2011. Jungle cat density from MMDM models ranged from 0.04 (SE = ± 0.02) to 0.11(± 0.037) individuals/km^2 depending on the buffer applied and the effective sample area. Abundance estimate from Royle and Nichol's Heterogeneity model (hereafter called RN Heterogeneity, on an average 0.47 (± 0.22) individuals/km^2) was higher than other estimates. The jungle cat sex ratio male (M):female (F) estimated at the reserve was 1:1.5 in 2010 and 1:1.2 in 2011. At a camera station where more than one individual was detected, the sex ratio was 1M:1F, 1M:2F and in 2010 and 1M:1F in 2011.

Data showed that body weight had a significant positive relation with home range (km^2) (Pearson correlation; R^2 = 0.75, n = 6, P = 0.04) while there was no correlation with detection probability (Pearson correlation; R^2 = 0.32, n = 8, P > 0.05), site occupancy (Pearson correlation; R^2 = -0.13, n = 8, P > 0.05) and RAI (Pearson correlation; R^2 = 0.21, n = 8, P > 0.05). There was a significant positive relationship between mean abundance (λ) and RAI (Pearson correlation; R^2 =0.83, n = 8, P = 0.005).

A total of 19 plant species (10 trees, six shrubs, two lianas, and one climber) were tagged and monitored twice a month, throughout the study period. For fruit biomass estimation, fruit production of the tagged trees was monitored every two weeks. Total fruit biomass per tree was estimated by weighing the total number of fruits on the maximum fruiting branch and then multiplying this value by the number of similar sized fruiting branches. The number of ripe fruits on representative branches was counted on every visit. I estimated the availability in g/m^2 of fruits for each month during the study period. Density of rodents and shrews were estimated using a trapping web design. Rodent trapping was conducted in 72 locations to assess seasonal changes in relative abundance. Totally 72 trapping webs (26 in dry thorn forest, 25 in deciduous and 21 in semi-evergreen forest) using 32 Sherman traps (H. B. Sherman Traps, Inc., Tallahassee, Florida) were laid around each camera location in the reserve. Trapping amounted to a total effort of 4416 trap-nights in the dry season and 3072 trap-nights in the wet season. Rodent and shrew density was estimated using program Distance 6.0 (Thomas et al., 2009). The best model was selected on the basis of the lowest Akaike Information Criteria (AIC) values (Burnham and Anderson 1998).

To monitor seasonal variation in abundance of ground-dwelling birds and hare, the intensive study area was divided into 1 km x 1 km grids and in each grid, a line transect (1 km) was laid (n = 69, total length = 69 km). Each transect was walked thrice in dry (Feb-May) and wet seasons (Sept-Dec) for two years (2009-2011). The total effort amounted to 702 km. All transects were walked in the early morning between 6.30 am to 8.30 am after sunrise. For every sighting, time, species, group/flock size, animal bearing (using a hand held compass) and the angular sighting distance (using a laser range finder) were recorded. Hare and bird species density was estimated using program Distance 6.0 (Thomas et al., 2009). Density was estimated separately for dry and wet season in each habitat. Densities were compared between habitats and seasons using Student's t-test (Zar 1999).

For dietary analysis, I established 36 trails (3-5 km) where each trail was surveyed systematically twice a month, along which small carnivore faecal samples were collected. I calculated biomass of different prey types consumed per predator by multiplying the average prey biomass with number of prey items consumed by small carnivores. I estimated the diversity of each predator's diet using the Shannon diversity index (Magurran 2004), randomizing the original order of scat samples (1000 iterations) using the software EstimateS (Colwell 2006). I expressed diet composition in 3 complementary ways: percentage of the fresh biomass consumed (PB), percentage of occurrence (PO), and relative frequency of occurrence (RFO).

I used Wilcoxon paired sample test (Wilcoxon 1945) to determine differences in observed frequencies of occurrence between food items and to determine differences in frequency occurrence and percent biomass between habitats. Diet composition (relative frequency of occurrence) of major food items (prey items were pooled) was compared between habitats by χ^2 contingency table analysis (Zar 1999). Mann-Whitney *U* test was applied to check for seasonal differences between species.

Shannon's (Pielou 1966) diversity index (*H'*) and Levins' (1968) index ($B = 1/\sum pi^2$) was also calculated. To assess similarity of food composition between small cat, civet and mongoose, the Pianka's niche overlap index was calculated (Pianka 1973) for dry and wet season. I used Jacob's index (*D*) to determine food selectivity of small carnivores. The Bonferroni simultaneous-confidence-interval approach of Marcum and Loftsgarden (1980) was applied to assess which food item was selected preferentially with ≤ 0.05 significance. I used SPSS 16 (SPSS 2007) statistical package to process data.

Grewia tiliifolia, *Grewia hirsuta* and *Lantana camara* produced fruits throughout the year. Some species like *Zizyphus mauritiana*, *Cassia fistula, Coffee spp*, *Zizyphus oenophilia* showed bimodal fruiting pattern, producing fruits from Feb-May and Oct-Dec. *Bischovia javanica*, *Piper nigrum*, *Diospyros montana*, *Symplocos cochinchinensis*, *Dillenia pentagyna*, *Embelia gardeneriana*, *Ficus spp*, *Gnetum ula* and *Cordia obliqua*, produced fruits exclusively from Feb-April indicating a short fruiting period. Species that fruited exclusively during the wet season were *Syzigium cumini* and *Bridelia crenulata*. *Antidesma zeylanicum* began fruiting at the end of the second wet season in December and continued fruiting till May.

Rodent abundance was higher in the dry season (95.25 individuals/ha) than wet season (65.98 individuals/ha).

***Dry season**:* In the dry season *Mus platythrix* was the most abundant species (19.2 ± 5.8 individuals/ha) in the thorn and deciduous forest (44.88 ± 10.27 individuals/ha). *Rattus rattus wroughtoni* was the most abundant species in the semi-evergreen forest (72.08 ± 6.44 individuals/ha). Across the reserve, *Mus platythrix* was the most abundant species (41.91 ± 9.09 individuals/ha).

***Wet season**:* In the thorn forest, *Mus booduga* was the most abundant species (38 ± 13.21 individuals/ha), in the deciduous forest *Cremnomys blanfordi* was the most abundant species (39.98 ± 5.71 individuals/ha) while *Millardia meltada* (0.74 ± 0.12 individuals/ha) was the least abundant species. Across the reserve *Cremnomys blanfordi* was found to be the most abundant species (18.57 ± 3.02 individuals/ha).

A total of 355 faeces of small cats were collected from dry thorn (n = 258) and deciduous (n = 97) forests, of these there were 831 civet faeces; dry thorn (n = 205), deciduous (n = 320) and semi-evergreen (n = 306) forests and 708 mongoose faeces;

dry thorn (n = 281), deciduous (n = 333) and semi-evergreen (n = 94) forests. In small cat faeces, small mammals were found to be the most common food items, accounting for more than 50.73% of the materials in fecal samples and 82.87% of the ingested biomass followed by invertebrates that accounted for about 21.29% of the materials in fecal samples and 1.66% of the ingested biomass. In civet faeces, fruits were the most common food items, accounting for more than 69.25% of the materials in fecal samples and 33.63% of the ingested biomass followed by invertebrates that accounted for about 16.77% of the materials in fecal samples and 5.91% of the ingested biomass. The key plant species of the civet's diet were *Grewia tiliifolia*, *Cassia fistula*, *Zizyphus oenophilia*, *Lantana camara*, *Cordia spp*, *Antidesma zeylanicum*, *Coffee spp*, *Embelia gardneriana*, *Bischovia javanica*, *Gnetum ula*, *Ficus spp*, *Symplocos cochinchinensis*, *Bridelia crenulata*, *Syzigium cumini*, *Caryota urens*, *Piper nigrum*, *Dillenia pentagyna*, *Diospyros montana*, *Zizyphus mauritiana*, and *Grewia tiliifolia*. The proportion of small sized fruits (0.3-3.5 cm) was high in the diet except for *Cassia fistula* fruits. In the diet of mongoose, invertebrates were the most common food items, accounting for more than 48.49% of the materials in fecal samples and 8% of the ingested biomass followed by small mammal that accounted for about 18.29% of the materials in fecal samples and 56.38% of the ingested biomass.

The correlation between ingested percent biomass by civet and available biomass in the environment revealed a positive trend (R^2 = 0.87, P = 0.02) in the dry season. However fruit diversity in the environment was weakly correlated with fruit diversity in faeces (R^2= 0.55, P = 0.06). Mongooses consumed pilli millipedes and dung beetles with greater frequency than other carnivores. Fruit occurrence in civet faeces was higher in the dry season than wet season (Mann–Whitney U test: *U* = 362.5, P = 0.04).

Civet and mongoose consumed small sized food items (1-10 g) in equal proportion and comparatively higher than small cats, on the other hand large sized prey (> 30 g) were consumed higher by small cats than civet and mongoose. Niche overlap between civet and mongoose and between cat and mongoose was high. Niche overlap between civet and mongoose was higher in the dry season than wet season. Niche breadth was highest in mongoose followed by civet and small cat.

Small cats selected *Tatera indica*, *Millardia meltada* and *Suncus montanus* positively and significantly more than their availability in the dry season. In the wet season small cats selected *Tatera indica* and *Millardia meltada* positively and significantly.

Civets selected unidentified rodent, *Grewia tiliifolia* and *Cassia fistula* positively and significantly more than its availability in the dry season. In the wet season, civets selected unidentified rodent, *Grewia tiliifolia*, *Lantana camara*, *Zizyphus oenophilia*, *Cassia fistula*, and *Diospyros montana* positively and significantly more than its availability. Mongoose selected unidentified rodent and *Millardia meltada* positively and significantly more in the dry season. In the wet season, mongoose selected *Millardia meltada* and *Suncus montanus* positively and significantly more.

Habitat use by small carnivores incorporates micro-scale (ground habitat features) and macro-scale (remotely sensed) variables into occupancy models. *Continuous variables* involved number of *Grewia tiliifolia* and *Zizyphus spp* trees, active termite mounds, small mammal richness, fruiting tree diversity, canopy cover, shrub cover, rock cover, ground cover, coffee shade-understory cover, distance to the nearest water source and village/settlement. *Categorical variables*- included presence/absence of major habitat types (dry thorn, deciduous, semi-evergreen and riparian), terrain, prey species, liana and vine at each camera station.

Remotely sensed variables were extracted using the Geographic Information System (GIS) in ArcGIS 9.3 (ESRI, Inc., Redlands, CA, USA). These included mean NDVI, elevation, slope, aspect, bioclimatic variables (Hijmans et al., 2005), surface water bodies, distance to water and village.

For each species, I fitted multiple models, representing different hypotheses about the processes that generated the data. Occupancy surveys described in MacKenzie et al., (2002) and Royle and Nichols (2003) use sample units as sites. The survey duration of 30 days was divided into 10 sampling periods of 3 days each to increase detection probability for each sampling period. Detection histories were pooled for two years from the 3 study sites and entered together along with variables as single-season models in PRESENCE v.4 (Proteus Wildlife Research Consultants, New Zealand; http://www.proteus.co.nz).

Jungle cat: The top-ranked model for the dry season (microscale), included rock cover and canopy cover and had an Akaike weight (w_i) = 0.20, implying a 20% chance of being the best of the models evaluated. The top ranked model in the wet season (microscale), included the thorn forest and had an Akaike weight (w_i) = 0.17 implying a 17% chance of being the best of the models evaluated. The top ranked model for the dry season (macroscale), included distance to village with an Akaike weight (w_i) = 0.09.

Rusty spotted cat: The top-ranked model for rusty spotted cat in the dry season (microscale), included thorn forest with an Akaike weight (w_i) = 0.11. The top ranked model for rusty spotted cat in the wet season (microscale), included *Tatera indica* with an Akaike weight (w_i) = 0.11. The top ranked model for rusty spotted cat in the dry season (macroscale), included precipitation of warmest quarter and slope with an Akaike weight (w_i) = 0.21.

Small Indian civet: The top-ranked model for small Indian civet in the dry season (microscale), included ground cover, vine, canopy cover, thorn forest, distance to water and rodent richness, with an Akaike weight (w_i) = 0.12. In the wet season (microscale), included thorn forest, distance to water, shrub cover and *Grewia tiliifolia* with an Akaike weight (w_i) = 0.55. In the dry season (macroscale) ΔAIC_C = 0.00, included precipitation of coldest quarter with an Akaike weight (w_i) = 0.36.

Common palm civet: The top-ranked model for common palm civet in the dry season (microscale), included riparian and distance to village/settlement with an Akaike weight (w_i) = 0.13. The top ranked model for common palm civet in the wet season (microscale), included distance to village and plain, with an Akaike weight (w_i) = 0.11. In the top ranked models, undulating terrain was positively associated while plain, shrub cover and distance to village were negatively associated with common palm civet occurrence. The top ranked model for common palm civet in the dry season (macroscale) ΔAIC_C = 0.00, included precipitation of warmest quarter and aspect, with an Akaike weight (w_i) = 0.18.

***Brown palm civet*:** The top-ranked model in the dry season (microscale), included steep, distance to village and distance to water, with an Akaike weight (w_i) = 0.27. The top-ranked model for brown palm civet in the dry season (macroscale) ΔAIC_C = 0.00, included aspect with an Akaike weight (w_i) = 0.27.

Stripe-necked mongoose: The top-ranked model for stripe-necked mongoose in the dry season (microscale), included rodent richness, canopy cover, and distance to water, with an Akaike weight (w_i) = 0.20. The top-ranked model for stripe necked mongoose in the wet season, included shrub cover and distance to water, with an Akaike weight (w_i) = 0.34. The top-ranked model for stripe necked mongoose in the dry season (macroscale) ΔAIC_C = 0.00, included precipitation of coldest quarter and NDVI, with an Akaike weight (w_i) = 0.17.

Ruddy mongoose: The top-ranked model for ruddy mongoose in the dry season (microscale), included deciduous forest and ground cover, with an Akaike weight (w_i) = 0.25. The top-ranked model for ruddy mongoose in the wet season, included distance to water and shrub cover, with an Akaike weight (w_i) = 0.09. The top-ranked model for ruddy mongoose in the dry season (macroscale), included precipitation of warmest quarter with an Akaike weight (w_i) = 0.19.

Grey mongoose: The top-ranked model for grey mongoose in the dry season (microscale), included deciduous and canopy cover, with an Akaike weight (w_i) = 0.15. The top-ranked model for grey mongoose in the wet season, included fruiting tree diversity and termite, with Akaike weight (w_i) = 0.15. The top-ranked model for grey mongoose in the dry season (macroscale), included temperature of warmest month and precipitation of warmest quarter, with Akaike weight (w_i) = 0.18.

For predictive habitat modeling, species distribution data was collected from camera trapping (methodology has been explained in chapter 3), direct sighting records, interviews with forest staff, local people (from villages and settlements, resort owners), and road kill incidents. Remotely sensed variables included classified forest cover from Forest Survey of India, land use land cover map (DIVAGIS version 7.1.7.2, http://www.diva-gis.org), elevation, slope, aspect, NDVI, bioclimatic variables, human influence index (Last of the WILD DATA v. 2 2005), actual evapo-transpiration, surface water bodies (rivers and streams) and distance to water and village. I used MaxEnt software (Phillips et al., 2006) because it performs better than other presence-only modeling techniques (Elith et al., 2006). A set of ascii environmental layers and a .csv file of known locations of a species were used to produce probability maps that predict the potential distribution of a species. As a result, the final reduced data set used in this study converged to a total of 17 environmental layers which were projected to the UTM zone to match their coordinates, clipped to the extent of the boundary along with 2 km buffer, resampled to the cell size of 30 arc-seconds (~ 1 km^2), and entered with the occurrence data into MaxEnt version 3.3.3 (http://www. cs.princeton.edu/~schapire/MaxEnt). For the variables with highest predictive value, I examined the response curves showing how each of these environmental variables affects the MaxEnt prediction (Phillips and Dudik 2008).

***Jungle cat*:** Annual precipitation had the highest predictive power (46%) to the model output for jungle cat. NDVI (March) was negatively related to predicted jungle cat presence, aspect and distance to water response curves showed bi-modal distribution, the response curve for elevation showed that highest predicted suitability areas in areas of low to medium elevation (200 – 900 m). The land cover categories with the highest predicted probability of jungle cat included deciduous and degraded areas in the south-eastern part of the reserve. The annual precipitation of the warmest quarter showed a skewed response curve with suitable conditions towards 210 mm of precipitation. Predicted jungle cat distribution in Mudumalai accounted for 38 km^2 of suitable habitat in the reserve with the Linear Quadratic Hinge feature type models.

***Rusty-spotted cat*:** Land cover type had the highest predictive power (30.4%) to the model output. Aspect, wetness index and annual precipitation of the warmest quarter were negatively related to predicted rusty-spotted cat presence. The response curve for elevation showed a positive relationship with predicted rusty-spotted cat suitability where high probabilities occurred at higher elevations (> 1200 m). The land cover categories with the highest predicted probability for rusty-spotted cat as calculated includes only degraded areas in the south-eastern part of the reserve. The response curve for annual precipitation of the warmest quarter showed high predicted suitability in areas with low precipitation 150-220 mm. The MaxEnt model generated a map of potential rusty-spotted cat distribution in Mudumalai showing 56 km^2 of suitable habitat with the Linear Quadratic feature type models.

***Leopard cat*:** Land cover type had the highest predictive power (67.8%) to the model output. Elevation and isothermality were positively related to predicted leopard cat presence. The land cover and forest type categories with the highest predicted probability for leopard cat as calculated includes sub-tropical evergreen and dense forests of the reserve. The MaxEnt model generated a map of potential leopard cat distribution in Mudumalai showing that high probability areas accounted for portion 58 km^2 of the reserve with the Linear feature type model.

***Small Indian civet*:** Elevation had the highest predictive power (31.7%) to the model output. Medium aspect (100°) and elevation (400-900 m) led to high probabilities. Probabilities were skewed towards plain areas. High probabilities peaked at 100 mm of annual precipitation of the coldest quarter. The MaxEnt model generated a map of potential small Indian civet distribution in Mudumalai showing that high probability areas accounted for 49 km^2 of the reserve with the Linear feature type model.

***Common palm civet*:** Annual precipitation of the warmest quarter had the highest predictive power (39%) to the model output. Annual precipitation of the warmest quarter was skewed towards 210 mm, actual evapotranspiration peaked at 825 mm and elevation at 850 m. Common palm civet probabilities were predicted highest in non-forest areas and dry deciduous habitats. Predicted high probabilities were achieved from low to high NDVI. The MaxEnt model generated a map of potential common palm civet distribution in Mudumalai showing that high probability areas accounted for 23 km^2 of the reserve with the Linear Quadratic Hinge feature type model.

***Brown palm civet*:** Annual precipitation of the warmest quarter had the highest predictive power (39.7%) to the model output. Predicted probabilities were skewed towards high NDVI (March) and actual evapotranspiration and moderate aspect. Probabilities showed an S-shaped curve from low values and gradually achieving stabilization at higher elevation (> 1400 m) and a bell-shaped curved with NDVI. Annual precipitation of the warmest and coldest quarter showed skewed response curves towards low values (240 mm and 100 mm respectively). The MaxEnt model generated a map of potential brown palm civet distribution in Mudumalai showing that high probability areas accounted for 34 km^2 in the north-western part of the reserve with the Linear Quadratic Hinge feature type model.

***Stripe-necked mongoose*:** Elevation had the highest predictive power (41.7%) to the model output. Predicted probabilities were skewed towards high actual evapotranspiration. High probabilities occurred at medium aspect, elevation (200-900 m) and landcover types; subtropical evergreen and dry deciduous forests. Probabilities were skewed towards low annual precipitation of the warmest and coldest quarter. The MaxEnt model generated a map of potential stripe-necked mongoose distribution in Mudumalai showing that high probability areas accounted for 57 km^2 depicting a with the Linear Quadratic Hinge feature type model.

***Ruddy mongoose*:** Elevation had the highest predictive power (21.6%) to the model output. The response curve for NDVI (March) and distance to water showed a bi-modal curve, actual evapotranspiration showed a bell-shaped curve with probabilities at moderate values. Probabilities were skewed towards 40-60° of aspect, 220 mm of annual precipitation of the warmest quarter and 100 mm of the coldest quarter. Probabilities dropped abruptly at lowest topography wetness index. The MaxEnt model generated a map of potential ruddy distribution in Mudumalai showing that high probability areas accounted for 40 km^2 towards the eastern region in the reserve with the Linear Quadratic Hinge feature type model.

***Grey mongoose*:** Annual precipitation of the warmest quarter and landcover categories had the highest predictive power (34.5% and 33.3%) to the model output. The response curve for actual evapotranspiration showed a bell-shaped curve. Probabilities were skewed towards 900 m of elevation. High probabilities were achieved for degraded landcover type, skewed towards low annual precipitation of the warmest quarter (200 mm) and high NDVI. Predicted probabilities showed an S-shaped curve from low to medium isothermal temperature. The MaxEnt model generated a map of potential grey mongoose distribution in Mudumalai showing that high probability areas accounted for 38 km^2 towards the south-eastern region in the reserve with the Linear Quadratic Hinge feature type model.

Prediction models showed the study species occurrence in unsurveyed areas in Mudumalai Tiger Reserve, thereby demonstrating the potential of the reserve and surrounding regions to support populations due to the habitat heterogeneity and climatic gradient in the landscape. The species distribution maps will help managers to identify areas of suitable habitat for conservation of small carnivores. Interestingly, areas outside Mudumalai, in the buffer zone were also predicted suitable for small carnivores, although at variable proportion depending upon their response to variables. Forests in the buffer area are fragmented due to cultivations, farming practices, plantations, livestock grazing, fuel-wood extraction, establishment of resorts and weekend-homes, which does not necessarily guarantee the presence of small carnivores. Further ground-validation through future surveys in the buffer zone and adjoining Reserved Forests would give a better sense of actual presence. The study species appeared to be closely associated with climatic conditions and habitats to suit their ecological needs. This allows managers to preserve sufficient suitable habitat in order to sustain their populations in the near future through field management practices. Models can be tested in future field surveys across the Nilgiri Biosphere Reserve and validate suitability maps through ground monitoring of species presence. Habitat modification driven by anthropogenic activities and climate change may cause range contraction of sensitive species and expansion of those tolerant to disturbances. Protected Areas are surrounded by land used for livestock grazing, plantations, and development activities. It is feared that over the years climatic changes might lead to conversion of forests to open savannah woodland and reduction in dense evergreen forests in and around the Mudumalai landscape. Although this would favor carnivores like *F. chaus, P. rubiginosus, V. indica, P. hermaphroditus, H. smithii* and *H. edwardsii*, it may cause range contraction or population decline of specialists like *P. bengalensis*, *P. jerdoni* and *H. vitticollis*. Hence it is suggested that appropriate habitat restoration measures should be taken in the reserve keeping in mind the habitat requirements of species occupying highly specialized and sensitive habitats. Bioclimatic, topographical and anthropogenic data must be gathered in the long-term to monitor species response to variables and simultaneously monitor their population trend. Such modeling approaches play important roles in rapid assessment of conservation planning, particularly for data-poor species. I anticipate that this study will encourage researchers and conservationists to carry out similar work in other Protected Areas, fragmented forests, Reserved Forests, plantations, and urban landscapes in the country, as an initial step to record new distributional data on lesser carnivores, and update their natural history and population status. Although most of my study species are assigned, 'Least Concerned' status by IUCN, they seem to respond to disturbance and bioclimatic variables; this necessitates the need to conduct full-fledged studies to ascertain true response to disturbance factors.

Apart from this study and after having reviewed available literature on small carnivores in India, it appears that the basic ecological aspects; behaviour, den sites, movement patterns, home-range and population are lacking. Although most wildlife researchers are aware of these species and report important observations, through short-notes, articles, road-kill incidents and opportunistic observations, very few undertake the initiative of exploring research questions through intensive studies on these species. In such a scenario, it becomes essential for small carnivore ecologists to popularize their species or make it interesting enough for local masses, students, researchers and forest managers to express concern for them. The other alternative would be to carry out ground-breaking research and investigate species interactions with climatic changes and anthropogenic pressure or use molecular tools to prove their importance to understand how humans can benefit from these species. Linkage of ecological and social sciences to answer complex questions about human aspects, interactions and benefits from small carnivores is yet to be explored. I only hope that herein I have encouraged the process by highlighting species-specific ecology at a regional scale.

Chapter 1

INTRODUCTION

1.1 Introduction

Small carnivores of tropical forests represent an ecologically diverse and influential guild of forest vertebrates playing key roles as predators and seed dispersers. Small carnivores, as referred to generally small body sized carnivore species, have varied life histories showing highly species-specific responses to habitat and human-induced habitat changes, and attain varied conservation status. Lesser carnivores far outnumber large carnivores in species richness and are much more diverse in their behavior and ecology. These species can be solitary to highly social, frugivorous to hypercarnivorous, and habitat specialists of climax communities or generalists that live in close proximity to humans. Given their smaller size and ability to thrive in diverse habitats, they are usually more abundant than large carnivores, yet their impact within communities is generally assumed to be relatively minor. Indeed, where large carnivores have been lost or where lesser carnivores have been introduced, there are important examples of them driving the community structure. There are also instances in which lesser carnivores fill ecological role such as dispersers of seeds, that large, hypercarnivorous carnivores cannot. In most cases, lesser carnivores are not the top, or apex predators in an ecosystem, but there are instances, such as on islands or where they have been introduced, where they can fill the role of apex predator and perhaps cause community level cascades. In these unique circumstances, their collective ecological impact may be similar to and as profound as that of larger carnivores.

In species-rich communities, complex interactions among predators and their prey may emerge (Sih et al., 1998). Predators can act in an additive fashion, such that the summation of their effects on their prey is greater than that of any single predator alone. The widespread ascendancy of lesser carnivores to the role of apex predator is a relatively recent phenomenon engendered by anthropogenic driven extinctions and range contractions of large carnivores (Laliberte and Ripple 2004).

Small carnivore (herpestids, mustelids, viverrids) diversity is centered in three major regions in the Indian sub-continent, in the Eastern Himalaya, North-east Hills and the Western Ghats (Sterndale 1884, Pocock 1939, Nowak 1999). There are 33 species of small carnivores in India including a unique assemblage in the Western Ghats. Seventeen species of small carnivores (excluding members of Family Canidae) that belong to mammalian Families; Felidae (small cats), Mustelidae, Viverridae and Herpestidae are known to occur in the Western Ghats. They are as follows; jungle cat (*Felis chaus*), leopard cat (*Prionailurus bengalensis*), rusty spotted cat (*Prionailurus rubiginosus*), fishing cat (*Prionailurus viverrinus*), Eurasian otter (*Lutra lutra*), smooth coated otter (*Lutra perspicillata*), small-clawed otter (*Amblonyx cinerus*), Nilgiri marten (*Martes gwatkinsi*), honey badger (*Mellivora capensis*), small Indian civet (*Viverricula indica*), common palm civet (*Paradoxurus hermaphroditus*), brown palm civet (*Paradoxurus jerdoni*), Malabar civet (*Viverria civettina*), grey mongoose (*Herpestes edwardsii*), ruddy mongoose (*Herpestes smithii*), brown mongoose (*Herpestes brachyurus*) and stripe-necked mongoose (*Herpestes viticollis*). Behavioural and ecological studies of most small carnivores through direct observation in tropical forests is difficult because of their rarity and nocturnal habits. Deriving meaningful estimates of their abundance is a challenging task. As national parks and sanctuaries become subjected to greater human use, carnivores are severely affected by developmental activities, wildlife trade and hunting (Johnsingh 1986, Ashraf et al., 1993, Gupta 1997). There is a large void in the information available on the ecology of most of the small Indian carnivores (Mukherjee 1989, Mukherjee 1998). They are in urgent need of a focused conservation program which can achieve many successes with many challenges and identify a range of future work. They suffer from common issues of lack of awareness of their plight and the deficiency of reliable data on their distribution and conservation requirements.

To best plan landscape conservation strategies within the context of Mudumalai Tiger Reserve, it is critical to understand the population ecology of India's IUCN-listed small carnivores. The population status of these carnivores and their feeding ecology are of considerable interest as India's forests shrink, fragment, and degrade due to increasing demands from the growing human population. The outstanding rate of forest loss in India is dramatically changing the landscape composition through fragmentation, leaving many species of carnivores and their prey threatened with extinction. Without careful consideration of population dynamics across species and habitats, conservation efforts may be poorly applied, thus delaying species recovery. This will critically improve our state of knowledge to protect lesser carnivores and aid the development of theory on their social structure, behavior, and evolution. The study will help in suggesting localized monitoring mechanisms wherever necessary as this is very important in minimizing surprise cases of extinction. A new generation of camera traps with the use of well-developed population deriving models has led to an increase in the use of remote surveying and monitoring methodologies for elusive species from which population estimates and relative abundance indices can now be made (Karanth 1995, Carbone et al., 2001). Camera traps have also enabled more accurate estimates of species richness, diversity and spatial distribution of small mammals. With long-term use, these tools will enable monitoring changes in populations over time. Habitat features may determine spatial distribution of small carnivores in an area since they exploit a variety of habitats and resources where resource partitioning is the outcome of species coexistence (MacArthur 1972, Cody 1974, Diamond 1978, Tilman 1982) especially for sympatric species. This study aimed to examine the abundance and occupancy of small carnivores with the application of photographic captures using camera traps and diet composition of small carnivores through faecal analysis. I also investigated seasonal species-habitat relationships from microhabitat (non-spatial) and remotely sensed features (spatial) to understand how habitat variables govern their occurrence pattern. I also predicted the distribution of small carnivores in the intensive area of Mudumalai Tiger Reserve (hereafter called Mudumalai).

1.2 Small felids

In palaeogeological terms, the Felinae radiated relatively recently and rapidly in the late Miocene 13–14 million years ago, with extinct and extant genera of the family Felidae derived from a common ancestor 27 million years ago (Macdonald 1992). Cats can be classified on basis of their body size and body weight (large ≥20 kg, medium 10-20 kg and small ≤ 10 kg). India's six species of small cats; jungle cat (*Felis chaus*), Palla's cat (*Octocolobus manul*), desert cat (*Felis silvestris*), leopard cat (*Prionailurus bengalensis*), fishing cat (*Prionailurus viverrinus*), rusty spotted cat (*Prionailurus rubiginosus*) are up against odds. Basic knowledge on their biology and ecology is scarce. Small cats are hypercarnivores-obligate meat eaters. Much about the behavior of small cats could be learnt from their tail morphology. Interesting anecdotes of lesser cats in India provide little concrete evidence on their population status (Parihar 1989, Haque and Vijayan 1993, Patel and Jackson 2005, Manakadan and Sivakumar 2006, Patel 2006, Vyas et al., 2007). This is likely because of the extreme difficulty in physically observing them and the inaccessibility of their habitat. This is a common issue in conservation because the smaller elusive species are often the most difficult to study. Schauenberg (1978) reported jungle cat reproduction, Dal (1954) and Rathore and Thapar (1984) gave notes on its feeding behaviour. Habitat selection and seasonal changes in jungle cat body weight were outlined by Haptenr and Sludskii (1972) and Roberts (1977). Density estimates of small cats ranged from 0.07 to 1 individual/km^2 (Gardner et al., 2010, Oliveira et al., 2010, Gupta 2011, Reppucci et al., 2011). Kiltie (1988) examined jaw lengths in regional assemblages of felids and found that female jaw length was closely related to modal prey weight. Since felids use a killing bite to dispatch prey, species that feed upon larger prey should have concomitantly wider jaw gapes hence, jaw lengths (and probably other morphological parameters) appear to have evolved to maximize efficiency in handling and killing common prey species and to minimize overlap with adjacent sized species within the assemblage. Evenness in distribution of jaw lengths in regional assemblages of felid species may possibly indicate partitioning of resources within the felid guild. While this may well be an evidence of character displacement, jaw lengths could equally have evolved as a response to size distribution of available prey. In South America, margay, jaguarundi, oncilla (*Leopardus tigrinus*) and Geoffroy's cat (*L. geoffroyi*) occur at higher densities in the absence of larger ocelots, suggesting that ocelots may compete directly or indirectly with the smaller cats (Olviera et al., 2008). Black-footed cats and African wild cats may benefit from the removal of larger competitors such as caracals and jackals (*Canis mesomelas*) (Sliwa 2006). Caracals have been observed killing and partly eating even African wildcats suggesting inter-specific competition (Caro and Stoner 2003).

Worldwide, small cats inhabit specialized habitats. The pampas cat thrives in open habitats, ranging north through the dry forests and scrub grasslands of Bolivia, Paraguay, and Brazil, and up the Andes mountain chain from Chile to Ecuador and possibly marginally into southwestern Colombia (Silveira 1995). The Geoffroy's cat is distributed throughout the pampas grasslands and dry Chaco shrub and woodlands, and around the alpine saline desert of north-western Argentina to 3300 m in the Andes

(Nowell and Jackson 1996). Most of its range is arid or semi-arid (Pereira et al., 2006), but it also occurs in wetlands (Sunquist and Sunquist 2002). The guina (*Leopardus guigna*) is the smallest felid in the Americas, having a constricted distribution, occurring only in central and southern Chile and marginally in adjoining areas of Argentina. Although the guina is forest-dependent, selecting areas of thicket understory, they use a variety of more open scrub habitat types (Dunstone et al., 2002). In Central America and parts of northern South America the oncilla (*Leopardus tigrinus*) may be most common in montane cloud forest, it is mostly found in lowland areas of Brazil, being reported from rainforests to dry deciduous forest, savannahs, semi-arid thorny scrub, and degraded secondary vegetation in close proximity to human settlement (Oliveira et al., 2008). The margay (*Leopardus wiedii*) reaches its greatest abundance in lowland rainforest (Oliveira et al., 2008). It also occurs in dry deciduous forest (Nowell and Jackson 1996), but seems to be absent from the semi-arid caatinga scrub of Brazil, with the possible exception of some evergreen forest enclaves. The jaguarundi (*Puma yagouaroundi*) ranges from Mexico through Central America and the Amazon basin to central Argentina and Uruguay, occurring in closed primary rainforests to open desert, scrubs, and grasslands, although in open areas it sticks towards dense cover (Nowell and Jackson 1996). The sand cat (*Felis margarita*) which occurs exclusively in the desert is highly fossorial and is known to Saharan nomads as ‘the cat that digs holes’ (Dragesco-Joffe 1993). Sand cat scats contained the remains of Cairo spiny mouse (*Acomys cahirinus*) and gecko (*Stenodactylus spp.*). Sand-dwelling rodents made up the majority (65–88%) of its stomach contents from carcasses collected in Turkmenistan and Uzbekistan (Schauenberg 1974). The black-footed cat (*Felis nigripes*), Africa’s smallest felid, is endemic to the short grasslands of South Africa. The sex-specific difference in prey size consumption by this species could help reduce intraspecific competition between them (Sliwa 2006). The wildcat (*Felis silvestris*) has the widest distribution of any felid being found throughout the drier regions of Africa into Europe (including Scotland), southwest and central Asia, and Russia (Nowell and Jackson 1996). The Palla’s cat (*Otocolobus manul*) has a strong association with rocky, steep areas and rarely found in open grasslands. The flat-headed cat (*Prionailurus planiceps*) is closely associated with wetlands, to a greater degree than the fishing cat, with a much smaller distribution, found only on the islands of Borneo and Sumatra and the Malayan peninsula. Most collection records for the flat-headed cat are from swampy areas, oxbow lakes, and riverine forest (Nowell and Jackson 1996), peat-swamp forest and logged forest (Bezuijen 2003). Fishing cats are strongly associated with wetland, swamps and marshy areas, oxbow lakes, reed beds, tidal creeks, and mangrove areas, along watercourses at elevations up to 1525 m, lowland areas, near degraded habitats and aquaculture ponds in Calcutta (Nowell and Jackson 1996). The Andean cat (*Leopardus jacobita*) occurs only in association with rocky outcrops in the arid zones of the high Andes, typically above 4200 m, a specialist predator of chinchillids (*Lagidium spp*; Marino et al., 2011). Basic reproductive studies, using fecal hormone and semen analysis, have been initiated in small cat species to broaden our knowledge on species specific reproductive physiology to help improve their captive-breeding success (Swanson 2006). Small cats are considered to be dietary generalists with small mammals commonly comprising a major proportion of their food.

Wildcats are well adapted to live near human habitations probably because of the abundance of natural prey (rodents), and people noted that their presence was useful in killing rodents and protecting their grain stocks. European farmers benefit from colonies of feral cats around their barns, and treasured in their homes. Though it may seem like such cats are performing a great service to humanity, it is virtually at the cost of eradicating native wild prey. Fur trade is a serious threat to small cats, especially those with attractively marked pelts, such as the leopard cat, fishing cat and rusty spotted cat in India, and the South American species. Nevertheless, small scale trade continues, usually 'under the counter', and particularly in tourism zones. Seizures of large numbers of skins in various places in India from time to time are evidence of this clandestine trade. Given the secrecy of their lives, and the few records confirming their presence in various regions in the country, planning conservation measures for small cats presents considerable difficulty. Moreover, since they lack the glamour of the big cats, it is difficult to persuade governments to set aside areas only for small cats. However, they share the habitats of the big cats, and like their bigger cousins, their real habitats are natural areas of forest, grasslands and wetlands. Protection of these ecosystems is therefore an essential element in ensuring the future of small cats. It is essential that once the required information is gathered on these species, long term studies along with necessary conservation initiatives will ensure the survival of these cats. Efforts to study the lesser cats have not been made enough and there is a lot of work to be done before any conclusion can be made about their status in India. It is still not certain where and how many of them still exist.

Efforts should be made to identify and suggest localized monitoring mechanisms to track and monitor lesser cats.

1.3 Viverrids

The Viverridae consists of 13 genera and 35 species (Gaubert et al., 2005a, Gaubert et al., 2005b, Wozencraft 2005), about 50% of which are IUCN red-listed and/or poorly known (http://www.iucnredlist.org; Schreiber et al., 1989). The members of the family Viverridae are characterized by the presence of scent glands external to the anal region (Wozencraft 1989). Viverridae are confined to the Old World tropics. They were the only carnivores to reach Madagascar. They do not even occur as fossils, in the New World. They differ from the family Herpestidae in the sense that their anus is not enclosed in the glandular pouch and they have a peneal gland in the genital region (Pocock 1941). Most of the members have retractile claws. Their ears are comparatively larger with well developed bursa on the external margin. They are distinguished in the external characters from the members of the family Felidae by the hind foot being five toed, the retention of the inter-ramal tuft of facial vibrissae and typically elongated muzzle. Many of the members have spots or stripes on the body and the tail has ring like marks (Pocock 1939). In viverrids, the total length (head, body and tail) varies from 320 to 1850 mm, where as body weight varies from around 600 gm in the case of African linsang (*Poiana richardsoni*) to 20 kg in the case of binturong (*Arctitis binturong*). They are either terrestrial or arboreal in nature and have wide variety of diet including small mammals, birds, insects and fruits. Viverrids are mostly solitary and nocturnal. They have diversified into a variety of biological roles: some, such as palm civets (*Paradoxurinae*), are chiefly frugivores. Little precise knowledge is available on the ecological role of this diverse carnivore fauna, but it is certain to be significant. Viverrids retain a number of phylogenetically primitive characteristics which are of great scientific interest. The African civet (*Civettictis civetta*) is known for its valuable source of a musk-like substance called “civet”. This yellowish secretion has the consistency of butter and is a product from scent glands located near the civet’s anus. Civet musk mainly comes from Ethiopia, where “civet farmers” keep up to 60 wild-caught male civets in cylindrical cages made of branches. Musk is collected every 9-12 days with a horn spatula, each collection amounting to 10-15 gms per animal. There are an estimated

180 civet farmers in Ethiopia, holding a total of over 2700 animals (Schreiber et al., 1989). An animal produces about 800 gm civet per year, representing a value of 350 U.S. dollars (in early 1988). During 1975-1978, Ethiopia exported a total of 5830 tons of musk, mainly to France, where it was used as a raw substance in the perfume industry. As the animals were not captive-bred, the possible vast increases in production of this valuable substance appear to be limited.

The family Viverridae is divided into four subfamilies Cryptoproctinae, Viverrinae, Hemigalinae and Paradoxurinae (Wozencraft 1989). They are found only in tropical and subtropical Africa and Asia. Of the six subfamilies, the viverrids in India are represented by two subfamilies Viverrinae and Paradoxurinae with nine species. They represent small to medium-sized, nocturnal, solitary predators, often inhabiting inaccessible areas (Nowak 1999). Viverrids show more ecological diversification in trophic specialization and substrate use than any other family of carnivores (Eisenberg 1981). They are widely distributed throughout Europe, Asia, East Indies, Africa, and Madagascar, with upto eight species occurring sympatrically in tropical habitats (Medway 1977). Despite the large number of species of viverrids within the order Carnivora (13%) (Wemmer and Watling 1986) and their wide distribution throughout the Old World tropics, the basic biology of most species is unknown (Joshi et al., 1995). Most studies indicate that viverrids feed primarily on fruit and forage exclusively at night. Researches hypothesize that the social organization and activity patterns of these civets are shaped by the distribution of food resources. The palm civet is more arboreal than the small and large Indian civet (Bartels 1964). Palm civets are distributed throughout southern and eastern Asia in a wide variety of habitats (Lekagul and McNeely 1977). Rabinowitz (1991) reported home-range size of the palm civet from mixed-deciduous and dry, evergreen forest at Huai Kha Khaeng Wildlife Sanctuary, Thailand which was 12 times larger than that found in Chitwan National Park. More home-range overlap was observed when ripe fruiting trees were at low density or clumped in distribution (Joshi et al., 1995). The Malabar civet (*Viverra civettina*) is one of the several mammalian endemics in the evergreen rainforest belt of southwest India. The large Indian civet (*Viverra zibetha*) is quite common in the Terai belt along Himalayan foothills, and in the Eastern and Northeastern parts of India (Prater 1971, Acharjyo and Patnaik 1987, Choudhury 1997, 1999, Menon 2003, Peter 2003) and the Malabar civet which is a critically endangered species is restricted to the lowland (coastal) Western Ghats (Jerdon 1874, Ashraf et al., 1993, Rai and Kumar 1993). It was earlier regarded as a subspecies of *Viverra megaspila* (Ellerman and Scott 1951, Honacki et al., 1982), but Lindsay (1928) and Pocock (1941) reviewed its taxonomic position and considered it as a separate species *V. civettina*. It is also the only viverrid listed in Schedule I of the Indian Wildlife (Projection) Act, 1972. Its closest relative, the large-spotted civet (*Viverra megaspila*), sometimes regarded conspecific with *V. civettina*, ranges widely from Burma eastwards through southeast Asia. There were only two possible sight records by Karanth (1986) in the Kudremukh area in Chikmagalur district, Karnataka, and another one by Kurup (Zoological Survey of India) at Thiruvalla in Pathanamthitta district, Kerala, in the 1970s. *Viverra civettina* has probably disappeared from most of the coastal tracts and its continued existence in various parts of the Western Ghats needs confirmation. The only indication of a captive Malabar civet was provided by Pocock (1941) who described a skin obtained from Trivandrum Zoo. Dead wild specimens were confirmed by Kurup (Zoological Survey of India). The spotted linsang (*Prionodon pardicolor*) is distributed over a considerable part of southeast Asia but seems to be rare everywhere. The golden palm civet, (*Paradoxurus zeylonensis*) is indigenous to Sri Lanka, but closely related to the brown palm civet (Groves et al., 2009). The Malay civet (*Viverra tangalunga*) is found on the Malay Peninsula, Sumatra, Borneo, Sulawesi, Maluku Islands and the Philippines (Corbet and Hill 1992, Kanchanasakha et al., 1998, Suyanto et al., 1998) where it occurs in a wide variety of habitats including forests, secondary habitats, cultivated land and the outskirts of villages (Wemmer and Watling 1986, Nowak 1999). The large-spotted Civet (*Viverra megaspila*) occurred widely in mainland south-east Asia from southern China and Myanmar, through Indochina and Thailand south to Peninsular Malaysia (Corbet and Hill 1992), Myanmar, Thailand, Vietnam and Malaysia primarily in lowland forest (Holden and Neang 2009), Lao PDR and China. The most distinctive subspecies of the small-toothed palm civet (*Arctogaeidia trivirgata*) occurs in moist west Java, from the south coast up to 1700 m. The Mentawai palm civet (*Paradoxurus lignicolor*) is a part of the unique plant and animal community in the Mentawait islands. There are no confirmed records of this species in the wild apart from only museum records. The Mentawai Islands are the only small islands where the banded palm civet (*Hemigalus derbyanus minor*) occurs. It is otherwise known from the larger land masses of Sumatra, Borneo, Peninsular Malaysia, and Thailand. The Hose's palm civet (*Diplogale hosei*) is endemic to a few mountain ranges in Sarawak and Sabah (East Malaysia). The Sulawesi palm civet (*Macrogalidia musschenbroekii*) a monotypic genus, occurs mainly in primary forest from sea level to montane rain forest and cloud forest. Lowland populations of the Sulawesi palm civet could be adversely affected by timber cutting and cultivation. The Owston's palm civet (*Chrotogale owstoni*) is the sole living representative of its genus. It may be largely terrestrial, and prefers habitats in the vicinity of rivers in primary and secondary forests. The genus *Cynogale* contains only two species (usually treated as subspecies), the otter civet (*C. bennettii*) and Lowe's otter civet (*C. lowe*). They are semi-aquatic with specialized morphological adaptations such as broad, webbed feet and muscles which close the nose and ears. The Abyssinian genet (*Genetta abyssinica*) appears to have a wide altitudinal and ecological range, from coastal plains and open dry lowlands to montane heather moorlands and Afro-alpine grassland. Behrens and Rompaey (2002) provided records of this species in montane dry forest (dominant species; tree heath (*Erica arborea*), curry bush (*Hypericum revolutum*) and Abyssinian rose (*Rosa abyssinica*), and recorded them at 3750 m. The giant genet (*Genetta victoriae*) is the largest species of its genus occurring in forests between the Zaire, Lualaba and Oubangi rivers and the rift valley in northeast Zaire. The aquatic genet (*Osbornictis piscivora*), "Africa's rarest viverrid," is the sole living representative of its genus. The species is thought to hunt in water, but presumably unlike the otter civets, which are reported to lie in ambush, it is discontinuously distributed within its range, and there seems to be a correlation between its occurrence and large homogeneous stands of *Gilbertiodendron* forests (Schreiber et al., 1989). The Leighton's linsang (*Poiana richardsoni liberiensis*) is one of the carnivores with a very restricted range in parts of the Upper Guinea rainforest belt, demonstrating the importance of this region as one of the core areas for viverrid conservation in Africa.

The level of coverage of species suitable ranges by existing and proposed IUCN reserves was low, and it is suggested that the total surface of protected areas be substantially increased in Africa and Asia (Papes and Gaubert 2007). These animals generally have a restricted distribution, so options in choosing prime habitats for preservation are limited. Great extents of the areas identified as suitable protected areas should be augmented to conserve these small carnivores.

Published research (Guan et al., 2003) has so far implicated the masked palm civet (*Paguma larvata*), as possible sources of SARS-like coronaviruses. The masked palm civet, common palm civet and small Indian civet are the species most commonly found in wildlife restaurants across the southeast Asia, but all nine species are eaten under the generic label of ‘civet meat’ (Bell et al., 2004). It is also worth noting that within the three subfamilies of viverrids represented across Indochina, there are a further 14 African species that are exploited within the African bushmeat trade (Schreiber et al., 1989). In common with a range of other species across this region, several of these small carnivore species are now threatened with extinction as a result of overexploitation at unsustainable levels by an expanding international trade in wildlife. Seven out of 11 viverrids are listed as threatened in the 2000 Vietnam Red Data Book (the large spotted civet, the spotted Linsang, the small-toothed palm civet, the binturong, Owston’s palm civet, Lowe’s otter civet and the Taynguyen civet). The IUCN Species Survival Commision Action Plan for the conservation of viverrids 15 years ago highlighted habitat loss and fragmentation, particularly of tropical moist forests and wetland ecosystems, as the major threats (Schreiber et al., 1989). That important collation of information on the status and conservation requirements of these small carnivore groups flagged the urgent need for population surveys and research into the ecological requirements of these little-studied species. The Action Plan also warned that ‘the impact of hunting was growing with the rapid increase in human populations’, which ‘results in a decrease in habitat quality and the fragmentation of Viverrid population, and that this problem appeared to be greatest in the Upper Guinea rainforests and parts of Asia, such as China, Taiwan and Vietnam (Schreiber et al., 1989). The important point is that 20 years ago, even in Africa, while other ‘more important’ wildlife species were often sold by hunters at local markets, these small carnivore species tended to be consumed at home and were therefore regarded as opportunistically hunted subsistence food. Whether viverrids will survive in degraded or alternative habitats is of crucial importance for further conservation planning.

1.4 Herpestids

The members of the family Herpestidae are characterized by the uniquely derived nature of their anal sac and the structure of the auditory bulla (Wozencraft 1989). They are distinguished from the members of the family Viverridae by the presence of a naked glandular pouch around the anus, with the anal glands opening into the pouch. They have long non-retractile and fossorial claws. The ears have no marginal bursa. The penis, which although short as compared to viverrids, has a well developed baculum and is without penieal glands (Pocock 1939). In Herpestids, the total length (head, body and tail) varies from 200 to 1600 mm and the weight varies from around 340 gm. Mongooses are among those that are able to open hard-shelled food such as eggs, mollusks and crustaceans. The striped-necked mongoose (*Herpestes vitticollis*) is the largest Asiatic mongoose. Mongooses are terrestrial in nature and have a diverse diet including small mammals, birds, reptiles, crabs, fish and insects. They are mostly gregarious and a few are solitary too. The family Herpestidae is divided into three subfamilies Galiidinae, Herpestinae and Mungotinae (Wozencraft 1989). They occur in most of Africa and Asia. In India the family Herpestidae is represented by subfamily Herpestinae with seven species. Most ecological information on mongooses is available from North America and Madagascar, where introductions resulted in the species becoming invasive where densities can be expected to be fairly high. Mongooses were introduced to a variety of islands which lacked suitable predators to control agricultural pests or snakes; in several cases, especially on some Caribbean islands, they demonstrated the risks of biological pest control by not only killing selected pest targets, but also devastating native endemic species not adapted to coping with efficient predators. Ancient Egyptians kept the ichneumon (*Herpestes ichneumon*) to kill snakes. *Herpestes* had a great impact on ancient Egyptian thinking. The advent of poultry breeding to which mongooses were detrimental and thought to be the cause for the decrease in the numbers of ichneumons kept in the houses of north Africa. Mongooses also featured prominently in the Middle and Far Eastern religions, frequently as guardians of wealth.

The collared mongoose (*Herpestes semitorquatus*) is brightly coloured and occurs only in Sumatra and Borneo, small Indian mongoose (*Herpestes javanicus*) in Mauritius was found to have a mean home range of 0.77 km^2 (0.25–1.10) (Roy et al., 2002). Cape grey mongoose home range was found between 0.21-0.63 km^2 (Cavallini and Nel 1990). *Herpestes auropunctatus*, had a large proportion of fruits, vegetation and insects in the diet (0.85), with home ranges of 0.31 km^2 (Gittleman and Harvey 1981). Mean estimated density is only available for introduced populations of the small Indian mongoose, and as a result are likely to be higher than densities in range habitats. However, since density estimates from range countries are lacking, these may prove to be helpful priors: 6.4 animals per hectare in St. Croix, 2.4/ha in Jamaica and 4.6/ha in Puerto Rico (Horst et al., 2001) or 1.8 animals/ha using capture-recapture estimates in the West Indies (Corn and Conroy 1998). The yellow mongoose (*Cynictis penicillata*) dens together with conspecifics in groups ranged between two to 13 individuals in size, but individuals typically forage alone or in pairs (Rasa et al., 1992, Cavallini 1993). Females are polyestrous, occasionally giving birth to two litters per season, and young typically disperse during spring (Rasa et al., 1992). In South Africa yellow mongooses occupy large number of bolt-holes and burrows (Manser and Bell 2004).

1.5 Study species

1.5.1 Jungle cat

Distribution and status

Felis chaus is listed as Least Concern in the IUCN Red List of Threatened Species as it is widespread and common particularly in India. Once thought it to be closely related to the lynx, which share its characteristic traits of tufted ears, long limbs and a short tail (Sunquist and Sunquist 2002), but it is actually a close relative of the domestic cat. A wide range of subspecies (recognized nine) have been proposed by Wozencraft (2005).

The jungle cat has a broad but patchy distribution. Population decline and range contraction are of concern, particularly in Egypt, in the Caucasus, and in the southwestern, central and Southeast Asia. In India, it is the most common small cat (Mukherjee and Groves 2007). They inhabit savannas, tropical dry forests and reedbeds along rivers and lakes in the lowlands and are rare in rainforests. Although they are adaptable animals, being found even in dry steppe, they prefer wetland environments with tall grasses or reeds in which to hide. They do not survive well in cold climates, and are not found in areas where winter snowfall is common (Sunquist and Sunquist 2002). They have been observed from sea levels to altitudes of 8000 ft (2400 m) or perhaps higher in the Himalayas. They frequent jungles or open country, and are often seen in the neighborhood of villages (Blanford 1888–91). They live in altitudes up to 2500 m (8200 ft), but are more common in the lowlands.

Even though jungle cats were known to be absent from the Malayan peninsula, south of the Isthmus of Kra, the possibility of occurrence of the species was reported from a highly fragmented forest in the Malaysian state of Selangor (Sanei and Zakaria 2010). In the 1970s, Southeast Asian jungle cats still used to be the most common wild cats near villages in certain parts of northern Thailand and occurred in many protected areas of the country (Lekagul and McNeely 1988). But since the early 1990s, jungle cats are rarely encountered and have suffered drastic decline due to hunting and habitat destruction. Today, their official Thai status is critically endangered (Lynam et al., 2006). In Cambodia, Laos and Vietnam, jungle cats probably once occurred widely in the secondary habitats, which is easily accessible to hunters and where hunting pressure is now very heavy. Due to unselective trapping and snaring, jungle cats appear quite rare nowadays in comparison to sympatric small cats. Skins are occasionally recorded in border markets, and live individuals, possibly taken from Myanmar or Cambodia, occasionally turn up in the Khao Khieo and Chiang Mai zoos of Thailand (Duckworth et al., 2005).

Jungle cats are rare in the Middle East. In Jordan, they are highly affected by the expansion of agricultural areas around the river beds of Yarmouk and Jordan rivers, where they are hunted and poisoned by farmers for attacking poultry (Abu-Baker et al., 2003). In Afghanistan they are also considered rare and threatened (Habibi 2003). In Europe, it is of marginal occurrence, with small populations in Cis-Caspian region and the Caucasus along the Caspian Sea. The European population has been rapidly declining since the 1960s. There was no record of this species in Astrakhan State Reserve (Russian Federation) since the 1980s. Marked population fluctuations are characteristic of this species in this region, probably because of absence of adaptations to cold winters. Despite these fluctuations the long-term trend in Europe is of decline in both population and area of occupancy. Data from Russia suggest that there are about 500 animals left in the wild (Prisazhnyuk and Belousova 2007). A very small population persists in Georgia (Duckworth et al., 2008a). This species is considered threatened in a number of range states in Europe and the Caucasus, and is included in the Red Books of the Russian Federation, Armenia, Azerbaijan and Georgia (IUCN 2007).

Ecology and behavior

Jungle cats are generally solitary in nature. They rest in abandoned burrows, tree holes, and humid caves under swamp rocks, or in areas of dense vegetation. Although often active at night, they are less nocturnal than many other cats, and in cold weather may even bask on a sunny day. Allayarov (1964) detected and identified two jungle cat burrows as small holes inside dense reed thickets lined with old cane leaves and fur along rivers in Uzbekistan. They have been estimated to travel between 3 and 6 km (1.9 and 3.7 mi) per night, although this likely varies depending on the availability of prey. There is a striking difference in body size of jungle cats in the west and the east of their distribution, with Israeli cats being 43% heavier than Indian cats (Mukherjee and Groves 2007). Territories are maintained by urine spraying and scent marking (Sunquist and Sunquist 2002). Tehsin (1996) reported that in Rajasthan, a sick jungle cat approached the bushes, ripped the leaves of *Eclipta alba* (Compositae), swallowed them and then walked some distance away, stretched and puked the food. Besides it is noted in Indian medical records this plant was used in liver disorders, regulating gall bladder glands, hepatitis and digestion. Tehsin and Tehsin (1988) also reported that in Rajasthan, it was observed hiding inside bushes, waiting for a jungle fowl's arrival, caught one of them and disappeared inside the dense bushes.

Somewhat larger than domestic cats, jungle cats range from 55 to 94 cm (22 to 37 in) in length, plus a relatively short 20 to 31 cm (7.9 to 12 in) tail, and stand about 36 cm (14 in) tall. Weight varies across the range from 3 to 12 kg. In Uzbekistan, they are known to travel 5 to 6 km (Allayarov 1964). In captivity pregnancy period is about 63-68 days (Green 1991), the birth interval is 93-131 days (Schauenberg 1979). Vertical bars are visible on the fur of kittens, which disappear in adult cats, although a few dark markings may be retained on the limbs or tail.

Areas with extensive deciduous Dipterocarp forest and at least scattered surface water are the species predominant known habitat in Indochina. However, areas such as the Nakai Plateau which support other forms of savanna-like vegetation may support the species. It is probably rare from all closed canopy forests like rainforest. The species may make use of agricultural areas with a low intensity of human use and which retain patches of scrub (Duckworth et al., 2005). In sandy and stony land, sparsely scattered bushes (Roberts 1977) alongside river or oasis (Heptner and Sludskii 1972).

Jungle cats have adapted well to irrigated cultivation, having been observed in many different types of agricultural and forest plantations throughout their range, such as sugar cane plantations in India (Tikader 1983, Khan and Beg 1986). In Israel they can be found around pisciculture ponds and irrigation ditches. Vereschagin (1959) noted that the jungle cat's use of the semi-arid plains of Azerbaijan increased with the development of a local irrigation system and decreased with its abandonment. However, moving the seasonally flooded riverine tugai vegetation (trees and shrubs with dense stands of tall reeds and grasses) of this region for livestock fodder, as well as plowing it under for agriculture, is known to be associated with the decline of jungle cat populations in the European-central Asian parts of its range (Nowell and Jackson 1996).

Jungle cats feed mainly on prey that weighs less than a kilogram. Small mammals, principally rodents, are the prey most frequently found in fecal contents (Gupta 2011). An earlier study in India's Sariska Tiger Reserve estimated that jungle cats eat three to five rodents per day (Mukherjee et al., 2004). Birds rank second in importance, but in southern Russia waterfowl are the main prey of jungle cat diet in the winter. With overwintering populations of waterfowl congregating in large numbers on unfrozen rivers and marshes, the jungle cat hunts among reed beds and along edges of wetlands, searching for injured or weakened birds. Other prey species are taken more opportunistically, including hares, nutria, lizards, snakes, frogs, insects, and fish (Heptner et al., 1992). In India, they have been seen scavenging over kills of large predators such as the Asiatic lion (Duckworth et al., 2008a). In a study in southern Uzbekistan, fruits of the Russian olive made up 17% of their diet in winter (Sunquist and Sunquist 2002). While jungle cats specialize on small prey, they are large and powerful enough to kill even young swine, sub-adult gazelles, and chital fawn (Sunquist and Sunquist 2002, Gupta 2011).

In the reed beds of the Nile delta they hunt water voles, frogs, fish and waterfowl, while in drier habitats they feed on everything from hares, gerbils and house mice to birds, snakes, lizards, and domestic poultry. Known to be active by day and by night, they are often spotted amidst human settlements, denning in old buildings. Their predators even include larger felines and canids such as tiger, leopard and dhole (Ramesh 2010). Jungle cats most often, hunt rodents, birds and frogs. They have been observed to be capable of swimming as much as 1.5 km at a stretch (Hinde and Hunter 2005). Near human settlements, they may feed on domesticated chicken and ducks. They also exhibit arboreal habits (Heptner et al., 1992). Like most other cats, they hunt by stalking and ambushing their prey, and they use reeds or tall grass as cover. They are adept at leaping, and sometimes attempt to catch birds in flight. Although they can run at up to 32 km/hr (20 mph), they rarely pursue prey that escapes their initial pounce (Sunquist and Sunquist 2002). Water and dense ground cover can be found in a variety of habitats, ranging from desert (where the cat is found near oases or along riverbeds) to grassland, shrubby woodland and dry deciduous forest, as well as cleared areas in moist forest (Nowell and Jackson 1996). Density estimates from natural tugai habitat in central Asia range from 4-15 individuals per 10 km^2, but where this vegetation type has declined due to development and their density may not exceed two cats per 10 km^2 (Nowell and Jackson 1996).

Females are sexually mature at the age of 11 months and give birth to litters of one to six kittens, although more than three are relatively unusual. Jungle cats sometimes raise two litters in a year (Heptner et al., 1992). Gestation lasts 63–66 days and is remarkably short for an animal of this size. Birth generally takes place between December and June, depending on the local climate, although females can sometimes give birth to two litters in a year. Before birth, the mother prepares a den in an abandoned animal burrow, hollow tree, or reed bed (Sunquist and Sunquist 2002).

Kittens weigh 43 to 160 gms (1.5 to 5.6 oz) at birth, tending to be much smaller in the wild than in captivity. Initially blind and helpless, they open their eyes at ten to thirteen days of age, and are fully weaned by around three months. Males usually do not participate in the raising of kittens, but in captivity have been observed to be very protective of their offspring, more than the females, or males of other cat species. Kittens begin to catch their own prey at around six months, and leave the mother after eight or nine months (Schauenberg 1979, Sunquist and Sunquist 2002).

Principal threats

Unselective trapping, snaring and poisoning around agricultural and settled areas have caused population declines in many areas throughout its range (Abu-Baker et al., 2003, Duckworth et al., 2005). India formerly exported large numbers of jungle cat skins before the species came under legal protection (over 300,000 were declared as being held by traders from Kashmir there when export was banned in 1979) (Sunquist and Sunquist 2002), as well as in Egypt and Afghanistan (Habibi 2003). Jungle cat skin is used for making hats, locally called ''over chamo'' in Arunachal Pradesh (Solanki and Chutia 2004). This species is often assessed as being in no danger and has therefore been ignored as a rare species. Jungle cats can do well in cultivated landscapes (especially those that lead to increased numbers of rodents) and artificial wetlands. However, reclamation and destruction of natural wetlands, ongoing throughout its range but particularly in the arid areas, still pose a threat to the species, as density in natural wetlands is generally higher (Nowell and Jackson 1996). In fact, in recent years a clearer picture has showed that this species could well be among the rarest of the small cats in Asia, and definitely the rarest one for which there is no protection within most of its current distribution since it is assumed to be common.

Conservation measures

The jungle cat is listed on CITES Appendix II. It is protected from hunting in some range states (India), but in many it receives no legal protection outside protected areas (Nowell and Jackson 1996). The species now receives legal protection from all hunting and trading within Afghanistan after being placed on the country's 2009 Protected Species List. Furthermore, given the amount of habitat loss occurring in riparian and wetland areas in Afghanistan, this species should be considered a research priority.

The ecology and status of the jungle cat is still poorly known (Nowell and Jackson 1996, Sunquist and Sunquist 2002). In Southwest and Southeast Asia, where it is considered rare and declining, more research needs to be undertaken to gain knowledge of current distribution, both in and outside Protected Areas (Abu-Baker et al., 2003, Duckworth et al., 2005). The jungle cat would also benefit from improved protection of natural wetlands and reed-beds, particularly in the more arid parts of its range, and improved legislation prohibiting fur trade. Some farmers consider the jungle cat a pest since it takes away poultry (Abu-Baker et al., 2003), hence conservation measures should include protection for domestic fowl and halting of indiscriminate poisoning and trapping of jungle cat. It is imperative to observe feeding habits of the jungle cat to find out the reasons for their dependency on domesticated animals, when in reality there could be plenty of prey available.

1.5.2 Rusty-spotted cat

Distribution status

The Rusty-spotted Cat (*Prionailurus rubiginosus*) is Felidae's smallest member and found only in India and Sri Lanka (Pocock 1939). It has been listed as Vulnerable by IUCN in 2002 as the total effective population size is below 10,000 mature individuals with a declining trend due to habitat loss, and no subpopulation containing more than 1,000 mature breeding individuals (Khan and Mukherjee 2008). The Indian population is listed on CITES Appendix I. The Sri Lankan population is included in CITES Appendix II. The species is fully protected over most of its range, with hunting and trade banned in India and Sri Lanka (Khan and Mukherjee 2008). The rusty-spotted cat rivals the black-footed cat as the world's smallest wild cat. It is smaller than a domestic cat. It is 35 to 48 cm (14 to 19 in) in length, with a 15 to 30 cm (5.9 to 12 in) tail, and weighs only 0.9 to 1.6 kg (2.0 to 3.5 lb). Males weigh about 1.5 to1.6 kg, and females weigh 1.1 kg (Phillips 1935). The short fur is grey over most of the body, with rusty spots over the back and flanks, while the underbelly is white with large dark spots. The darker colored tail is thick and about half the length of the body, and the spots are less distinct. The tail averages about 50% of head-body length (Pocock 1939). There are six dark streaks on each side of the head, extending over the cheeks and forehead (Sunquist and Sunquist 2002). Rusty-spotted cats have been observed frequently by researchers (Kittle and Watson 2004, Manakadan and Sivakumar 2006, Patel 2006, Vyas et al., 2007).

Rusty-spotted cats have a relatively restricted distribution. They mainly occur in moist and dry deciduous forests as well as scrub and grassland, but are likely to be absent from evergreen forest (Nowell and Jackson 1996). They prefer dense vegetation and rocky areas (Kittle and Watson 2004, Patel 2006). It is difficult to say whether distribution is continuous throughout India because the species' habitat preferences are poorly understood. In Sri Lanka, Phillips (1935) stated that "it is rarely seen far away from jungles", while De Alwis (1973) terms it "the ubiquitous wildcat of Ceylon'' equally comfortable in the high montane forests of Horton Plains

(2135 m) or the sizzling sandy wastes of the Hambantota coastline". In India, Prater (1971) described its habitat as grassland, scrub and forest. Similarly, residents of 45 villages in the Dangs semi-evergreen forest described its habitat as rocky areas and hill slopes, but not forest edges (Worah 1991).

Perhaps these seeming inconsistencies can be explained in terms of interspecific competition or ecological separation, although this subject has scarcely been investigated for the small tropical Asian cats. The closely related leopard cat is found throughout much of India, but is absent from Sri Lanka. It is possible that the rusty-spotted cat is the more common of the two species in the drier, more open vegetation types of India (Athreya 2010), while the leopard cat predominate moist forests. This would explain the concentration of rusty-spotted cat records in southern India, and the infrequent and seemingly isolated reports from more northern regions. Rusty-spotted cats can tolerate modified habitats; females with kittens have been found denning in tea plantations of Sri Lanka (Phillips 1935), and in the attics of houses in southern India, areas surrounded by paddy fields and coconut plantations. In the latter case, it was noted that the species was virtually unknown to local residents. A rusty-spotted cat was photographed in 1993 in an old farm house in a mango plantation in Bansda National Park in Gujarat. It was also observed on a tree in Gir Wildlife Sanctuary (Mitra 2005). According to Karanth (1993), these cats can be found in farmland throughout southern India's Deccan Plateau, and in the outskirts of Bangalore city.

In India, they were long thought to be confined to the south, but records have established that they are found over much of the country (Kittle and Watson 2004). They were observed in Gir Wildlife Sanctuary and National Park, Tadoba-Andhari Tiger Reserve in Maharashtra, on India's east coast, and in eastern Gujarat (Pathak 1990, Dubey 1999, Manakadan and Sivakumar 2006, Patel 2006). Camera trapping revealed their presence in the Pilibhit Tiger Reserve in the Indian Terai and the Nagzira Wildlife Sanctuary in Maharashtra (Anwar et al., 2010, Patel 2010). In western Maharashtra, there is a breeding population of rusty-spotted cats in a human dominated agricultural landscape, where rodent densities are high (Athreya 2010).

In Sri Lanka, there are a few records from montane and lowland rainforest. There are two distinct populations, one in the dry zone and the other in the wet zone (Deraniyagala 1956). As indicated by the patchy and infrequent nature of collections and observations this remains speculative until basic natural history studies have been carried out.

Ecology and behaviour

Very little is known of the rusty-spotted cat's behavior in the wild. They are apparently nocturnal (Chakraborty 1978, Pathak 1990) and partly arboreal, spending the day resting in dense cover or shelter such as hollow logs. It feeds mainly on rodents, birds, lizards, frogs, or insects. The diet of the rusty-spotted cat has not been properly documented; Phillips (1935) reported without elaboration that it feeds upon small mammals and birds. Local people in both Sri Lanka and India have reported that they are most visible after heavy rain, when they emerge to feed on rodents and frogs (de Alwis 1973, Worah 1991). They are known to prey on domestic poultry (Phillips 1935, Pocock 1939).

While dense vegetation and rocky areas are preferred (Worah 1991, Kittle and Watson 2004, Patel 2006), rusty-spotted cats have been found in the midst of agricultural and settled areas (Nowell and Jackson 1996, Mukherjee 1998, Nekaris 2003). They are highly arboreal (Sunquist and Sunquist 2002), and Patel (2006) observed cats pouncing down from tree branches when hunting prey. Most observations were in the night (Mukherjee 1998, Nekaris 2003, Kittle and Watson 2004, Patel 2006, Vyas et al., 2007). Once it was seen hunting frogs, but small rodents as the main prey were reported from a series of observations by Patel (2006) and Nekaris (2003); seeking out such prey is likely why the cats venture into cultivated areas. The animal was sighted in Nugu, at 1950 hrs, on a fig tree (*Ficus bengalensis*) at a height of about 5 m, the tree was 16 m tall and another sighted in Bandipur National Park and one in Sira of Tumkur (Kumara and Singh 2007). The other sighting at three and a half kilometers from Sira town in Karnataka was near a roadside Tamarind tree (*Tamarindus indica*) at 23:30 hrs (Kumara and Singh 2007). It was at a height of about 2 m, the tree was about 6 m tall. The cat remained on the tree for about 5 min until it moved to another branch, due to disturbance. The sighting locality was close to human habitations, which was adjacent to a reserved forest. The general forest type of the region is dry scrub or dry deciduous.

Oestrus usually lasts for five days. The mother prepares a den in a secluded location, and gives birth to one or two kittens after a 65-70 day gestation. At birth, the kittens weigh just 60 to 77 g (2.1 to 2.7 oz), and are marked with rows of black spots. The cat reaches sexual maturity at around 68 weeks, by which time it has developed the distinctive adult coat pattern of rusty blotches. Adults weigh about 0.9 kg (Sunquist and Sunquist 2002). Rusty-spotted cats have lived for twelve years in captivity, but their lifespan in the wild is unknown (Pocock 1939).

Principal Threats

Habitat loss and the spread of cultivation are serious problems for wildlife in both India and Sri Lanka. Although there are several records of rusty-spotted cats from cultivated and settled areas, it is not known to what degree cat populations are able to persist in such areas. There have been occasional reports of rusty-spotted cat skins in trade (Nowell and Jackson 1996). In some areas, they are hunted for food or as livestock pests (Sunquist and Sunquist 2002).

Deforestation and the spread of cultivation are serious problems for wildlife in both India and Sri Lanka. As far as rusty-spotted cats are concerned, it is not known if populations can persist in cultivated landscapes, and individuals taking poultry are vulnerable to persecution. A long coat made of rusty-spotted cat fur was found for sale in Kathmandu, Nepal (Van Gruisen and Sinclair 1992). Early reports on rusty-spotted cats refer to hybridization with domestic cats as common occurrences, but they have not been substantiated.

1.5.3 Leopard cat

Distribution and status

The leopard cat is a widespread and relatively common species (Nowell and Jackson 1996, Sunquist and Sunquist 2002), although some island subspecies are included in the Red List. Although there is a declining population trend in parts of its range due to habitat loss and hunting, the species is stable in many areas, even thriving in some altered habitats including oil palm and sugar cane plantations (IUCN 2007).

The leopard cat is a widespread in Asia (Nowell and Jackson 1996, Duckworth et al., 1999, Holden 2001, Duckworth et al., 2005, Lynam et al., 2006, Yasuda et al., 2007). It is found throughout most of India, west into Pakistan and Afghanistan (Habibi 2003), through the Himalayan foothills, across most of China, and north to the Korean peninsula and into the Russian Far East (Nowell and Jackson 1996). It is found throughout Southeast Asia, and on the islands of Sumatra, Java, Borneo and Taiwan. It is found on numerous small offshore islands of mainland Asia (Nowell and Jackson 1996, Sunquist and Sunquist 2002). The leopard cat is the only wild felid found in Japan, where it occurs on the small islands of Tsushima and Iriomote, and the Philippines, where it occurs on the islands of Palawan, Panay, Negros and Cebu. In the Philippines, there are recent (2007) unconfirmed reports from the island of Masbate.

Higher survival rates (92%) of this species were recorded in a protected area with little human influence, compared with lower rates in areas with greater human activity (53-82%) (Haines et al., 2004). The small population (approximately 100) of leopard cat on Japan's 710 km^2 Tsushima Island, considered the subspecies as occurs in northeastern mainland Asia, has decreased over the last 30-40 years (Izawa et al., 2007). It has been reported to occur in some reserves of Karnataka (Karanth 1986). It was sighted in Sharavathi Valley Wildlife Sanctuary, Bandipur Tiger Reserve, Talakavari Wildlife Sanctuary, Pushpagiri Wildlife Sanctuary and in a coffee estate in Virajpet adjacent to Brahmagiri Wildlife Sanctuary in Kodagu district (Kumara and Singh 2007). The species was quite common in Kodagu and fringes of coffee estate adjacent to Bhadra Wildlife Sanctuary in Chikmagalur.

Ecology and behaviour

The species can range upto 3000 m in parts of its range which extends into the Himalayas along river valleys. It occurs in a broad spectrum of habitats, from tropical rainforest to temperate broadleaf and marginally, coniferous forest, as well as shrub forest and successional grasslands. The northern boundaries of its range are limited by snow cover; the leopard cat avoids areas where snow is more than 10 cm deep. It is not found in the cold steppe grasslands, and generally does not occur in arid zones, although there are a few records from relatively dry and barren areas of Pakistan. Leopard cats occur commonly in dense secondary growth, including logged areas, and have been found in agricultural and forest (rubber tree, oil palm, sugarcane) plantations. The species can live close to rural settlements. Leopard cats are excellent swimmers, and have successfully colonized offshore islands throughout their range (Nowell and Jackson 1996, Sunquist and Sunquist 2002).

In the most comprehensive study, based on a large sample size of 20 radio-collared cats in Thailand's Phu Khieu Wildlife Sanctuary, mean home range size (95% MCP) was 12.7 km^2, larger than in other areas of Thailand (4.5 km^2) (Grassman et al., 2005), Borneo (3.5 km^2: Rajaratnam 2000), or on Japan's Iriomote island (Schmidt et al., 2003). There was no significant difference between male and female home range size. Males weigh (mean ± SD) 2.9 ± 0.38 kg and females weigh 2.3 ± 0.27 kg in Thailand (Grassman et al., 2005). Open and closed forest habitats were used in proportion to their occurrence, and activity patterns showed crepuscular and nocturnal peaks. In Borneo, Rajaratnam et al., (2007) found that leopard cats hunted rodents in oil palm plantations, and used forest fragments for resting and breeding. Murids dominated the diet (85-90%: Grassman et al., 2005, Rajaratnam et al., 2007). Other small mammals, eels and fish have also been reported in their diet as well as occasional scavenging on carrion (Nowell and Jackson 1996).

Principal threats

Until banned in 1988 leopard cat skins were widely exported from China to Western Europe, after which they were exported to Japan. Between 1984 and 1989 China exported c. 2,00,000 skins a year and stockpiles were estimated at over 8,00,000. Collected by specialist hunting tribes, the skins were usually sent to Kashmir to be fashioned into garments for export and for Indian markets. Following violent political unrest in Kashmir, traders established a market in Kathmandu which benefits from tourism. Although commercial trade is much reduced, the species continues to be hunted throughout most of its range for fur, food, and as pets. They are also widely viewed as poultry pests and killed in retribution. Island populations are small and seriously threatened in the Philippines and Japan. Leopard cats can hybridize with domestic cats, as is shown by the popular domestic breed, the "safari cat". Hybridization in the wild has been reported, but is not considered a significant threat. Although the species is less dependent on forest cover than others, habitat loss and fragmentation is still a major threat across most of its Asian range (Nowell and Jackson 1996).

Conservation measures

Included on CITES Appendix II; populations in Bangladesh, India and Thailand are included on Appendix I. The species is protected at the national level over part of its range, with hunting prohibited in Bangladesh, Cambodia, Hong Kong, India, Indonesia, Japan, Malaysia, Myanmar, Nepal, Pakistan, Philippines, Russia, Thailand and Taiwan, and hunting and trade regulations in place in South Korea, Lao PDR and Singapore (Nowell and Jackson 1996). The species is on Afghanistan’s 2009 Protected Species List, banning all hunting and trading of this species within the country.

1.5.4 Small Indian civet

Distribution and status

The Small Indian Civet (*Viverricula indica*) is found across south and Southeast Asia as well as in the Indonesian archipelago. This species is listed as Least Concern due to its widespread geographical distribution and habitat use, with evidence from many range states of healthy populations in agricultural/secondary landscapes. Some populations may be locally depressed by snaring/hunting but not to the extent of threatening the species. It persists in the face of heavy hunting that has wiped out some other ground-dwelling species, and it is mainly an open-forest and edge species, so that it has probably benefited from the rampant forest conversion of the last century in Southeast Asia. There are no recent records for Sumatra, but it is possible that the species was not native there.

This species is currently known to occur in south and central China (Wang and Fuller 2001, 2004), Hong Kong (Suen 2002), most of India (Mudappa 2002), Lao PDR (Duckworth 1997), Myanmar (Su Su 2005), Thailand (Rabinowitz 1991, Austin and Tewes, 1999), Vietnam (Roberton 2007), Cambodia and Sri Lanka (Duckworth et al., 2008b). It was formerly known from Singapore but its current status there is unclear. Corbet and Hill (1992) include all of Sumatra for the species' distribution range, but only four individuals are known, all from one locality in the far north. Other records from Sumatra have not been seen, and it is suggested to restrict its range to the north, and the lack of recent records from Sumatra is still puzzling

(Duckworth et al., 2008b). It has been introduced to Madagascar (Lekagul and McNeely 1977), Socotra (Pocock 1939) and Zanzibar. The current status of these introduced populations is also poorly known.

The population status of this species is less well known than other Southeast Asian small carnivores, because recent survey efforts have not been appropriate for assessing the species. This is best shown by records from Myanmar. Su Su (2005) found that it was the second most common species of small carnivore recorded in Hlawga Wildlife Park in Myanmar, a secondary small isolate of semi-natural habitat in the outskirts of Yangon, a former capital city, subject to barely controlled snaring and other forms of encroachment, where only one other species of small carnivore persists in significant numbers. The habitat degradation and hunting patterns in Myanmar where the species is common are representative of much of non-Sundaic Southeast Asia, and incidental records from various other sites (Duckworth and Robichaud 2005) imply large populations in total, although it is likely that in Vietnam and Lao at least populations will have been somewhat depleted by snaring. In Cambodia, where more camera-trapping has taken place in deciduous forest areas, it is commonly camera trapped. The same factor was suspected to be responsible for the relatively few recent records from Lao PDR (Duckworth et al., 1999), and may also have generated earlier remarks such as in Thailand, this species is rather rare (Lekagul and McNeely 1977); in fact, it is common in the degraded parts of Khao Yai national park. It is also abundant over large areas of India. It is common in deciduous forests of Dak Lak, Vietnam (Le Xuan Canh et al., 1997) and more widely in the country (Roberton 2007). No information has been sought for the Sundaic populations except for Sumatra, where the species status has always been unclear.

Ecology and behaviour

This species has been recorded in semi-evergreen and deciduous, mixed deciduous, bamboo, scrubby areas, grasslands and riverine habitat (Duckworth 1997, Le Xuan Canh et al., 1997, Mudappa 2002, Su Su 2005). The species measures 100 cm in body length (including a tail length of 38 cm) and 2.5 kg in body weight (Kumar and Umapathy 1999). This species is nocturnal and mostly terrestrial (Mudappa 2002). An adult male was radio-tracked in Thailand and had a home range of 3.1 km^2 (Rabinowitz 1991). The niche of *V. indica* in Myanmar is that of a solitary, nocturnal predator of small vertebrates and arthropods, that forages widely at ground level in different habitat types and rests during daytime in dense shrubbery near the ground (Su and Sale 2007). In Lao PDR, this species is found in semi-evergreen and deciduous forest including adjacent degraded areas (Duckworth et al., 1999). In other countries it is tolerant of habitat degradation and lives in proximity to human communities (Lekagul and McNeely 1988) and an active avoidance of closed evergreen forest was shown in Myanmar (Than Zaw et al., 2008). In Thailand, it is mostly found in long grass or scrub habitat, particularly in areas near villages, where it may live in drains, outhouses, and roofs, eating domestic poultry (Su Su 2005) as well as rats, mice, birds, snakes, fruit, and roots, as well as carrion (Lekagul and McNeely 1988). It produces a litter of three to five, and the life span is eight to nine years (Lekagul and McNeely 1988). In Myanmar this species was recorded from both mixed deciduous forest and bamboo forest (Su Su 2005). This species was rarely seen in the undisturbed rainforests of Kalakad-Mundanthurai Tiger Reserve (KMTR) in India, and was mostly seen near garbage dumps (Mudappa 2002). In KMTR they were not camera-trapped frequently in rainforests, but were the most camera-trapped species in grasslands and riverine habitat (Mudappa 2002, Ramesh et al., 2012). This species is nocturnal, and mostly terrestrial and insectivorous. Wang and Fuller (2001) conducted a study on the ecology of this species near Taohong Village in northern Jiangxi Province, southeastern China, from April 1993 to November 1994. Wang and Fuller (2003) conducted a study on the food habits of this species in a rural agricultural area of southeastern China (Taohong Village, Jiangxi Province) by analyzing its scats, the study was conducted between June 1992 and November 1994, and reported that this species ate mostly mammals, with moderate insects and plants.

Principal threats

The extent to which extensive habitat loss and degradation are a threat to this species remains unclear as in most or all of its range areas it is more common in altered landscapes than in closed-canopy old-growth evergreen forest. This species is hunted for its meat and scent (Gupta 2004) in some portions of its range which potentially might reduce populations. Ground-dwelling small carnivores are exposed to hunting, particularly with snares. This is occurring in much of its range, including Lao PDR (Duckworth 1997) and Thailand, with snaring found even in some protected areas. There has been an increased demand for civet meat in Chinese and Vietnamese markets (Bell et al., 2004, Lynam et al., 2006). In India animals are in captivity for collection of 'civet', a fixative used internationally in the perfume industry and domestically for various purposes; even in areas of heavy collection, the animals remain common in the degraded forest, scrub and agricultural landscapes covering most of peninsular India. The converse seems to be true: it remains more common than most other species of small carnivores in heavily encroached areas of southern China, than Myanmar (Su Su 2005).

Conservation measures

In Myanmar, this species is totally protected under the Wildlife Act of 1994 (Su Su 2005). This species is listed in Category II of the China Wildlife Protection Law (1988) (Li et al., 2000) and as 'Vulnerable' in the China Red List (Wang and Xie 2004). It is listed on CITES Appendix III (India). It has been recorded in many Protected Areas (Duckworth 1997, Mudappa 2002, Su Su 2005).

1.5.5 Common palm civet

Distribution and status

It is normally found in Ceylon, Bangladesh, Brunei Darussalam, Singapore, Myanmar, India and Pakistan, Burma and Southern China, south to Sumatra and Java, east to the Philippines, Borneo, Celebes and the Lesser Sunda Island. In Malaysia, the common palm civets are wide spread on the mainland, in Langkawi, Pulau Penang, and Pulau Tioman. Kumara and Singh (2007) sighted 32 animals; the sightings included two animals each in Brahmagiri-Makut and Sirsi-Honnavara, nine animals in Nagarahole, five animals in Chamundi hill, three animals each in Sharavathi Valley Wildlife Sanctuary and Bandipur National Park, and four animals each in Talakaveri Wildlife Sanctuary and Pushpagiri Wildlife Sanctuary. Although 15 animals were sighted in the Western Ghats region, the sightings were mostly from moist deciduous or deciduous forests.

Ecology and behavior

Paradoxurus hermaphroditus live in tropical forests, plantations, fruit orchards and often finds its way into human settlements. This species often colonizes the eaves of houses or outbuildings. During the day, palm civets most often rest in trees covered with heavy vines; they also rest in tree cavities and occasionally in tall trees (Joshi et al., 1995). Palm civets forage exclusively at night. The likelihood of encountering predators during the day may favour nocturnal foraging behavior. They are also expert climbers and spend most of their lives in trees. Palm civets choose the tallest and largest tree in the immediate area. It exhibits considerable overlap of ranges among conspecifics. Social organization and activity patterns of these civets are shaped by the distribution of food resources and the activities of larger mammalian predators. This species is more arboreal than the larger Indian civet. Non-overlapping territories often occur when food resources are evenly distributed and defense is uneconomical when food resources are superabundant. When foraging in the same area, civets repeatedly use the same resting trees. Resting trees with vines and holes, preferred by the civets, are used for several consecutive days. When they are threatened, they usually hiss and spit like cats. They have often been observed to breed in house roofs in coastal plains of Udupi, and also in dry plains, such as Bidar district in Karnataka with little forest, however, they are very rare or absent in areas completely bare and without any vegetation (Kumara and Singh 2007). Pillay (2009) sighted two pairs of common palm civet in Anamalai Tiger Reserve on the road from Anamalai town to Valparai, three pairs at different locations in Chinnar Wildlife Sanctuary, while driving along the road from Marayoor town to Chinnar; all these sightings occurred in dry deciduous habitats.

They are identified by the three dark stripes along the back, on a creamy or dark grey background, and by the dark mask across the eyes and muzzle. It weighs from 4 to 11 pounds (1.8 to 5 kg). Their head and body length is approx. 17 to 28 inches (43.2 to 71cm). Pinnae are small and faintly pointed muzzle. It has a long body with short legs. They have coarse grayish to brown coat with black-tipped guard hairs. Three rows of black spots run along each side of its body. This species is known to occur in secondary forests, fruit orchards, near villages or in the tops of trees close to human habitation, indeed it sometimes inhabits the roof spaces of rural properties. They usually nest, however, in hollow trees where they raise two or three young.

Palm civets are primarily frugivorous, feeding on berries and pulpy fruits, including those of *Ficus* trees and palms (Shanahan et al., 2001). The niche of *P. hermaphroditus* in a regenerated degraded forest in Myanmar is summarized as that of a small, solitary, nocturnal frugivore which mostly feeds and rests in the canopy of trees typical of mixed deciduous forest (Su and Sale 2007). They generally eat small vertebrates, insects and ripe fruits. Besides that, they are very fond of palm sap, therefore their common name is also called the Toddy cat for its apparent fondness for sucking the sap from palm trees; the same sap is used to make the alcoholic drink called 'toddy'.

The sap is used by natives to make sweet liquor called "toddy", which gives the palm civet its common name – Toddy Cat. The palm civet is also fond of coffee cherries. They eat the outer fruit and the coffee beans pass through their digestive tract. Palm civet's natural diet includes flesh and large number of wild fruits. Toddy cat reproduce throughout the year although it has been recorded that kittens are mostly found from October to December. During the brief periods of mating and when the females have their young, the civets occupy resting trees together. Normally, kittens are born in a litter of 2 to 5 young. They are usually born in a hollow tree, boulder crevices or a space among the rocks. Their eyes are closed at birth. Sexual maturity is attainted at 11 – 12 months. In captivity, the common palm civet can live up to 22 years.

Principal threats

The common palm civets are often killed by fruit agriculturalists. In south India they are often live trapped since they are considered as pests in coconut plantations, coffee estates and orchards. The civets produce secretion that is sweet, musky and pleasant to human nose. Therefore, civets are hunted for their musk which is gathered by scraping the civets' anal sac. This species is also prized for their meat in South-east Asia.

Conservation measures

The habitat of palm civets can be saved by preventing over-logging activities in rainforests. Local hunting and habitat loss are the major factors affecting the status and distribution of small carnivore species. The government must continue to monitor logging activities. Understanding utilization of wild natural food will allow better maintenance of species in captivity, with breeding populations to reduce the potential pressure on natural populations (Balakrishnan 2000). The economic importance of civets is yet to be explored in India. In addition, road network with busy traffic in forest areas has led to many road kills of civets (Sheshadri and Ganesh 2011). Habitat disturbance and illegal hunting by the local tribes (Kani and Kurumba) in south India should be investigated in detail. This is an adaptable species that seems to thrive in a variety of environmental conditions from a gradient of disturbed habitats to pristine intact forests.

1.5.6 Brown palm civet

Distribution status

This species is listed as Least Concern because in view of its abundance within highly disturbed and fragmented areas, with many animals living in plantation-dominated landscapes, and its large population known from surveys in several areas. However this species has a restricted distribution, and there is continuing habitat loss and conversion of coffee/cardamom plantations (which hold substantial numbers of the species) into tea (which does not support it), but not at rates nearly sufficient to drive habitat-based population declines even for Near Threatened. Brown palm civets are considered to be restricted to the evergreen forests and the adjacent forests of the Western Ghats, ranging from Brahmagiri in the south to Khanapur (Belgaum district) in the north. These factors mean that the species ought to be monitored may warrant concern in the future; the role of the remaining tall forest fragments as sources for the animals living in artificial habitats is still not clear. Although the species does not currently qualify for Near Threatened there is some concern that it may be declining in some portions of its range. This species is endemic to southern India (Wilson and Reeder 2005), where it is found in the Western Ghats (Pocock 1939, Corbet and Hill 1992, Mudappa 1998). The distribution of this species has been poorly documented due to its nocturnal and arboreal habits. Its population status is poorly known. It is not as rare as it was previously generally believed to be, at least in relatively undisturbed rainforest (Mudappa 2002). It was the most frequently sighted small carnivore in

Kalakad-Mudanthurai Tiger Reserve (KMTR) between May 1996 and December 1999 (Mudappa 2002). Ryley (1913) found them to be fairly abundant in Coorg, though not nearly as common as *Paradoxurus hermaphroditus* (Rajamani et al., 2002). It appears to be fairly common in Kakachi-Upper Kodayar (Ganesh 1997) and other areas above 1,000 m within KMTR in the Agasthyamalai hills and Anamalai hills (Mudappa 2001). It was most common in altitudes above 1000 m, though they were seen as low as even 700 m (Mudappa 2002).

Ecology and behaviour

It is largely arboreal, nocturnal, and mainly frugivorous, feeding on nearly 40 rainforest trees and liana fruit species, though it does supplement its diet with birds, rodents, and insects (Pocock 1939). It is often found in coffee plantations and elevated (above 500 m) moist forests (Ashraf et al*.,* 1993). Occasionally it will feed on the ground, as indicated by success in live-trapping and camera trapping in KMTR (Mudappa 2002). It is known mostly from tropical rainforests but has also been recorded from coffee estates in Coorg and Anamalais (Ryley 1913, Mudappa 2001). The animal weighed (mean ± SD) 2.4 ± 0.8 kg in KMTR (Mudappa 2001). In a survey conducted in the Western Ghats in 2001-02, 23 sightings of this species were in evergreen forests, including five in high altitude montane evergreen forest or sholas (Rajamani et al*.,* 2002). It was recorded in both undisturbed, large patches of contiguous forest, as well as in fragments surrounded by tea plantations and human habitations. They were recorded on forest trails and along main roads, often exposed to traffic. Rajamani et al., (2002) found that this species may be more dependent on the structure and floristics of forests, rather than altitude.

Principal threats

Populations may be threatened by habitat destruction due to mining activities in Kudremukh (which has since been closed by the government), hydroelectric projects in Anamalais, and large-scale plantations of coffee, cardamom, and tea in and around Protected Areas (Ashraf et al., 1993). Hunting is unlikely to be a major threat to this species; however, illegal hunting is still common in privately owned plantations. As it is strictly frugivorous and arboreal, rainforest fragmentation is a likely threat to this species (Mudappa 2002). It is able to survive in heavily encroached areas provided some fragments remain with relatively unbroken canopy and adequate food resources, such as coffee and cardamom, but not tea, Eucalyptus, or teak (Rajamani et al*.,* 2002).

Conservation measures

This species is listed on CITES Appendix III (Wilson and Reeder 2005), as well as Schedule II part II of the Indian Wildlife (Protection) Act, 1972. This species is likely to be found in 25 protected areas within its distribution (Ashraf et al., 1993). More surveys are needed to determine the abundance and distribution of this species, due to concern about threats from commercial plantations of coffee, tea, *Eucalyptus spp*, and teak, as well as other development activities (Menon and Bawa 1997). Long-term protection of primary rainforests, both large tracts as well as fragments, is imperative to the conservation of this species. More surveys are urgently needed to determine the true abundance and distribution of this species. More information about possible threats would allow more certainty over its status.

To further our understanding of this species and its current status there is a need to increase the intensity of camera trapping and night surveys in identified areas. Equally important areas will require constant interactions with local communities living to gather information regarding possible sightings. Other methods of increasing information relating to the brown palm civet include creating awareness among the local masses.

1.5.7 Stripe-necked mongoose

Distribution and status

This species is listed as Least Concern due to its wide distribution, presumed large population, occurrence in a number of Protected Areas, tolerance to some degree of habitat modification, and because it is unlikely to be declining at nearly the rate required to qualify for listing in a threatened category. The stripe-necked mongoose is found in Southwest India (Mudappa 1998) and Sri Lanka (Santiapillai et al., 2000). In India, this species is found particularly in the Western Ghats and other hill tracts in the Nilgiris from Coorg (now Kodagu) to Travancore (Pocock 1939, Prater 1971, Medway 1978, Phillips 1984, Corbet and Hill 1992, Mudappa 1998), and Dharwar, as well as near Bombay (Blanford 1888-1891) to Cape Comorin (Van Rompaey and Jayakumar 2003). In Sri Lanka it is found across a range of elevations from high hills to lowlands, being most common between 400 and 1400 m (Van Rompaey and Jayakumar 2003). Population assessments for the stripe-necked mongoose have been made but these are not recent (Van Rompaey and Jayakumar 2003); nevertheless, the species appears to be relatively common in many areas of its range. In India, it is rare in the northern part of its range, and most abundant in Travancore (Jerdon 1874). It is also common in the Nilgiri and Palni plateaus in the High Wavy Mountains (Hutton 1949), and on the Valparai Plateau in the Anamalai Hills. It is not uncommon in Coorg, although less common than *Herpestes edwardsii* (Van Rompaey and Jayakumar 2003). In Sri Lanka, it used to be fairly common in the higher hills of the Central Provinces, but seems to be declining (Van Rompaey and Jayakumar 2003). It is relatively common in the Sri Lankan interior and is "moderately plentiful" in the Horton Plains area and around Gamaduwa (Van Rompaey and Jayakumar 2003). The species is not uncommon in the low-country dry zone along the banks of the Menik Ganga (Van Rompaey and Jayakumar 2003), and is present, but not common in the wet zone in the Kalutara District (Phillips 1984).

Ecology and behaviour

The stripe-necked mongoose has been recorded in deciduous and evergreen forest, swampy clearings, plantations, open scrub and along watercourses (Webb-Peploe 1947, Van Rompaey and Jayakumar 2003). In deciduous forests it is usually found in swampy clearings, along watercourses, and in open scrub (Krishnan 1972) as well as in rice fields. Of the 11 sightings in Protected Areas, seven were in dry deciduous forest, three in moist deciduous forest, and one in a teak plantation. In Valparai, Anamalai Hills, there were a dozen sightings between April and December of 2002, with animals seen foraging along streams in riverine forests and swamps, and also in tea plantations. In Sri Lanka, its distribution may encompass lowland dry zone forest and it is rarely sighted in disturbed areas or close to human settlements however, the species is adaptable and can tolerate relatively high disturbance.

It is more common in the hills than in the lowlands (Hill 1939), and has been found up to 2200 m (Van Rompaey and Jayakumar 2003). It is diurnal and feeds on small mammals, birds, birds' eggs, reptiles, fish, insects, grubs, and roots (Van Rompaey and Jayakumar 2003). The typical litter size is two to three and an animal in captivity was recorded as living for nearly 13 years (Van Rompaey and Jayakumar 2003). Measurements for Indian males (n = 4) are as follows; mean length of head and body is 52.9 cm, mean length of tail is 31.5 cm; weighs upto 1 kg while females (n = 3): mean length of head and body is 47.4 cm, mean length of tail is 29.7 cm weighs 2.7 kg (n = l) (Pocock 1941). Usually solitary but it's often observed in pairs. It was often sighted in the early mornings and late evenings along stream beds. Most of the time it was observed digging out grubs from the soft damp soil/mud (Phillips 1984). A stripe necked mongoose has been observed chasing even a small chital (*Axis axis*) fawn (Krishnan 1972), mouse deer (*Moschiola indica*) (Johnsingh 1983) or even scavenge on large predator kills (Ramachandran 1985).

Principal threats

There are no major threats to the global population of the stripe-necked mongoose, although major threats are present at the local scale in the form of hunting, trade and road kills. This species is hunted for meat that is eaten by several tribes and for its hair that is used for making shaving brushes, paint brushes, and good luck charms (Hanfee and Ahmed 1999). They are also regularly killed by hunting dogs (Webb-Peploe 1947). All mongoose species are in demand for the wildlife trade (Van Rompaey and Jayakumar 2003), however, this threat is regional in scale. The loss of habitat is a threat however, there is likely to be no significant level of population decline at the species scale.

Conservation measures

The stripe-necked mongoose is on Schedule IV of the Indian Wildlife (Protection) Act, 1972, and has been recorded from many protected areas throughout its range (Van Rompaey and Jayakumar 2003). The Indian population is listed on CITES Appendix III.

1.5.8 Ruddy mongoose

Distribution and status

This species is listed as Least Concern in view of its wide distribution, presumed large population, and because it is unlikely to be declining at nearly the rate required to qualify for listing in a threatened category. The impacts of habitat loss, degradation and hunting on populations are unknown but the population is not suspected to be declining at a rate sufficient to qualify for Near Threatened. This species not only has a wide geographical distribution, but it also occurs in varied vegetation types from arid regions in the plains of northern and western India to high altitudes (> 2000 m) of southern India, as well as in human-dominated agricultural landscapes. More information is needed to determine the true status of this species and there is a need to monitor its trends. The ruddy mongoose is found in southern India: Nilgiri Biosphere Reserve (Yoganand and Kumar 1995), central India and Sri Lanka. There are recent records in northern India from Madhav National Park, Madhya Pradesh, Sariska Tiger Reserve, and Rajasthan. The population status of the ruddy mongoose is unknown, but the species is believed to be common in some forests of central India Ruddy Mongoose is thought to be absent in coastal and evergreen forests of the Western Ghats. They occur in dry forests and forests with rocky outcrops, and are absent in completely barren areas. Kumara and Singh (2007) sighted five animals in Nagarahole, three in Bandipur National Park, one in Hasanur forests in Chamarajnagar and Savandurga forests in Magadi of Bangalore district, and six in Daroji Bear Sanctuary in Bellary district. All sightings were either in the morning or in the evening, in dry forests or rocky areas. Animals were often seen in pairs. They were sighted frequently in Bhadra and Bandipur (Karanth 1986, 1988).

Ecology and behavior

The ecology of the ruddy mongoose remains to be studied. Most records of this species are from forested areas including dry forests, dry thorn areas, and disturbed forests, although there are also fewer records from open areas and secluded rice paddy fields. In India, this species was found exclusively in dry forests, and was never sighted near human settlements. The elevation range is 50 to 2200 m in South

India. The ruddy mongoose is crepuscular, hunting by day as well as by night as it hunts, feeds, and rests in trees. In India, it is frequently sighted scavenging road kills. This species is large (1.7 kg; weight of a road-killed animal) and resembles the grey Mongoose but is reddish-brown with a grizzled appearance and a black-tipped tail (Prater 1971). It has the habit of walking with the tip of its tail turned upwards, a distinctive behavioural trait (Menon 2003).

Principal threats

Little is known of direct threats to the ruddy mongoose but there appears to be no major threats to the global population. Local-scale major threats include hunting and snaring by tribes and villagers.

Conservation measures

The Indian population is listed in CITES Appendix III (Wozencraft 2005), and Schedule IV of Indian Wildlife (Protection) Act, 1972. In central India people consider the mongoose to be sacred, and thus it is not killed there. The species occurs in numerous protected areas.

1.5.9 *Grey mongoose*

Distribution and status

The common grey mongoose (*Herpestes edwardsii*) is listed as Least Concern in view of its wide distribution, presumed large population and adaptability to human-dominated landscapes, and because it is unlikely to be declining at nearly the rate required to qualify for listing in a threatened category. This species is mainly found in southern Asia mainly India, Pakistan, Nepal, Sri Lanka and some other parts of Asia. It is commonly found in open forests, scrub lands and cultivated fields, often close to human habitations.

Ecology and behavior

It lives in burrows, hedgerows and thickets, among groves of trees, taking shelter under rocks or bushes and even in drains. Usually found singly or in pairs. It preys on rodents, snakes, birds' eggs and hatchlings, lizards and variety of invertebrates. Their tail length equals their body length. Body length: 14 to17 inches (36 to 45 cm) Tail length: 17 inches (45 cm), weight: 2 to 4 lb. (0.89 to 1.4 kg). The habitat and ecology of the Indian Grey Mongoose is known from few studies, however, it has been recorded in disturbed areas, in dry secondary forests, and thorn forests, but seems to be a commensal with humans as well. This species was often recorded near human settlements in central India, where it was seen near garbage bins, garbage dumps, scavenging on carrion, and on roads (Choudhury et al., 2011). The species seems to be most common in disturbed areas, in dry secondary forests and thorn forests. This species has been found up to 2100 m (Corbet and Hill 1992) and feeds on insects and snakes (Santiapillai 2000). Home range estimates for Indian grey mongoose (*Herpestes edwardsii*) from the Nilgiri Biosphere Reserve (single individual for a period of four months) was found to be 3.4 to 4.9 hectares (0.039 to 0.049 km^2), and an overall range of 0.15 km^2 (Kumar and Umapathy 1999).

Principal threats

This species has no major threats occurring across the whole of its range, however, it does experience some regional threats. Kalle (2011) notes that the grey mongoose is often captured and sold as a pet. This species is very common in Nagapattinam and often local farmers trap them in cages and sold for meat. Gypsies from northern India use hook snares to capture individuals for skins, which are then sold in local markets in Nepal (Choudhury et al., 2011). All mongoose species are in demand for the wildlife trade (Van Rompaey and Jayakumar 2003): the meat is eaten by several tribes and the hair is used for making shaving brushes, paint brushes, and good luck charms (Hanfee and Ahmed 1999).

Conservation measures

The Indian Grey Mongoose is listed on CITES Appendix III in India (Wozencraft 2005). In 2002 in India, the government upgraded the Mongoose species, to Part II of Schedule II of Wildlife (Protection) Act 1972. This species is found in numerous protected areas. Field surveys, ecological studies, habitat protection and monitoring of threats are needed.

1.6 Justification of study

Effective conservation of small carnivores requires detailed knowledge of their current distribution, population status and ecological requirements. Although our understanding on their distribution and ecology has increased, comprehensive information at a regional scale in most parts of the country is still lacking. The extreme difficulty in physically observing them and the inaccessibility of their habitat is one of the common issues in small carnivore research due to their elusive behavior and rarity which makes it most difficult to study in the wild. There is deficiency in reliable field data on their distribution and conservation requirements. All that is left of them is interesting anecdotal information providing little concrete evidence on their population status. Keeping the aforementioned caveats in mind, this study was initiated from 2009 to 2011 in Mudumalai Tiger Reserve.

1.7 Study objectives

1. To determine the abundance and occupancy of small carnivores in Mudumalai Tiger Reserve.
2. To study the dietary pattern of small carnivores.
3. To determine spatial and non-spatial variables governing the occupancy of small carnivores and
4. To predict the distribution pattern of small carnivores through habitat suitability modeling in Mudumalai Tiger Reserve.

1.8 Organization of the thesis

The thesis is structured into six chapters, each chapter consists of an introduction to the topic, elaboration on methods and analysis used, results arrived at and discussion of the results. First chapter deals with the study species; small cats, civets and mongooses, scope of the study and its objectives. Chapter 2 describes the study area, Mudumalai Tiger Reserve. Chapter 3 deals with the occupancy and abundance of small carnivores. Chapter 4 deals with the dietary pattern of small carnivores. Chapter 5 deals with the differences in spatial and non-spatial variables governing the occupancy of small carnivores and Chapter 6 deals with predicting distribution pattern of small carnivores in and around Mudumalai Tiger Reserve.

Chapter 2

STUDY AREA

2.1 Introduction

What constitutes Mudumalai Tiger Reserve today came under British control only after 1805. Coffee was introduced in the 1850s and large forest tracts were cleared for coffee plantations. A part of the area which was cleared for coffee also witnessed gold-mining between 1874 and 1880. In 1885 however, all these activities ceased with the development of operational and conservation plans. In 1914, the Government took over Mudumalai and declared it a reserved forest. In 1940, an area of approximately 60 km^2 was declared as a Wildlife Sanctuary. In 1977, the whole of Mudumalai and part of the Sigur range were organized as a Wildlife Sanctuary. In 1986, the area was incorporated in the Nilgiris Biosphere Reserve, India's first such reserve. The Mudumalai Tiger Reserve extends along the eastern slopes of the Western Ghats in Tamil Nadu. Primary forests in Asia, particularly in the Western Ghats of Peninsular India, are vanishing at an alarming rate due to the increased human activity (Parthasarathy 1999), and their species composition is altered by generalist or inferior species (Anitha et al., 2007). Out of 2100 species of flowering plants endemic to Peninsular India, about 818 are found in the Nilgiris and adjoining areas (Mohanan and Balakrishnan 1991). The Western Ghats is one of the 8 hot spots of biodiversity in the world (Myers et al., 2000), listed among the 200 globally most important ecoregions (Olson and Dinerstein 1998) in the world.

2.2 Location

The study area, Mudumalai Tiger Reserve (11^0 32'- 11^0 43' N; 76^{0}22' - 76^{0}45' E) was declared a Tiger Reserve in April 2007. It is situated at the tri-junction of Tamil Nadu, Karnataka and Kerala states (Fig 1 and 2.). It is contiguous with Wayanad wildlife Sanctuary on the north-west, Bandipur Tiger Reserve on the north, the south and the east the Singara and Sigur Reserved Forests which forms the boundary of Nilgiri North Division. Mudumalai is located within the Nilgiri Biosphere Reserve (5520 km^2).

Fig 1. The location of Mudumalai Tiger Reserve at the trijunction of Tamil Nadu, Karnataka and Kerala in the Western Ghats.

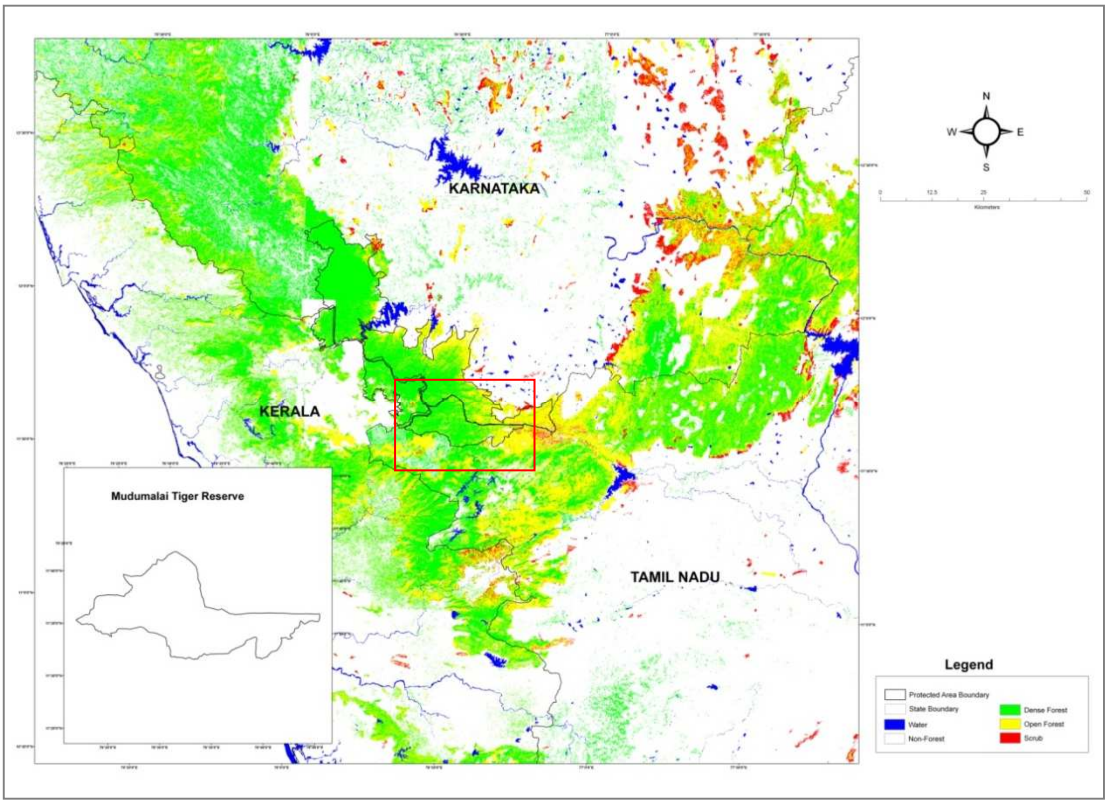
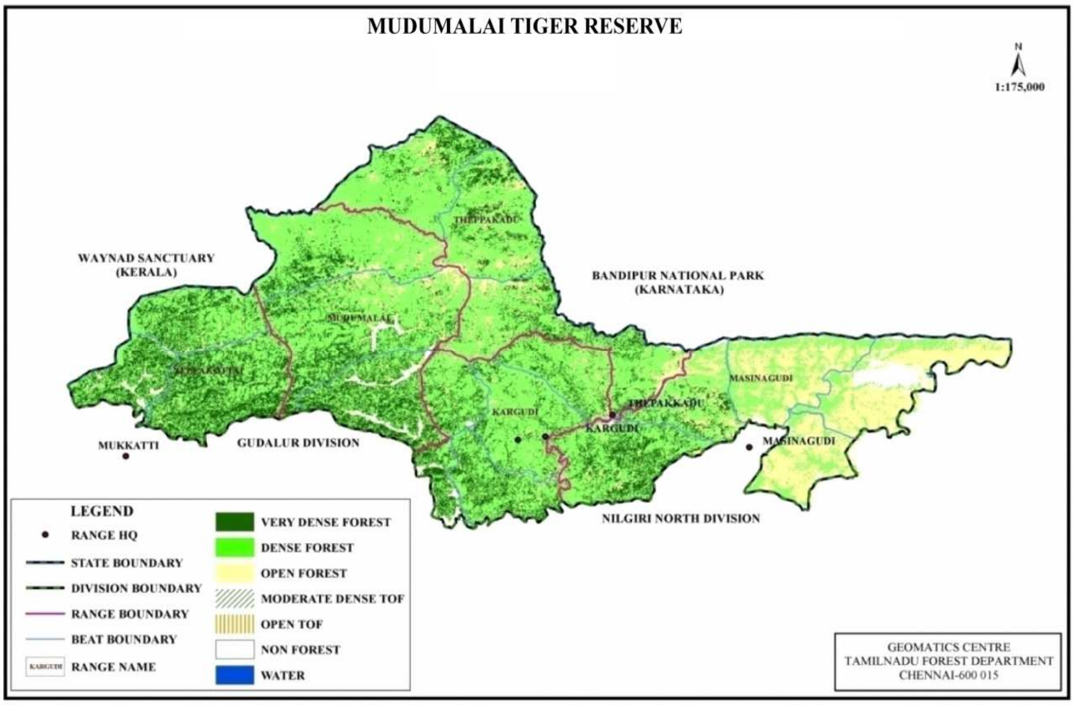


Fig 2. Mudumalai Tiger Reserve, Tamil Nadu, showing administrative zones.



Mudumalai plays an important role in biodiversity conservation by offering habitat contiguity of about 3300 km^2 with three other protected areas in the region, namely Nagarahole and Bandipur National Park and Waynad Wildlife Sanctuary through forest corridors between the Western and Eastern Ghats. Mudumalai is a part of the single largest tiger population in India, acting as source populations for the Northern and Eastern parts of the Western Ghats landscape complex with great potential for long-term tiger conservation (Jhala et al., 2008, Ramesh 2010, Kalle et al. 2011). The Reserve (Sanctuary) was created in 1940, the first in southern India, with a designated area of 60 km^2. In 1956, it was enlarged to 295 km^2 and later to a further 321 km^2, which is its present extent. The Core Zone of this reserve (102 km^2) has been notified as National Park. The present park was under the control of Thirumalapad kovilagam until late 18th century and in 1914 this area was declared as a Reserved Forest. The Reserve has five ranges (Mudumalai, Theppakadu, Kargudi, Masinagudi and Nelakottai), 18 beats and 35 forest compartments.

2.3 Physical features

Soil: There are two types of soils in the Reserve; black sandy loam soil containing over 50% of sand and gravel and red heavy loam soil. The red soil is generally present in the southern part of the Sanctuary, where rainfall is plentiful making the soil fertile. The northern part of the Sanctuary has blackish soil, which does not favour retention of moisture, and possibly has more infertile mineral. The eastern part of the Sanctuary has gravel soil, with very poor rainfall and poor soil moisture content resulting in stunted growth of trees.

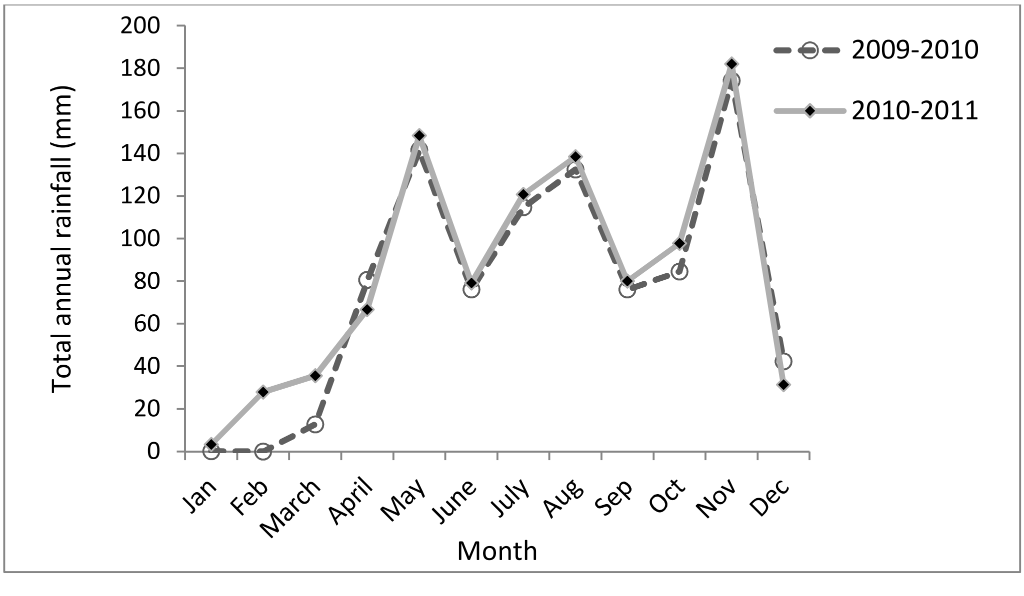
Terrain: The terrain is mostly gently undulating but flat towards the eastern portion with an average elevation ranging from 960 m to 1266 m. Morganbetta of Masinagudi range is the highest peak (1266 m) in Mudumalai. Narathibetta on the western side of the park is the second highest peak (1188 m). The park areas located in Wayanad plateau is characterized by the presence of several swamps and `vayals’ varying in size.

2.4 Climate

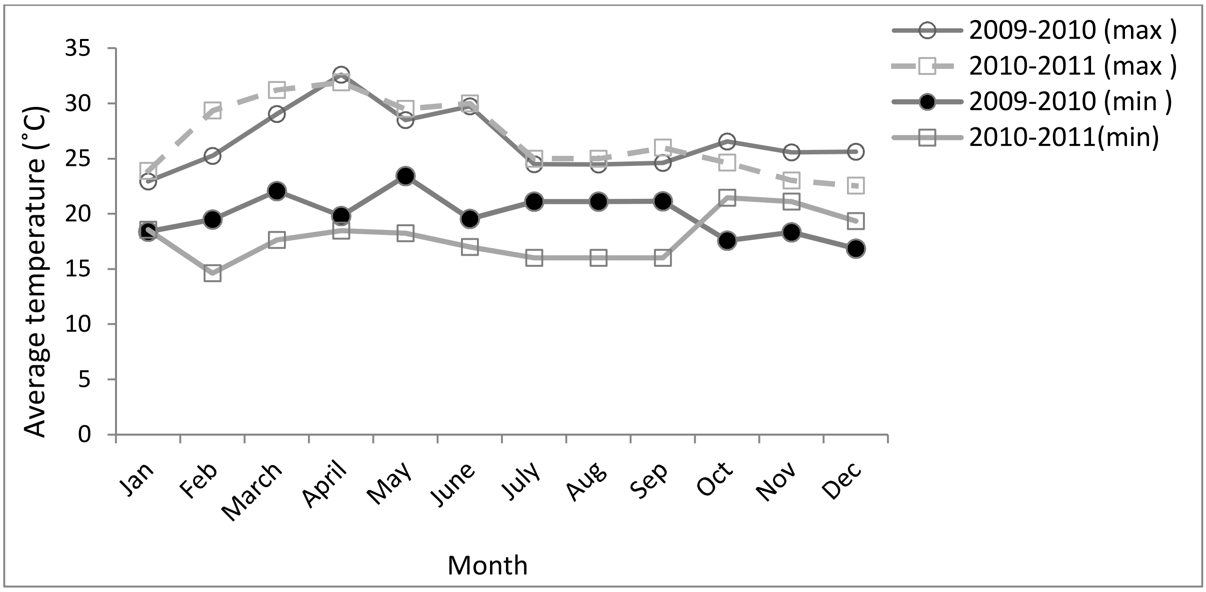
The Reserve has a long wet season and a short dry season. It receives rainfall from south-west and north-east monsoons. Two peaks of rainfall were observed in the eastern part of the reserve, one during the month of May (140-160 mm) and the second peak during November (180 mm). The western part of the Sanctuary receives rainfall almost throughout the year starting from March and the peak rainfall is received during the months from June to October. The south-west monsoon starts by May and ends by August whereas the north- east monsoon starts by September and ends by December. Based on the climate of the area, there are three distinct seasons recognized: dry season (January to April), first wet season (May to August) and second wet season (September to December) (Varman and Sukumar 1995). There is a decreasing rainfall gradient from the west and south to the east and north (Suresh et al., 1996). The rainfall has a marked east-west gradient, with the eastern areas getting the least amount of heavy rains (1000 to 2000 mm). The temperature ranges from 8^0C in December to 35^0 C in April. Annual rainfall and temperature in Mudumalai Tiger Reserve are given in Fig. 3a and b).

Fig 3. Total annual rainfall pattern in Mudumalai Tiger Reserve, Tamil Nadu (January 2009 to December 2011). Source- Weather monitoring station, Mudumalai, Indian Institute of Science, Centre for Ecological Sciences.

a)



b) Annual temperature in Mudumalai Tiger Reserve, Tamil Nadu (2009-2011).



2.5. Hydrology

Mudumalai is characterized by the presence of several swamps and vayals which provide wallowing grounds for herbivores. The central part of the Reserve is slightly elevated with seasonal streams and a perennial river. The Moyar River is an important source of water and with its tributaries drains the Tiger Reserve. It originates from the Nilgiri Mountains near the Pykara falls and meanders through the Reserve for a distance of 20 km. On the eastern part of the Reserve there is an artificial canal through which water is drawn from the Maravakandy reservoir from Masinagudi to Moyar reservoir (about 9 km) for power production and this is the major water source for wildlife. Avarhalla is another perennial water source in the eastern part of the Sanctuary. Throughout the year, seepage water from Maravakandy dam flows through Avarhalla draining the forest. Mavanhalla is a perennial stream found in the western side of the sanctuary and about two decades back it had become a seasonal stream due to a check dam that was built across this river to supply water to Bokkapuram and Mavanhalla villages. There are four major perennial artificial waterholes; Ombetta, Game hut, Compartment no.3 check dam and Narathi check dam. The seasonal waterholes include Mavinhalla, Bidharhalla, Segur River, Doddakattihalla, Hebhalla, Imberhalla, Bennahole, Kakkanhalla and Mukkattihole. Water in these seasonal streams flow only during rainy season but water is available for wildlife in puddles throughout the year. During the months of March, April, and May water flow in Moyar gets almost cut off, in situations like this water from Singara power house is released so as to maintain the water flow in the river.

2.6 Vegetation

Champion and Seth (1968) classified the vegetation type in Mudumalai as Southern Tropical dry thorn forest, Southern Tropical dry deciduous forest, Southern Tropical moist deciduous forest, Southern Tropical semi-evergreen, Moist bamboo brakes and Riparian fringing forest.

Southern Tropical dry thorn forest: Tropical dry thorn forest occurs along the foothills of Nilgiris on the eastern side of the Sanctuary. This region receives an average annual rainfall of 600-900 mm. The trees are stunted and the following major species are; *Acacia chundra*, *A. suma*, *A. leucopholea*, *Premna tomentosa*, *Dalbergia lanceolaria*, *Anogeissus latifolia*, *Ziziphus xylopyrus*, *Ziziphus mauritiana*, *Sapindus emarginatus*, *Erythroxylon monogynum*, *Canthium parviflorum*, *Acacia pennata*, *Randia dumetorum* and *Capparis spp.*

Southern Tropical dry deciduous forest: Dry deciduous forest occurs over a major portion of the Tiger Reserve where the rainfall is between 900 mm and 1200 mm. This type of forest extents up to the northern boundary and on the west, dry deciduous forests can be seen up to Game hut. Tall grasses like *Themeda triandra* and *T. cymbaria*, *Heteropogon contortus* and *Cymbopogan flexuous* are found in patches of open canopy. The trees found in this habitat shed their leaf during dry season and it is prone to fire during dry season. The common tree species are *Anogeissus latifolia*, *Grewia tilifolia*, *Terminalia crenulata*, *T. tomentosa*, and *Kydiya calycina* and *Tectona grandis.* The Doddagatti block is widely occupied with *Shorea roxburghii.*

Southern Tropical moist deciduous forest: Moist deciduous forest is found in regions where the rainfall is between 1600 and 2000 mm. This type of forest is found in Benne, Mudumalai and Theppakadu blocks. The canopy is closed in this forest and the trees are tall. The common tree species in moist deciduous forest includes *Lagerstroemia microcarpa*, *Terminalia crenulata*, *Tectona grandis* and *Dalbergia latifolia*. The ground vegetation includes several species of Orchids, *Amorphophallus*, *Zingiber*, *Curcuma* and *Solanum spp.*

Southern Tropical semi-evergreen: These forests are found in the Southwest region of Mudumalai. The rainfall in this region exceeds 2000 mm. It is characterized by closed canopy (80-90%), high tree richness, vines, lianas and epiphytes. The common tree species in this habitat include *Olea dioca*, *Glochiodion velutinum*, *Toona ciliate*, *Elaeocarpus tuberculatus*, *Casseria ovoides*, *Litsea mysorensis* and *Cinnamomum malabaricum*. Climbers like *Todalia asiatica*, *Watakaka volubilis*, *Gnetum ula*, *Entada scandens* are also found in semi evergreen forest.

Moist Bamboo brakes: Bamboo thickets are found amidst dry deciduous, moist deciduous and semi evergreen forests. There are two species of bamboo; *Bambusa arundinacea* and *Dendrocalamus strictus.* They are found along the fringes of riparian forests and swamps.

Riparian forest: Among all forest types, from a vantage point, a green strip of riparian forest is clearly visible along the dry seasonal and perennial streams. Tree species found in this type of forest remain green throughout the year. The common plant species found in riparian forests include *Mangifera indica*, *Pongamia glabra*, *Terminalia arjuna*, *Bischofia javanica*, *Linociera malabarica*, *Syzygium cumini*, *Dalbergia latifolia*, *Bambusa arundinacea* and *Dendrocalamus strictus*.

2.7 Fauna

Thirteen percent of mammal species found in India are present here. There are about 50 species of fish, 21 species of amphibians, 34 species of reptiles, 227 species of birds and 55 species of mammals reported from the reserve. The diversity of mammals is high in the dry deciduous and dry thorn forest than in the other habitats. Tiger, leopard and dhole are the three major carnivores present in the study area. Mudumalai supports a fabulous assemblage of herbivores such as chital (*Axis axis*), sambar (*Rusa unicolor*), muntjac (*Muntiacus muntjak*), wild pig (*Sus scrofa*), Indian chevrotain (*Tragulus meminna*), gaur (*Bos gaurus*), Asian elephant (*Elephas maximus*), four-horned antelope (*Tertracerus quadricornis*), black-buck (*Antelope cervicapra*), black-naped hare (*Lepus nigricollis*), and arboreal mammals such as bonnete macaque (*Macaca radiata*), common langur (*Seminopithecus entellus*), Indian giant squirrel (*Ratufa indica*) and Indian flying squirrel (*Petaurista philippensis*) are also found here. The area supports a wide variety of medium and small sized omnivore and carnivores such as sloth bear (*Melursus ursinus*), striped hyena (*Hyaena hyaena*), jungle cat (*Felis chaus*), leopard cat (*Prionailurus bengalensis*), rusty spotted cat (*Prionailurus rubiginosus*), common palm civet (*Paradoxurus hermaphroditus*), small Indian civet (*Viverricula indica*), grey mongoose (*Herpestes erdwardsii*), ruddy mongoose (*Herpestes smithii*) and stripe-necked mongoose (*Herpestes vitticollis*). Mudumalai is also domicile to the Indian porcupine (*Hysterix indica*), and Indian pangolin (*Manis crassicaudata*). There are a variety of small mammals present such as lesser bandicoot rat (*Bandicota bengalensis*), black rat (*Rattus rattus*), Indian gerbil (*Tatera indica*), white-tailed wood rat (*Cremnomys blanfordi*), soft-furred field rat (*Millardia meltada*), house mouse (*Mus musculus*), little Indian field mouse (*Mus booduga*), Bonhote's mouse (*Mus famulus*), spiny-tailed field mouse (*Mus platythrix*) and Malabar spiny dormouse (*Platacanthomys lasiurus*). Deciduous forest habitats support the highest small mammal abundance and biomass in the reserve (Venkataraman et al., 2005). Domestic livestock (cattle, buffalo and goat) occur in the village areas inside the Sanctuary.

2.8 Ecological, economic history and anthropogenic pressure

The Nilgiri areas and Sigur Plateau have been populated for several years, and local people have been in cultural contact with the rest of India for centuries. The entire Niligiri range was predominantly pastoral country, that was populated by various hunter-gatherers and shifting cultivating groups. Between 1886 and 1890, the administration surveyed the forest areas, declared large tracts of the Sigur Plateau as Reserved Forest and started to log the commercially valuable sandalwood. The State also generated revenue by collecting grazing fees, selling fuelwood coups, leasing rights for collection of NWFPs and selling bamboo and other timber, apart from sandalwood. Presently there are 21 tiny hamlets (settlements) within the Reserve. People living in these settlements include Mountain Chetties and tribes; Kattu Naickers, Paniyas, Kurumbas, Irula. Mountain Chettie's primary occupation is to perform agricultural practices in swamp vayal habitats and rear cattle. The Paniya tribes work in agriculture fields of Chetties and they also collect tuber, honey, and perform fishing operations in the Reserve for their sustenance. There are also Kurumba, Kattu Naicker tribes and some Irula living near Theppakadu, Kargudi and Thoraplli. The Kurumbas are basically hunter-gatherers. Kurumba, Kattu Naickers and Irula collect tuber, honey, and mushroom and also scavenge on carnivore kills. These people work as fire watchers, tourist guides mahout and anti-poaching watchers. Moyar is a non-tribal village located as an enclave in the eastern part of the park. This is a village with electricity board employees and a few resident Irula tribes, a grazier and agrarian tribal community. Retired electricity board employees have settled in this village. The village has expanded in the last ten years by encroaching and clearing the revenue land adjacent to the forest areas. These villagers cultivate ragi, coconut, garlic, cabbage, beans etc. These villagers own large number of livestock. Cattle grazing in the forest occur illegally. There are fairly a few villages abutting the eastern and southwestern part of the park, to name a few; Mavanhalla, Chemmanatham, Masinagudi, Bokkapuram, Singara, Thorapalli and Bospara. These villagers cultivate paddy, tapioca and raise plantations like tea, coffee and pepper. Use of forest resources in Mudumalai is influenced by multiple factors. In terms of livelihood of the traditional communities, livestock rearing and collection of non-wood forest products (NWFPs) are very important, the latter both for cash income and subsistence use. Peripheral communities used the forest resources in a varied fashion, with NWFPs contributing differently to different income groups. Where there was no restriction on forest use, higher income groups used resources more heavily than lower income groups, and hence would suffer most from any restriction on forest use. People's reliance on forests evidently declined with increased level of education and opportunities in non-forestry vocations (Hegde and Enters 2010). Forests were still very important to the household economy of the local people both in terms of food security and cash income.

Chapter 3

ESTIMATING RELATIVE ABUNDANCE AND OCCUPANCY OF SMALL CARNIVORES

3.1 Introduction

In many monitoring programs one of the most difficult steps to overcome is obtaining data on elusive species. It is all the more challenging to estimate abundance of some species that are nocturnal, cryptic and not individually identifiable from natural markings, by using conventional non-invasive techniques, such as distance sampling or photographic capture-recapture methods. In biodiversity monitoring, it has become increasingly important to use presence-only data due to the high cost of collecting count data with a strict protocol. However, recent advances in 'occupancy modeling' of animal presence data derived from photographic captures might provide solutions to the problems of monitoring elusive species. Site occupancy (presence/absence) modeling is recognized as an effective technique for monitoring populations of secretive species on a landscape scale, historically considered a daunting or even impossible task. Site occupancy modeling may be the only feasible metric for monitoring population status of some species, such as those with recapture probabilities too low to use mark-recapture models effectively (Dorcas and Willson 2009). For categorical, presence–absence data to be useful, however, estimates of species-specific detection probabilities must be incorporated into surveys. A specific occupancy approach – the Royle and Nichols (2003), allows for reliable estimation of abundance at best, and of an index of abundance or occupancy rate at the least, without the need for individual identification of animals. A variety of methods are available for estimating animal abundances but all involve the issue of estimating detection probabilities for specific kinds of count statistics (Seber 1982, William et al., 2002). Depending on the species being studied, the techniques available for gathering appropriate data, and incorporating the limitations of time, money and effort, only one or just a few of these methods may be suitable.

The density of small carnivores is perceived to be relatively low, thus making mark-recapture studies highly impractical for conspecifics. It is not always possible to obtain photographs of individuals within the species from camera traps, however some studies have successfully applied a camera trap based mark-recapture framework to determine densities of fossa (*Cryptoprocta ferox*), Malagasy civet (*Fossa fossana*) and genet *Genetta genetta*, fisher (*Martes pennant*), Geoffroy's cat (*Leopardus geoffroyi*) (Gerber et al., 2010, Sarmento et al., 2010, Jordan et al., 2011, Pereira et al., 2011). Mark-recapture methods require repeated efforts to capture or observe animals (Otis et al., 1978, Pollock et al., 1990) and even observation-based methods such as distance sampling (Buckland et al., 2001) or multiple observers (Cook and Jacobson 1979, Nichols et al., 2000) are viewed as being time and effort consuming. Despite the logistic constraints, these methods have been widely applied. A potential approach to estimating abundance involves shifting the focus from numbers of animals to numbers of sample units occupied by animals (Royle and Nichols 2003). Methods employing this general approach are based on presence-absence data from the sampling units. Royle and Nichols (2003) have developed a model based on this focus to estimate abundance from repeated presence-absence data or point counts. They link the probability of detecting presence and the abundance at a sampling unit by using repeated detection-non-detection data gathered from occupancy surveys, suggesting a maximum likelihood approach at estimating the parameters (that includes abundance). In spite of the relative ease with which presence-absence data may be gathered, achieving large samples for analysis as suggested by Royle and Nichols (2003) for even practical estimates of the parameters might be difficult. Therefore this study has relevance to advancing both the scientific understanding and conservation of a variety of rare and endangered animal species in the study area and elsewhere in the world.

Management and conservation of species relies heavily on understanding the variation in population abundance or density of a target species. Many small carnivore species, which are of special concern for conservation and management, tend to be difficult to study due to their elusive habits, diminutive size and crepuscular or nocturnal activities. The population density of carnivores is governed by several factors, including prey availability (Carbone and Gittleman 2002, Ramesh 2010), habitat structure (Pereira 2009) and hunting by humans (Duckworth et al., 2005).

Camera trapping is becoming increasingly common in documenting elusive lives of small carnivores (Gardner et al., 2010, Gerber et al., 2010, Pereira et al., 2011). Despite the frequent need to estimate the effective trapping area, there is still much debate on a robust solution; most recommendations suggest variations in *ad hoc* boundary-strip methods (Soisalo and Cavalcanti 2006, Maffei and Noss 2007). Field studies have recently provided empirical support for the use of a maximum likelihood spatially-explicit capture-recapture model (Kalle et al., 2011). Despite the availability of newer models, it is still common for studies to use traditional *ad hoc* density estimation techniques (Negroes et al., 2010). Small carnivores often have low detection rates, even with intense sampling efforts, which can either inhibit the application of closed capture-recapture analyses or simply provide imprecise estimates (White et al., 1982, Maffei et al., 2004).

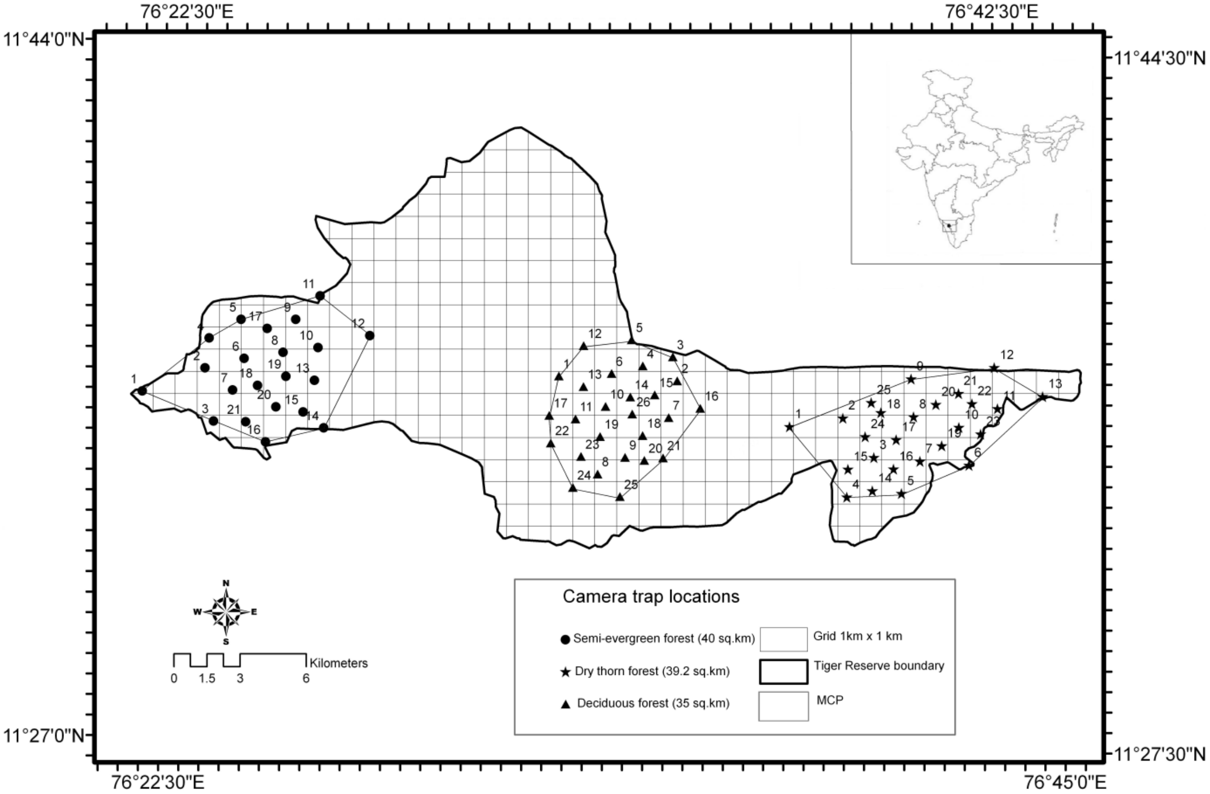
However, modern analytical approaches that shift focus from counting animals to counting patches or sites potentially occupied by them can permit estimating populations of even cryptic and non-uniquely identifiable species. They must also deal with the fact that not all animals present even within a sampled unit are detected during the survey, and therefore that the probability of detecting an animal in the sampled area is often less than one (Williams et al., 2002, Karanth et al., 2004). Camera traps can indicate relative abundance of a species with the assumption that photo detection rates are related to animal abundance (Morruzzi et al., 2002). Although trapping rate surveys (e.g. Carbone et al., 2001) have been used as indicators of animal abundance, detection probability is generally not estimated while doing so. As a result, its use is controversial both on theoretical and practical grounds (Jennelle et al., 2002). Although Rowcliff et al., (2008) recently published an alternative approach to estimating animal abundance from photographic captures this method requires prior biological knowledge of movement rates of animals as well as completely random placement of traps, potentially rendering its use in the field difficult. If species presence data from camera trap photos could be used to model and derive animal abundances - or at least to derive reliable indices of abundance, (which can effectively deal with imperfect detections) - it will be a major advance in animal monitoring science.

In this chapter, I investigated the possibility of estimating abundance of small carnivores using presence/absence data from repeated samples using camera traps. The present study also attempted to compare small carnivore occupancy and detection probability between habitats and seasons in Mudumalai. I also compared jungle cat density using various estimators based on camera trapping from mark-recapture. Further I hypothesized body weight correlates of small carnivores with the following attributes; home range, detection probability, site occupancy and relative abundance index.

3.2 Materials and methods

I selected 114 km^2 of intensive study area which was divided into three intensive sampling zones; deciduous (35 km^2), semi-evergreen (40 km^2) and the dry thorn forest (39 km^2) to conduct camera trap surveys for two years (2009-2010 and 2010-2011) (Fig 4). Camera trap survey was carried out in the dry and wet season in deciduous and dry thorn forest while the semi-evergreen forest was sampled only in the dry season due to inaccessibility in the wet season. I conducted a thorough preliminary survey to identify suitable sites where indirect evidence, such as tracks or scats was detected. I deployed passive-infrared camera traps in a systematic grid 1 x 1 km^2 using Deercam DC300 (DeerCam, Park Falls, USA) and Stealthcam (Bedford, Texas, USA). The mean inter-camera trap distance was 1.31 km. Each year I deployed 25 pairs of camera traps in the deciduous forest, 21 in the semi-evergreen forest and 25 in the dry thorn forest. Each camera trap survey was sampled for 30 days. Stations consisted of two independently operating passive-infrared cameras mounted on opposite sides of a trail or dirt road to get photocaptures of small carnivores. Cameras were kept approximately 25 cm above the ground and set to be active for 24 h/day. No bait or lure was used at any location to attract animals. The photocapture delay was set to 1 min and sensitivity was set to high. I checked sampling stations on an average, every 3 days to ensure continued operation and replaced batteries and film when necessary.

Fig. 4. Camera trap locations for capturing small carnivores in Mudumalai Tiger Reserve (2009-2011).



3.2.1 *Relative abundance index*

Photographic rate is the relative index of the animal’s spatial use and a crude abundance estimate (Carbone et al., 2001). Photographs provided information on date and time of the picture taken. Sometimes, individuals were photographed from only one of the two cameras operating at a single camera station. The event of capturing an individual, whether it was photographed by two camera traps or one, was considered to be an independent record of that individual. On some occasions, individuals were captured more than once at a camera station during a period (<1 min); thus, to avoid pseudo-replications, I considered the first capture of the animal as an independent record. Photos with more than one individual in the frame were counted as single detection for the species. I calculated relative abundance index (here after called RAI) for each camera trap location by dividing the total number of independent records from the total trap nights x 100 (Carbone et al., 2001, O'Brien et al., 2003). Hence the RAI for each species is presented as the number of photo captures per 100 trap nights. RAI was used to facilitate comparisons with other studies. To compare RAI of a species between habitats and seasons, I used Student's t-test in program SPSS 16 (SPSS 2007).

3.2.2 *Occupancy analysis*

Since the identification of individuals within some small carnivore species is difficult, the estimation of abundance using capture-recapture models used in camera-trap surveys of large cats was not possible. One possible solution is to use occupancy as a surrogate for abundance (MacKenzie and Nichols 2004). MacKenzie et al., (2002) developed a model to estimate site occupancy and detection probability based on repeated presence–absence data from multiple sites. Royle and Nichols (2003) extended this model to allow for abundance-induced heterogeneity. The Royle and Nichols (2003) model assumes that populations are closed and individuals are distributed in space according to a Poisson process. If these assumptions are violated, the estimated parameters should not be interpreted as abundance but rather as a random effect (MacKenzie et al., 2006). However, occupancy estimates will still be less biased than models that do not include heterogeneity. For analysis, I defined the minimum time between two independent events as 1 min. This means that if the same species was photographed more than once by the same camera in the course of 1 min, this was only counted as one event. Capture histories were developed for each location consisted of a string of 10 trapping occasions by pooling 3 successive days as 1 trapping occasion (eg., days 1–3 = first trapping occasion, days 4–6 = second trapping occasion, etc.). For each occasion the target species could get a 1 or a 0, where 1 indicates that the animal was captured at the sampling station during that trapping occasion and 0 if it was not captured. This was necessary in order to increase the detection probability for each sampling period. I used Royle and Nichols (2003) heterogeneity model and repeated-count method (Royle 2004). The Royle and Nichols (2003) model assumes that variation in animal abundance from one site to the next is probably the most important source of heterogeneity in detection probability among sites. They suggest that by basing heterogeneity in detection probability on varying site-specific animal abundance, it is possible to exploit this very relationship to estimate the abundance parameter, and consequently derive the probability of occupancy. They use the occupancy based approach and assume that the detection probability of a given species at a particular site is directly dependent on the abundance of that species in that site for a given animal-specific detection probability. Consequently, the heterogeneity in detection probabilities across sites is caused by the heterogeneity in abundance across those sites. And, by modeling the variation in abundances according to some probability distribution model (e.g., Poisson, when the number of animals inhabiting one site is random and independent of the number of animals at other sites), they build a model based on maximum likelihood to arrive at estimates of abundance in these sites.

Modelling was conducted based on *a priori* ecological information and home range estimates of these species from available literature. Small Indian civet, common palm civet and small cat home ranges vary from 2 to 6 km^2, hence I produced jackknife abundance estimates from Royle and Nichol's (2003) model assuming that each capture (1 = present, 0 = absent) at every trap site is independent of other sites. There could be a possibility that these species would even travel beyond 1 km^2 thus modeling based on the aforementioned assumption would give unbiased abundance estimates. The Royle and Nichols (2003) model assumes that populations are closed and that individuals are distributed in spaces according to a Poisson process , and the probability of detecting an animal at a site is a function of how many animals are actually at that site. The spatial distribution of animals is simply how many animals occur at each camera site within the study area. Each of the camera sites will contain some number of animals (some sites may contain 0 animals). That number, the site abundance, is a function of the mechanism governing the distribution. The spatial distribution of animals can meet Poisson assumptions when the number of animals inhabiting one camera site is random and independent of the number of animals at other sites.

While mongooses have home range sizes ranging from 0.1–1.5 km^2 (Kumar and Umapathy 1999) and brown palm civet (Mudappa 2001) ranges from 0.35 to 29.58 ha, I calculated average jackknife abundance estimates from Repeated count model (Royle

2004) assuming that animal abundance in the form of photographic captures could include multiple individuals at a camera site since the inter-camera trap distance exceeds the species mean home range. Eventually there are chances of multiple individuals occupying a camera trap site. Both models were analyzed in program Presence 4.1 (http://www.mbrpwrc.usgs.gov/software/presence.html).

To investigate the relative abundance for nine species of small carnivores I looked at differences in detection probability, occupancy rates and average abundance between seasons in each habitat. Hence estimated average abundance (λ) is interpreted as individuals per km^2. Mean abundance estimates were compared between habitats and seasons using Student's t-test in program SPSS 16 (SPSS 2007). I performed a Pearson correlation test between mean abundance (λ) as obtained from occupancy approach and relative abundance index (captures/100 trap nights), body weight (kg) and home range (km^2), body weight (kg) and mean detection probability (*p*), body weight (kg) and mean site occupancy (ψ), and body weight (kg) and RAI. Body weight (kg) and home range (km^2) are mean values across available studies (Table 9).

3.2.3 *Jungle cat density estimation*

Density was estimated only from captures in the dry thorn forest due to insufficient captures in the other sampling zones. Stations consisted of two independently operating passive-infrared cameras mounted on opposite sides of a trail or dirt road to provide a photographic-capture of both flanks of each animal, thus improving individual identification in recaptures. However due to independent sensors in each camera, on some occasions I failed to get enough pictures of both flanks largely due to the delay in response of the sensor, which resulted in single flank pictures. Due to the difference in right and left flank pictures, I used the flank which yielded maximum unique individuals for abundance estimation. The size of the minimum convex polygon (MCP) formed by joining the peripheral camera locations was 39.2 km^2 (Fig. 4). The sampling effort for each survey was 690 trap nights (23 locations × 30 days).

No bait or lure was used at any location to attract jungle cats. Capture histories, developed for each jungle cat older than a year was identified by camera-trapping.

Individuals were distinguished in photographs by their distinctive pelage pattern, body shape or sex after which I constructed capture histories necessary for closed capture-recapture analyses. The pictures that failed to present diagnostic details to identify individuals (n = 11, 16.6% of the total pictures obtained across both years) were excluded from analysis.

After applying a buffer equivalent to 1/2 of the mean maximum distance of recaptures (MMDM) to each sampling station, no ‘holes’ were left within the sampling areas such that no individual cat could have an entire home range located between sampling stations (Karanth and Nichols 2002, Reppucci et al., 2011). Hence the mean inter-camera trap distance was maintained at 1.31 km. I checked sampling stations on an average, every 3 days to ensure continued operation and replaced batteries and film when necessary. To increase individual capture probability and to make it > 0.10 per trapping occasion, as recommended by Otis et al., (1978) and White et al., (1982), the trapping history of each individual consisted of a string of 10 trapping occasions by pooling 3 successive days as 1 trapping occasion (eg., days 1–3 = first trapping occasion, days 4–6 = second trapping occasion, etc.). For each trapping occasion an individual could get a 1 or a 0, where 1 indicates that the animal was captured at any of the sampling stations during that trapping occasion and 0 if it was not captured.

I estimated the population size using software CAPTURE (Rexstad and Burnham 1991), considering the results of model M_h, which assumes heterogeneity among individuals in their capture probabilities (White et al., 1982) and is the most appropriate model for felids (Periera et al., 2011) because of the unequal access to sampling stations by different individuals (Karanth and Nichols 2002).

To estimate the density of jungle cat using the method described by Karanth and Nichols (1998), I applied the standard camera-trapping and capture–mark–recapture population models (Otis et al., 1978, Pereira et al., 2011). Density was estimated as $D = N/A(W)$, where N is the estimated abundance produced by CAPTURE and $A(W)$ is effective trapping area depending upon the buffer applied. I applied two methods to calculate $A(W)$. In the first method I applied three buffers to the convex hull of the camera trap locations (Karanth and Nichols 1998, 2002) to estimate the area effectively sampled by the surveys: (1) Because no published information about the home-range size of jungle cat exists I used home range estimates from cats of similar body size; for example mean home-range radius of the female Geoffroy's cat (*Leopardus geoffroyi*); 0.77 km (Manfredi et al., 2006) and female European wildcat (*Felis silvestris*) 0.85 km (Monterroso et al., 2009), hence I used 0.81 km as the mean home range radius (2) the mean maximum distance moved (MMDM) for the individuals recorded at more than one station (Periera et al., 2011). Because of the overall small number of individuals recorded at more than one station in our study, I calculated the MMDM by pooling the data from both the years (n = 4) (3) and half of the MMDM.

Finally, the absolute density (and SE) was estimated following Silver et al., (2004) and Maffei et al., (2004), using each of the three buffer types mentioned above. One critical assumption for the closed population estimate is that the population should be demographically and geographically closed (Otis et al., 1978, Rexstad and Burnham 1991). Therefore, I tested the closure assumption by applying a closure test implemented in CAPTURE.

The density was also estimated using maximum likelihood-based spatially explicit capture–recapture model (ML-SECR) in program DENSITY 4.45 (Efford 2009). The likelihood-based ML-SECR model has two components: state mode which describes the distribution of the animal's home range centers in the landscape and observation model which describes capture probability as a function of the distance from the home range center to the trap location (Pledger and Efford 1998, Efford et al., 2004). In this analysis, Poisson distribution was assumed, buffer of 1.2 km was applied, and proximity trap option was chosen which allowed for multiple captures on the same occasion. Half normal detection function was fitted to the distance between the home range center and trap.

An estimation of the sex ratio of jungle cat for both the years was performed by identifying the sex of the individuals by observation of the genital area when pictures made it possible. In the remaining cases, I attempted to identify sex using other characteristics (Fig. 5) such as body robustness, the relative size of the head, and the robustness of the jaw and forehead (Pereira 2009).

Fig 5. Camera trap photographs of jungle cat with individual identification markings on the body in Mudumalai Tiger Reserve.

a) A male jungle cat identified as JL2 was photographed on Jagalikadav trail on 1 March 2010 at 5:00 am.

b) JL2 was recaptured on an elephant trail in the Moyar beat on 8 March 2010 at 00:45 am.

c) A female jungle cat identified as JL11 was photographed on an elephant trail in the Moyar beat on 16 March 2010 at 23:27 pm.

3.3 Results

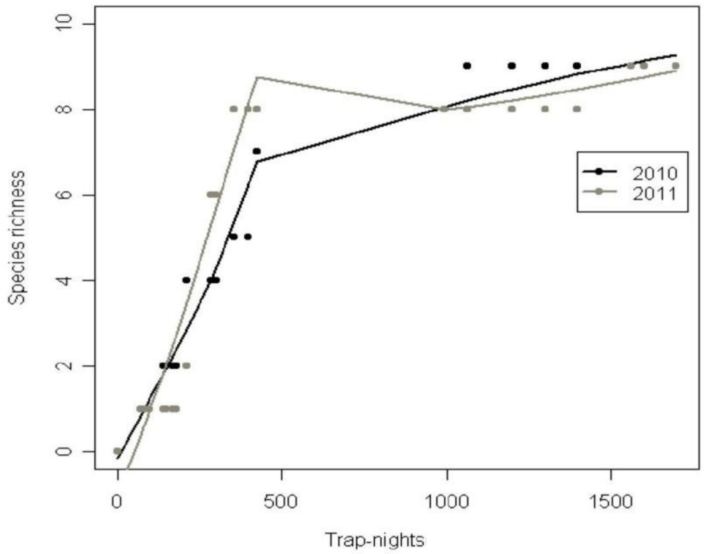
3.3.1 *Capture success*

Totally, 7380 trap-nights yielded 448 photographs of nine small carnivore species of which 16.07 % (n = 72) were of jungle cat, 1.34 % (n = 6) leopard cat, 2.46 % (n = 11) rusty-spotted cat, 19.87 % (n = 89) small Indian civet, 8.26 % (n = 37) common palm civet, 4.46 % (n = 20) brown palm civet, 14.73 % (n = 66) stripe-necked mongoose, 21.43 % (n = 96) ruddy mongoose and 11.38 % (n = 51) grey mongoose (Table 1).

3.3.2 *Sample adequacy*

The first appearance of species occurred on an average of 9.25 camera-days. The number of days a species was first captured did not differ between seasons (t test, t = 0.26, df = 7, P = 0.83). Small carnivores were captured on an average 8.32 camera-days in the dry thorn forest, 12.8 camera-days in deciduous forest and 16.4 days in semi-evergreen forest. Small carnivore richness stabilized on an average of 1500 trap nights (Fig 6).

Fig 6. Increase in species richness with cumulative number of camera trap-nights in Mudumalai during 2010 and 2011.



3.3.3 *Relative abundance index*

Average capture frequencies of small carnivores across both years ranged from 0.13 captures/100 trap-nights to 2.56 captures/100 trap-nights. Capture frequencies for jungle cat ranged from 0.13-3.33 captures/100 trap-nights), leopard cat 0.13-0.4 captures/100 trap-nights, rusty-spotted cat 0.13-0.51 captures/100 trap-nights, small Indian civet 0.63-2.38 captures/100 trap-nights, common palm civet 0.4-1.15 captures/100 trap-nights, brown palm civet 0.13-1.9 captures/100 trap-nights, stripe-necked mongoose 0.13-3.47 captures/100 trap-nights, ruddy mongoose 0.53-1.87 captures/100 trap-nights and grey mongoose 0.13-2.18 captures/100 trap-nights (Table 1).

Jungle cat RAI did not differ between seasons in dry thorn forest (t test; t = 0.88, df = 1, P = 0.53). Small Indian civet RAI did not differ between seasons in deciduous forest (t test, t = 1.2, df =1, P = 0.44) and dry thorn forest (t test, t = -0.88, df = 1, P = 0.47). In the dry season small Indian civet RAI did not differ between dry thorn and deciduous forest (t test, t = 1.07, df =1, P = 0.40), dry thorn and semi-evergreen forest (t test, t = -0.77, df = 1, P = 0.60) and deciduous and semi-evergreen forest (t test, t = 0.02, df = 1, P = 0.98). Small Indian civet RAI was similar for dry thorn and deciduous forest even in the wet season (t test, t = 0.94, df =1, P = 0.52).

Common palm civet RAI did not differ between seasons in dry thorn forest (t test, t = 0.89, df =1, P = 0.54). It did not differ between dry thorn and deciduous forest in the dry season (t test, t = -1.63, df = 1, P = 0.35) and wet season (t test, t = -0.50, df = 1, P = 0.71). Stripe-necked mongoose RAI did not differ between seasons in deciduous forest (t test, t = -0.40, df =1, P = 0.76). It did not differ between deciduous and semi-evergreen forest in the dry season (t test, t = 1.16, df = 1, P = 0.45).

Ruddy mongoose RAI did not differ between seasons in deciduous forest (t test, t = 0.08, df = 1, P = 0.94) and dry thorn forest (t test, t = 3.12, df = 1, P = 0.20). It did not differ between forest types in the dry (t test, t = 1.76, df = 1, P = 0.22) and wet seasons (t test, t = -0.45, df = 1, P = 0.73).

Grey mongoose RAI was significantly higher in dry thorn than deciduous forest (t test, t = -12.76, df = 1, P = 0.02) in the dry season. Although it was higher in the dry thorn than deciduous forest in the wet season, it was not significant (t test, t = -5.92, df = 1, P = 0.11, Table 1b).

Table 1. Photocaptures and relative abundance index (captures/100 trap nights) of small carnivores in Mudumalai Tiger Reserve (2010 and 2011).

(a)

		Deciduous#				Dry thorn*				Semi-evergreen^	
		Dry		Wet		Dry		Wet		Dry	
Species	Year	Photocaptures	RAI	Photocaptures	RAI	Photocaptures	RAI	Photocaptures	RAI	Photocaptures	RAI
Jungle cat	2010	1	0.13			26	3.33	11	1.41	-	
	2011	-	-	2	0.26	14	1.79	17	2.18	1	0.16
	Average						2.56		1.79		
Leopard cat	2010	1	0.13	1	0.13	-	-	-	-	-	
	2011	-	-	3	0.40	-	-	-	-	1	0.16
	Average				0.26						
Rusty-spotted cat	2010	1	0.13	1	0.13	1	0.13		-	-	-
	2011	-	-	1	0.13	4	0.51	3	0.38	-	-
	Average				0.13		0.32		-		-
Small Indian civet	2010	7	0.93	5	0.67	9	1.15	6	0.77	4	0.63
	2011	16	2.13	7	0.93	3	0.38	17	2.18	15	2.38
	Average		1.53				0.76		1.47		1.5
Common palm civet	2010	2	0.40	1	0.13	9	1.15	4	0.51	-	-

	2011	4	0.53	6	0.80	5	0.64	6	0.77	-	-
	Average		0.93		0.46		0.89		1.15	-	-
Brown palm civet	2010	1	0.13	-	-	-	-	-	-	7	1.11
	2011	-	-	-	-	-	-	-	-	12	1.90
	Average										1.42
Stripe-necked mongoose	2010	14	1.87	26	3.46	1	0.13	-	-	1	0.16
	2011	10	1.33	6	0.80	-	-	-	-	9	1.43
	Average		1.53		2.13				-		0.79
Ruddy mongoose	2010	14	1.87	5	0.67	23	2.95	10	1.28	-	-
	2011	4	0.53	12	1.60	17	2.18	11	1.41	-	-
	Average		1.2		1.13		2.56		1.34	-	-
Grey mongoose	2010	1	0.13	1	0.13	17	2.18	8	1.026	-	-
	2011	2	0.26	1	0.13	15	1.92	6	0.77	-	-
	Average		0.26		0.13		2.05		0.9		

$^{\#}$ = 750 trap-nights, * = 780 trap-nights, $^{\wedge}$ = 630 trap-nights, - = no photocaptures

(b) Overall photocaptures and relative abundance index (captures/100 trap nights) of small carnivores in Mudumalai Tiger Reserve (2010 and 2011).

		Dry[$]		Wet[*]	
Species	Year	Photocaptures	RAI	Photocaptures	RAI
Jungle cat	2010	27	1.25	11	0.72
	2011	15	0.69	19	1.24
	Average	21	0.97	15	0.98
Leopard cat	2010	1	0.05	1	0.06
	2011	1	0.05	3	0.19
	Average	1	0.05	2	0.52
Rusty spotted cat	2010	2	0.092	1	0.065
	2011	4	0.18	4	0.26
	Average	3	0.136	2.5	0.16
Small Indian civet	2010	20	0.92	11	0.72
	2011	34	1.57	24	1.57
	Average	27	1.24	17.5	1.14
Common palm civet	2010	11	0.51	5	0.33
	2011	9	0.42	12	0.78
	Average	10	0.46	8.5	0.55
Brown palm civet	2010	7	0.32	-	-
	2011	12	0.55	-	-
	Average	9.5	0.43	-	-
Stripe-necked mongoose	2010	16	0.74	26	1.69
	2011	19	0.88	6	0.39
	Average	17.5	0.81	16	1.04
Ruddy mongoose	2010	37	1.71	15	0.98
	2011	21	0.97	23	1.503
	Average	29	1.34	19	1.24
Grey mongoose	2010	18	0.83	9	0.59
	2011	17	0.79	7	0.46
	Average	17.5	0.81	8	0.52

[$] = 2160 trap-nights, * = 1530 trap-nights, - = no photocaptures

3.3.4 *Detection probability*

Estimates of detection probability varied from 0.06 to 0.53, with standard errors from 0.04 to 0.16. Detection probability varied between species and seasons while for some it was constant (Table 2-5). For some species with very low detections (leopard cat and rusty spotted cat), abundance estimates could not be calculated for certain surveys due to low captures.

Wet season: Detection probability of small Indian civet, common palm civet did not differ between dry thorn and deciduous forest (t test, t = 3.13, df = 1, P = 0.19; t = -2.24, df = 1, P = 0.27, Table 2). Although detection probability of ruddy mongoose was higher in deciduous than dry thorn forest (t test, t = 1.96, df = 1, P = 0.30), it was not found significant (Table 3).

Dry season: Detection probability of small Indian civet was higher in dry thorn than deciduous forest (t test, t = -3.06, df = 1, P = 0.04) and higher in semi-evergreen than deciduous (t test, t = -6.70, df = 1, P = 0.05) however it was significantly higher in semi-evergreen than dry thorn forest (t test, t = -4.69, df = 1, 0.04) (Table 4).

Stripe-necked mongoose detection probability did not differ in wet and dry seasons (t test, t = 1.2, df = 1, P = 0.44) in deciduous forest (Table 4). Common palm civet detection probability was found higher in the deciduous than dry thorn forest (t test, t = -5.92, df = 1, P = 0.05).

In the dry season overall detection probability of small carnivores ranged from 0.05 to 0.14 with standard errors from 0.04 to 0.05 while in the wet season it ranged from 0.06 to 0.31 with standard errors from 0.04 to 0.15.

3.3.5 *Occupancy and average abundance*

Habitat-wise estimates of occupancy varied from 0.04 to 0.65, with standard errors from 0.04 to 0.3 (Tables 2-5). Average abundance ranged from 0.04 to 1.13 with standard errors from 0.04 to 0.7 within habitats (Tables 2-5). Modeled site occupancies were observed greater than naïve occupancy.

Wet season: Ruddy mongoose site occupancy was significantly higher than common palm civet in the dry thorn forest (t test, t = 5.11, df = 1, P = 0.03).

Dry season: Jungle cat occupancy was significantly higher than common palm civet in the dry thorn forest (t test, t = 15.61, df = 1, P = 0.04). Ruddy mongoose and grey mongoose occupancy was similar to common palm civet occupancy in the dry thorn forest (t test, t = 1.91, df =1, P = 0.31 and t = 2.67, df = 1, P = 0.23).

Between seasons: Jungle cat occupancy was significantly higher in the wet season than dry season (t test, t = 27.01, df =1, P =< 0.001) in the thorn forest. Although average abundance of stripe-necked mongoose was found higher in the dry season than wet season in deciduous forest (t test, t = -4.9, df = 1, P = 0.06), it was not significant. Occupancy of stripe-necked mongoose was higher in dry season than wet season (t test, t = -5.73, df = 1, P = 0.01) in the deciduous forest.

Table. 2. Detection probability (r_{mean}), average abundance/km^2 (λ_{mean}), site occupancy (Ψ_{mean}) and total abundance (N_{mean}) with associated mean standard errors (± SE) for small carnivores based on camera trap data in the wet season using the Royle-Nichols Heterogeneity model.

Species	Habitat	r_{mean} ± SE	λ_{mean} ± SE	Naïve occupancy	Ψ_{mean} ± SE	N_{mean}±SE	95% Confidence interval$_{\mathrm{mean}}$
Leopard cat	DEC(2011)	0.088 ± 0.085	0.14 ± 0.13	0.12	0.13 ± 0.11	3.41 ± 3.29	3.04-9.86
Small Indian civet	DEC(2010)	0.112 ± 0.084	0.20 ± 0.19	0.16	0.18 ± 0.15	5.08 ± 4.65	4.03-14.21
	DEC(2011)	0.094 ± 0.073	0.28 ± 0.22	0.16	0.24 ± 0.17	6.99 ± 5.58	3.94-17.94
	Average	0.103 ± 0.079	0.24 ± 0.20	0.16	0.21 ± 0.16	6.04 ± 5.12	3.98-16.07
Common palm civet	DEC(2010)	0.106 ± 0.099	0.13 ± 0.12	0.08	0.12 ± 0.08	3.18 ± 3.09	2.86-9.25
	DEC(2011)	0.097 ± 0.093	0.14 ±0.13	0.08	0.13 ± 0.11	3.41 ± 3.29	3.04-9.87
	Average	0.101 ± 0.096	0.13 ± 0.13	0.08	0.12 ± 0.09	3.29 ± 3.19	2.95-0.31
Jungle cat	DT(2010)	0.24 ± 0.098	0.13 ± 0.08	0.11	0.12 ± 0.07	3.48 ± 2.13	0.69-7.65
	DT(2011)	0.097 ± 0.053	0.67 ± 0.38	0.35	0.49 ± 0.19	17.53 ± 9.93	1.94-36.99
	Average	0.17 ± 0.076	0.40 ± 0.23	0.23	0.31 ± 0.13	10.50 ± 6.03	1.32-22.32
Rusty-spotted cat	DT(2011)	0.089 ± 0.086	0.13 ± 0.13	0.11	0.12 ± 0.11	3.43 ± 3.30	3.051-9.90
Small Indian civet	DT(2010)	0.074 ± 0.069	0.26 ±0.25	0.12	0.22 ± 0.20	6.86 ± 6.40	6.04-19.84
	DT(2011)	0.065 ± 0.048	0.89 ± 0.69	0.35	0.58 ± 0.268	23.28 ± 17.91	11.82-58.38
	Average	0.07 ± 0.059	0.57 ± 0.48	0.23	0.40 ± 0.23	15.06 ± 12.15	9.432-39.11
Common palm civet	DT(2010)	0.28 ± 0.16	0.043 ± 0.043	0.04	0.041 ± 0.041	1.08 ± 1.10	1.08-3.24
	DT(2011)	0.17 ± 0.11	0.103 ± 0.085	0.076	0.096 ± 0.077	2.62 ± 2.22	1.74-6.98
	Average	0.22 ± 0.135	0.073 ± 0.064	0.058	0.068 ± 0.059	1.85 ± 1.66	1.41-5.11

DEC = Deciduous, DT = Dry thorn, SEV = Semi-evergreen

Table. 3. Detection probability (c_{mean}), average abundance/km^2 (λ_{mean}), site occupancy (Ψ_{mean}) and total abundance (N_{mean}) with associated mean standard errors (± SE) for small carnivores based on camera trap data in the wet season using the Repeated count model.

Species	Habitat	c_{mean} ± SE	λ_{mean} ± SE	Naïve occupancy	Ψ_{mean} ± SE	N_{mean} ±SE	95% Confidence interval$_{\text{mean}}$
Stripe-necked mongoose	DEC(2010)	0.53 ± 0.08	0.197 ± 0.089	0.12	0.18 ± 0.07	5.00 ± 2.25	0.58-9.42
	DEC(2011)	0.15 ± 0.093	0.16 ± 0.11	0.08	0.147 ± 0.098	4.02 ± 2.96	2.02-9.80
	Average	0.34 ± 0.09	0.18 ± 0.10	0.10	0.164 ± 0.086	4.51 ± 2.61	0.72-9.61
Ruddy mongoose	DEC(2010)	0.19 ± 0.06	0.40 ± 0.16	0.16	0.33 ± 0.11	9.99 ± 3.98	2.17-17.79
	DEC(2011)	0.40 ± 0.10	0.12 ± 0.07	0.08	0.112 ± 0.062	2.97 ± 1.74	0.44-6.39
	Average	0.29 ± 0.08	0.26 ± 0.11	0.12	0.22 ± 0.08	6.48 ± 2.86	0.87-5.70
Ruddy mongoose	DT(2010)	0.079 ± 0.053	0.54 ± 0.40	0.15	0.41 ± 0.22	14.04 ± 10.39	6.34-34.42
	DT(2011)	0.12 ± 0.06	0.37 ± 0.22	0.23	0.31 ± 0.15	9.64 ± 5.66	1.44-20.73
	Average	0.09 ± 0.06	0.46 ± 0.31	0.20	0.359 ± 0.18	11.84 ± 8.03	3.89-27.58
Grey mongoose	DT(2010)	0.39 ± 0.12	0.08 ± 0.06	0.04	0.076 ± 0.051	2.04 ± 1.45	0.79-4.88
	DT(2011)	0.096 ± 0.074	0.26 ± 0.20	0.20	0.23 ± 0.15	6.74 ± 5.29	3.63-17.11
	Average	0.24 ± 0.097	0.17 ± 0.13	0.12	0.151 ± 0.103	4.39 ± 3.37	2.21-10.99

DEC = Deciduous, DT = Dry thorn, SEV = Semi-evergreen

Table. 4. Detection probability (r_{mean}), average abundance/km^2 (λ_{mean}), site occupancy (Ψ_{mean}) and total abundance (N_{mean}) with associated mean standard errors (± SE) for small carnivores based on camera trap data in the dry season using the Royle-Nichols Heterogeneity model.

Species	Habitat	r_{mean} ± SE	λ_{mean} ± SE	Naïve occupancy	Ψ_{mean} ± SE	N_{mean} ±SE	95% Confidence interval_{mean}
Small Indian civet	DEC(2010)	0.074 ± 0.062	0.41 ± 0.35	0.16	0.33 ± 0.22	10.2 ± 8.7	6.81-27.21
	DEC(2011)	0.089 ± 0.064	0.39 ± 0.31	0.30	0.32 ± 0.201	9.79 ± 7.73	5.36-24.94
	Average	0.082 ± 0.063	0.401 ± 0.33	0.23	0.33 ± 0.21	9.99 ± 8.20	6.087-26.073
Common palm civet	DEC(2011)	0.15 ± 0.10	0.113 ± 0.09	0.08	0.105 ± 0.083	2.76 ± 2.33	1.79-7.33
Jungle cat	DT(2010)	0.164 ± 0.061	0.472 ± 0.205	0.31	0.37 ± 0.13	12.27 ± 5.35	1.79-22.75
	DT(2011)	0.114 ± 0.056	0.47 ± 0.24	0.27	0.38 ± 0.15	12.36 ± 6.28	0.042-24.67
	Average	0.139 ± 0.058	0.475 ± 0.22	0.29	0.37 ± 0.14	12.31 ± 5.81	0.92-23.71
Rusty-spotted cat	DT(2011)	0.142 ± 0.098	0.105 ± 0.092	0.08	0.102 ± 0.081	2.79 ± 2.35	1.83-7.41
Small Indian civet	DT(2010)	0.123 ± 0.07	0.30 ± 0.189	0.20	0.26 ± 0.14	7.81 ± 4.91	1.81-17.42
	DT(2011)	0.108 ± 0.101	0.12 ± 0.12	0.08	0.113 ± 0.102	3.13 ± 3.02	2.79-9.05
	Average	0.115 ± 0.085	0.21 ± 0.154	0.14	0.185 ± 0.12	5.47 ± 3.96	2.30-13.24
Common palm civet	DT(2010)	0.25 ± 0.097	0.129 ± 0.079	0.08	0.121 ± 0.069	3.35 ± 2.03	0.63-7.33

	DT(2011)	0.13 ± 0.086	0.159 ± 0.11	0.11	0.149 ± 0.098	4.19 ± 2.99	1.67-10.06
	Average	0.19 ± 0.092	0.144 ± 0.096	0.095	0.135 ±0.083	3.77 ± 2.51	1.15-8.69
Small Indian civet	SEV(2010)	0.15 ± 0.103	0.14 ± 0.11	0.10	0.13 ± 0.10	2.83 ± 2.42	1.93-7.59
	SEV(2011)	0.16 ± 0.073	0.38 ± 0.19	0.24	0.31 ±0.13	7.95 ± 4.03	0.05-15.85
	Average	0.15 ± 0.088	0.26 ± 0.15	0.17	0.22 ± 0.11	5.39 ± 3.22	0.94-11.72

DEC = Deciduous, DT = Dry thorn, SEV = Semi-evergreen

Table. 5. Detection probability (c_{mean}), average abundance/km^2 (λ_{mean}), site occupancy (Ψ_{mean}) and total abundance (N_{mean}) with associated mean standard errors (± SE) for small carnivores based on camera trap data in the dry season using the Royle-Nichols Repeated count model.

Species	Habitat	c_{mean} ± SE	λ_{mean} ± SE	Naïve occupancy	Ψ_{mean} ± SE	N_{mean} ± SE	95% Confidence interval$_{\text{mean}}$
Brown palm civet	SEV(2010)	0.084 ± 0.061	0.45 ± 0.32	0.24	0.36 ± 0.204	9.47 ± 6.74	3.74-22.68
	SEV(2011)	0.0796 ± 0.045	0.78 ± 0.43	0.33	0.54 ± 0.19	16.33 ± 9.05	1.40-34.06
	Average	0.105 ± 0.068	0.44 ± 0.31	0.24	0.35 ± 0.19	9.31 ± 6.55	3.53-22.13
Stripe-necked mongoose	DEC(2010)	0.14 ± 0.062	0.39 ± 0.19	0.28	0.33 ± 0.13	9.99 ± 4.87	0.46-19.53
	DEC(2011)	0.077 ± 0.059	0.49 ± 0.37	0.20	0.39 ± 0.23	12.25 ± 9.36	6.10-30.59
	Average	0.11 ± 0.061	0.44 ± 0.28	0.24	0.36 ± 0.18	11.12 ± 7.11	2.82-25.06
Ruddy mongoose	DEC(2010)	0.19 ± 0.070	0.29 ± 0.13	0.20	0.26 ± 0.10	7.44 ± 3.39	0.79-14.08

	DEC(2011)	0.15 ± 0.103	0.11 ± 0.092	0.08	0.105 ± 0.08	2.77 ± 2.33	1.804-7.34
	Average	0.17 ± 0.087	0.204 ± 0.11	0.14	0.181 ± 0.092	5.10 ± 2.86	0.51-10.71
Grey mongoose	DEC(2010)	0.16 ± 0.062	0.404 ± 0.17	0.04	0.33 ± 0.12	10.55 ± 4.56	1.604-19.51
Ruddy mongoose	DT(2010)	0.27 ± 0.064	0.32 ± 0.12	0.23	0.277 ± 0.087	8.43 ± 3.14	2.27-14.58
	DT(2011)	0.064 ± 0.046	1.135 ± 0.901	0.40	0.66 ± 0.26	29.47 ± 23.45	16.49-75.43
	Average	0.17 ± 0.055	0.73 ± 0.51	0.31	0.47 ± 0.17	18.95 ± 13.29	7.11-45.01
Grey mongoose	DT(2010)	0.16 ± 0.062	0.404 ± 0.17	0.19	0.33 ± 0.12	10.55 ± 4.56	1.60-19.51
	DT(2011)	0.071 ± 0.044	0.83 ± 0.51	0.30	0.56 ± 0.22	21.63 ± 13.15	4.14-47.40
	Average	0.12 ± 0.053	0.62 ± 0.34	0.24	0.45 ± 0.17	16.094 ± 8.857	1.27-33.45
Stripe-necked mongoose	SEV(2011)	0.091± 0.066	0.48 ± 0.35	0.33	0.38 ± 0.21	10.17 ± 7.48	4.49-24.83

DEC = Deciduous, DT = Dry thorn, SEV = Semi-evergreen

Table. 6. Detection probability (c_{mean} & r_{mean}), average abundance/km^2 (λ_{mean}), site occupancy (Ψ_{mean}) and total abundance (N_{mean}) with associated mean standard errors (± SE) for small carnivores based on camera trap data across habitats in the dry season using Royle-Nichol models.

Species	Survey	Detection probability ± SE	λ_{mean} ± SE	Naïve occupancy	Ψ_{mean} ± SE	N_{mean} ± SE	95% Confidence interval$_{\text{mean}}$
Jungle cat*	2010	0.15 ± 0.048	0.17 ± 0.06	0.12	0.16 ± 0.053	12.41 ± 4.55	3.50-21.31
	2011	0.10 ± 0.045	0.19 ± 0.08	0.11	0.17 ± 0.069	13.89 ± 6.07	2.00-25.78
	Average	0.13 ± 0.046	0.18 ± 0.07	0.11	0.164 ± 0.061	13.15 ± 5.31	2.75-23.54
Rusty-spotted cat*	2011	0.048 ± 0.046	0.15 ± 0.15	0.056	0.142 ± 0.13	11.09 ± 11.01	10.74-32.75

Small Indian civet*	2010	0.083 ± 0.034	0.37 ± 0.15	0.19	0.31 ± 0.105	26.79 ± 10.96	5.31-48.27
	2011	0.11 ± 0.035	0.38 ± 0.12	0.24	0.32 ± 0.083	27.56 ± 8.82	10.27-44.84
	Average	0.096 ± 0.034	0.37 ± 0.13	0.21	0.31 ± 0.094	27.17 ± 9.89	7.79-46.55
Common palm civet*	2010	0.16 ± 0.06	0.11 ± 0.05	0.083	0.101 ± 0.042	7.65 ± 3.40	0.98-14.32
	2011	0.05 ± 0.039	0.26 ± 0.20	0.097	0.23 ± 0.15	18.83 ± 14.52	9.63-47.29
	Average	0.105 ± 0.05	0.185 ± 0.12	0.09	0.16 ± 0.19	13.24 ± 8.96	5.30-30.80
Brown palm civet#	2010	0.086 ± 0.049	0.15 ± 0.08	0.083	0.13 ± 0.072	10.49 ± 6.01	1.30-22.28
	2011	0.101 ± 0.045	0.18 ± 0.08	0.097	0.16 ± 0.068	12.80 ± 5.87	1.30-24.30
	Average	0.093 ± 0.047	0.165 ± 0.08	0.09	0.145 ± 0.07	11.64 ± 5.94	1.3-23.29
Stripe-necked mongoose#	2010	0.11 ± 0.042	0.22 ± 0.09	0.12	0.198 ± 0.071	15.85 ± 6.41	3.29-28.41
	2011	0.068 ± 0.034	0.39 ± 0.19	0.17	0.32 ± 0.129	27.97 ± 13.67	1.17-54.77
	Average	0.087 ± 0.038	0.30 ± 0.14	0.15	0.26 ± 0.1	21.91 ± 10.04	2.23–41.59
Ruddy mongoose#	2010	0.21 ± 0.041	0.25 ± 0.07	0.14	0.22 ± 0.051	17.70 ± 4.74	8.42-26.99
	2011	0.05 ± 0.028	0.61 ± 0.33	0.19	0.46 ± 0.18	44.24 ± 24.03	2.86-91.34
	Average	0.13 ± 0.03	0.43 ± 0.2	0.17	0.34 ± 0.115	30.97 ± 14.38	5.64-59.16
Grey mongoose#	2010	0.19 ± 0.053	0.13 ± 0.05	0.083	0.12 ± 0.043	9.42 ± 3.49	2.57-16.26
	2011	0.092 ± 0.037	0.26 ± 0.11	0.11	0.23 ± 0.082	18.53 ± 7.61	3.61-33.44
	Average	0.14 ± 0.045	0.19 ± 0.08	0.097	0.17 ± 0.062	13.97 ± 5.55	3.09-24.85

* = Royle-Nichol's Heterogeniety model, # = Repeated count model

Table.7. Detection probability (c_{mean} & r_{mean}), average abundance/km^2 (λ_{mean}), site occupancy (Ψ_{mean}) and total abundance (N_{mean}) with associated mean standard errors (± SE) for small carnivores based on camera trap data across habitats in the wet season using Royle-Nichol models.

Species	Habitat	Detection probability ± SE	λ_{mean} ± SE	Naïve occupancy	Ψ_{mean} ± SE	N_{mean} ± SE	95% Confidence interval$_{mean}$
Jungle cat*	2010	0.24 ± 0.088	0.07 ± 0.04	0.058	0.063 ± 0.036	3.34 ± 1.96	0.5-7.18
	2011	0.088 ± 0.039	0.41 ± 0.19	0.21	0.335 ± 0.123	20.85 ± 9.47	2.29-39.41
	Average	0.164 ± 0.063	0.24 ± 0.11	0.13	0.19 ± 0.079	12.09 ± 5.71	1.39–23.29
Leopard cat*	2011	0.091 ± 0.084	0.06 ± 0.06	0.039	0.063 ± 0.056	3.30 ± 3.06	2.69-9.29
Rusty-spotted cat*	2011	0.061 ± 0.061	0.13 ± 0.13	0.058	0.12 ± 0.11	6.62 ± 6.42	5.97-19.21
Small Indian civet*	2010	0.081 ± 0.049	0.22 ± 0.13	0.12	0.20 ± 0.11	11.19 ± 6.80	2.14-24.53
	2011	0.068 ± 0.035	0.59 ± 0.30	0.25	0.45 ± 0.16	30.12 ± 15.22	0.29-59.95
	Average	0.074 ± 0.042	0.40 ± 0.21	0.18	0.32 ± 0.13	20.65 ± 11.01	1.21-42.24
Common palm civet*	2010	0.061 ± 0.061	0.13 ± 0.13	0.058	0.12 ± 0.11	6.62 ± 6.42	5.97 - 19.21
	2011	0.1605 ± 0.071	0.10 ± 0.05	0.078	0.094 ± 0.048	5.01 ± 2.71	0.29 - 10.32
	Average	0.11 ± 0.066	0.11 ± 0.09	0.068	0.11 ± 0.079	5.81 ± 4.56	3.13 – 14.76
Stripe-necked mongoose#	2010	0.52 ± 0.074	0.10 ± 0.04	0.039	0.094 ± 0.04	5.04 ± 2.26	0.62-9.46
	2011	0.11 ± 0.075	0.09 ± 0.06	0.039	0.082 ± 0.054	4.36 ± 3.02	1.56-10.29
	Average	0.31 ± 0.15	0.09 ± 0.05	0.039	0.088 ± 0.045	4.7 ± 2.64	1.09–9.87

Ruddy mongoose#	2010	0.056 ± 0.034	0.52 ± 0.31	0.176	0.41 ± 0.18	26.76 ± 15.56	3.73-57.25
	2011	0.22 ± 0.054	0.21 ± 0.07	0.157	0.18 ± 0.059	10.64 ± 3.71	3.37-17.91
	Average	0.14 ± 0.044	0.36 ± 0.19	0.167	0.29 ± 0.12	18.7 ± 9.63	3.55–37.58
Grey mongoose#	2010	0.19 ± 0.073	0.09 ± 0.05	0.078	0.086 ± 0.043	4.62 ± 2.40	0.08 - 9.31
	2011	0.055 ± 0.0404	0.39 ± 0.27	0.157	0.32 ± 0.18	19.82 ± 13.92	7.46 - 47.10
	Average	0.12 ± 0.056	0.24 ± 0.16	0.12	0.20 ± 0.11	12.22 ± 8.16	3.77–28.20

* = Royle-Nichol’s Heterogeniety model, # = Repeated count model

3.3.6 *Jungle cat capture success*

The sampling effort of 690 trap-nights for 2010 and 2011 yielded 66 pictures of jungle cat (16 right flanked and 19 left flanked in 2010 and 13 right flanked and 7 left flanked in 2011) from which I obtained encounter histories for 6 individuals from left flank in 2010 and 7 from right flank in 2011. Jungle cats were detected at 39% of camera stations in 2010 and 43% in 2011. Few individuals were recaptured multiple times at multiple camera stations within a survey day. Although no cubs or sub-adults were camera-trapped, I had a direct observation of a mother with two kittens in the field. The closure test conducted in CAPTURE was consistent with the assumption that the jungle cat population for 2010 ($z = -0.382$, $P = 0.35$) and 2011 ($z = -1.217$, $P = 0.11$) was closed.

3.3.7 *Estimates of population sizes, densities and sex ratio for jungle cat*

The mean maximum distances moved (MMDM) by individual jungle cats detected at multiple sampling stations in 2010 (n = 2) was 2.87 km and 3.23 km (n = 2) in 2011. Effectively sampled areas from traditional approaches under different buffer ranged from 35.5 to 150 km^2 (Table 8).

In 2010 the capture probability (*p-hat*) of jungle cat was 0.22 for left flank; population size *(N)* under model M_h was 6.6 (SE = ± 2.5) and in 2011 capture probability was 0.14 for right flank; population size *(N)* was 9.3 (SE = ± 3.3) in the sampled area. Spatially explicit maximum likelihood density estimate for 2010 was 0.08 (SE = ± 0.04) individuals/km^2 and 0.09 (SE = ± 0.05) individuals/km^2 in 2011. Jungle cat density from MMDM models ranged from 0.04 (SE = ± 0.02) to 0.11(± 0.037) individuals/km^2 depending on the buffer applied and the effective sample area (Table 8). Abundance estimate from Royle and Nichol's Heterogeneity model (hereafter called RN Heterogeneity, on an average 0.47 (± 0.22) individuals/km^2) was higher than other estimates (Table 8).

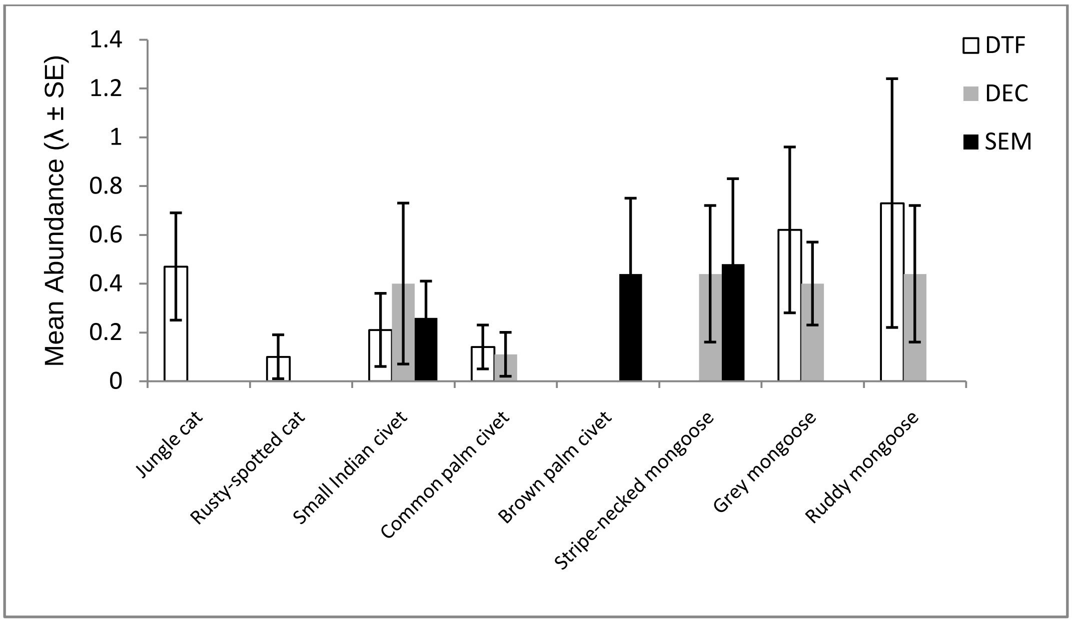
The sex ratio male (M):female (F) estimated at the reserve was 1:1.5 in 2010 and 1:1.2 in 2011. At a camera station where more than one individual was detected the sex ratio was 1M:1F, 1M:2F and in 2010 and 1M:1F in 2011.

Table 8. Jungle cat abundance estimates and statistical parameters in Mudumalai Tiger Reserve.

Survey	*N*(± SE)(95% CI)[a]	*p-hat*[b]	*d-bar*(± SE)(km)[c]	g0(± SE)[d]	MMDM (km)[e]	Method	ETA(km^2)[f]	Boundary strip width (km)	Density(± SE) individuals per km^2
2010	6.6(± 2.5) (6-22.7)	0.23	1.68 (± 0.57)		2.87(± 1.45)	HR radius[g]	62.9	0.81	0.105(± 0.04)
						MMDM/2[h]	87.27	1.52	0.076(± 0.03)
						MMDM	150.05	3.05	0.044(± 0.02)
				0.053 (± 0.02)		ML-SECR[i]	74.61	1.36	0.081(± 0.04)
	12.27(± 5.35) (1.79-22.75)			0.16 (± 0.06)	-	RN Heterogeneity[j]	-	-	0.47 (± 0.205)
2011	9.3(± 3.3) (7.3 - 25.4)	0.14	2.202(± 1.01)		3.23(± 1.62)	HR radius	62.91	0.81	0.15(± 0.05)
						MMDM/2	87.27	1.52	0.11(± 0.04)
						MMDM	150.05	3.05	0.062(± 0.02)
						ML-SECR	74.61	1.36	0.097(± 0.05)
	12.36(± 6.28) (0.042-24.67)			0.11 (± 0.06)	-	RN Heterogeneity	-	-	0.47 (± 0.24)

[a] = Population size, [b] = capture probability, [c] = Mean recapture distance, [d] = Detection probability, [e] = Mean maximum distance moved, [f] = Effective trapping area, [g] = Home range radius [h] = Half mean maximum distance moved, [i] = Maximum likelihood, [j] = Royle and Nichol's heterogeneity model

Fig 7. Abundance (individuals/km^2) from occupancy models for small carnivores in dry thorn (DT), deciduous (DEC) and semi-evergreen forest (SEM) in Mudumalai. Estimates were taken only for the dry season for comparison across species.



3.3.8 *Correlates of small carnivore body weights*

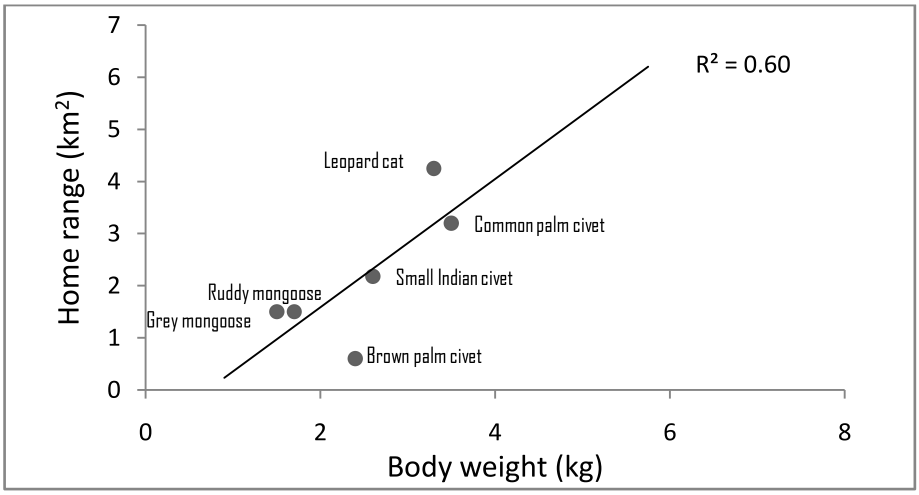
Data showed that body weight had a significant positive correlation with home range (km^2) (Pearson correlation; $R^2 = 0.75$, $n = 6$, $P = 0.04$, Fig 8a) while there was no correlation with detection probability (Pearson correlation; $R^2 = 0.32$, $n = 8$, $P > 0.05$, Fig 8b), site occupancy (Pearson correlation; $R^2 = -0.13$, $n = 8$, $P > 0.05$, Fig 8c) and RAI (Pearson correlation; $R^2 = 0.21$, $n = 8$, $P > 0.05$, Fig 8d). There was a significant positive relationship between mean abundance (λ) and RAI (Pearson correlation; $R^2 = 0.83$, $n = 8$, $P = 0.005$, Fig 8e).

Table 9. Body weight (kg) and home ranges (km^2) of small carnivores from literature.

	Weight (kg)	**Home range (km^2, unless otherwise stated)**
Jungle cat	4 - 5.75 (Mukherjee and Groves 2007)	-
	4-6 (Lekagul and McNeeley 1977)	
Rusty-spotted cat	0.9 (N = 1, Sunquist and Sunquist 2002)	-
Leopard cat	2.7 ± 0.44 (X ± SD, Grassman et al., 2005)	12.4 - 14 (Grassman et al., 2005)
	2.3 - 4.2 (Howard and Wildt 1990)	2.5 - 4.1 (Grassman 2000)
	3-5 (Lekagul and McNeeley 1977, Rabinowitz 1990)	3.56 - 6.6 (Rabinowitz 1990)
Small Indian civet	3 - 4 (Prater 1971, Corbet and Hill 1992,)	3.1 (N = 1, Rabinowitz 1991)
	2 (female road-kill specimen from present study)	
	2.5 (Kumar and Umapathy 1999)	2.17 (N = 1, Kumar and Umapathy 1999)
Common palm civet	3.5 (N =1, female live trapped in the present study)	
	2 - 5 (Lekagul and McNeeley 1977)	14.1 ± 2.3 ha (Joshi et al., 1995)
	2.4 (N = 4, Sum 2011)	25 - 103 ha (Nakashima and Jumrafiah 2010)
	2.7 - 4.5 (Prater 1971, Dhungel 1985, Corbet and Hill 1992)	3.2 (N =1, Rabinowitz 1991)
Brown palm civet	2.4 ± 0.8 (X ± SD, N = 7, Mudappa and Chellam 2001)	0.35 - 29.58 ha(N = 7, Mudappa 2001)
Grey mongoose	1 - 2 (N = 1, live-trapped individual in the present study)	
	1.4 – 2 (Corbet and Hill 1992, Prater 1971, Kumar and Umapathy 1999)	15.5 ha (N=1, Kumar and Umapathy 1999)
Ruddy mongoose	1.2 - 1.5 (female road-kill specimen in the present study)	-
	c. 1.5 (Corbet and Hill 1992, Prater 1971)	
Stripe-necked mongoose	3.2 (Corbet and Hill 1992, Prater 1971)	-

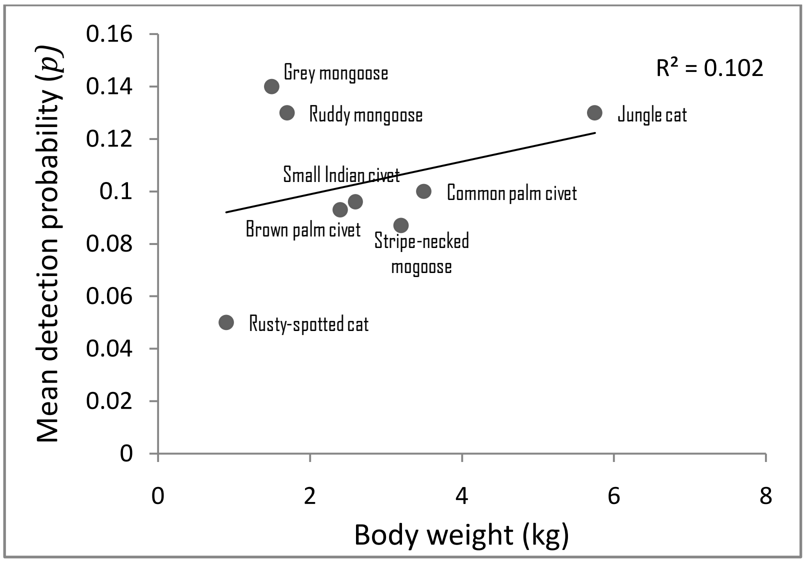
- = Data unavailable

Fig 8. a) The relationship between body weight (kg) and home range size (km^2) of small carnivores in Mudumalai. Home range and body weight estimates were taken from available literature. Jungle cat and stripe-necked mongoose was excluded since their home range estimates are unavailable.



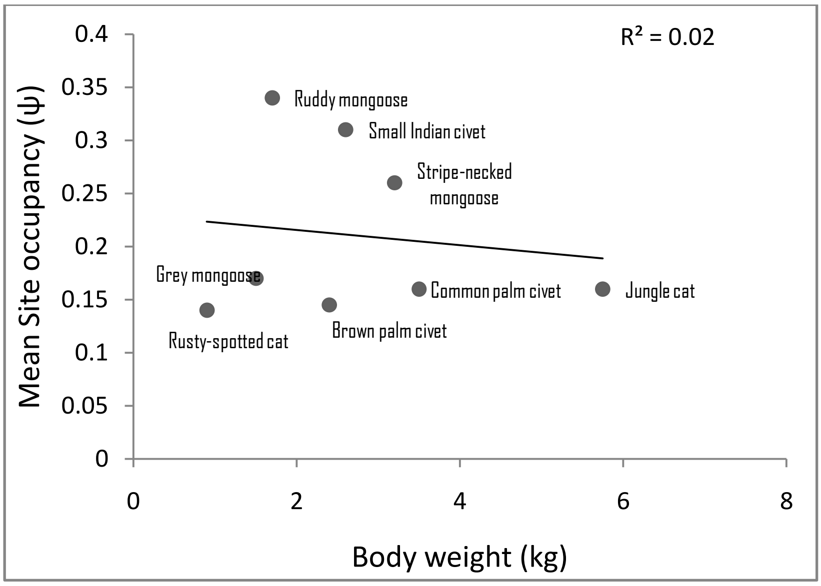
Body weight (kg) and home range size (km^2) obtained from Table 9.

b) Relationship between body weight (kg) and mean detection probability for small carnivores in Mudumalai.



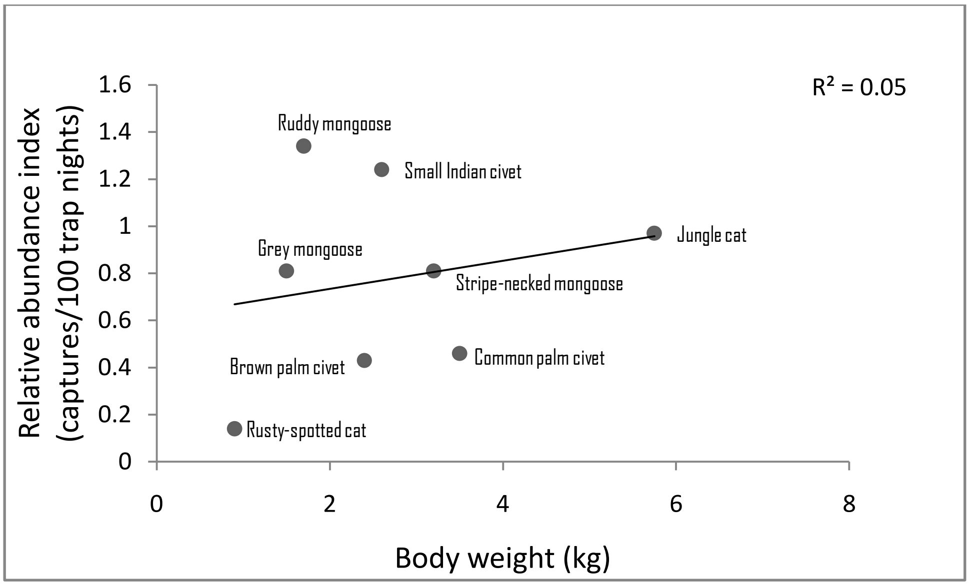
Body weight (kg) obtained from Table 9.

c) Relationship between body weight (kg) and mean site occupancy (as obtained from Table 6) for small carnivores in Mudumalai.



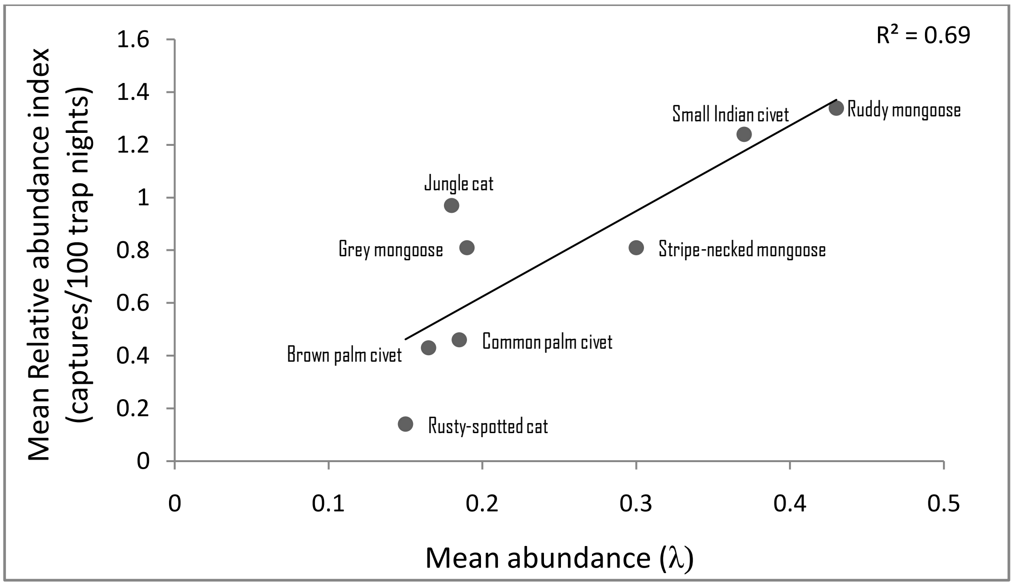
Body weight (kg) obtained from Table 9.

d) Relationship between body weight (kg) and relative abundance index (captures/100 trap nights) for small carnivores in Mudumalai.



Body weight (kg) obtained from Table 9.

e) Relationship between mean abundance (λ) and relative abundance index (captures/100 trap nights) for small carnivores in Mudumalai.



3.4 Discussion

3.4.1 *Occupancy and habitat preference*

I provided estimates of detection probability, site occupancy and relative abundance for rare and cryptic carnivores; small felids, viverrids and herpestids in Mudumalai. There was considerable interspecific variability in species detectability. Occupancy estimates for the most frequently occurring species were statistically robust and useful as a population index over time, although results should be interpreted cautiously. Confidence intervals for most species were high indicating small sample size of captures. Home ranges of study species were unknown for the area, and for some species like the lesser cats, it could exceed the subunit area sampled such that occupancy estimates represent an area that is used by the species (MacKenzie et al., 2006). Occupancy and abundance estimates are comparable to a few other studies (Nag 2008, Johnson et al., 2009, Gupta 2011). This study suggests the potential affinity of study species towards a particular forest type. Brown palm civet was frequently trapped in the semi-evergreen forest, while common palm civet, jungle cat, grey mongoose and rusty spotted cat in the dry thorn and deciduous forest and stripe-necked mongoose in deciduous and semi-evergreen forest. This also indicates that brown palm civet, jungle cat and common palm civet are habitat specialists.

The lack of records for some species in Mudumalai like the leopard cat was probably due to its rarity and elusiveness in the study area as compared to the relatively wet forests of the North-east India where they are known to be more common (Datta 2008). RAI of small Indian civet (0.43 in Pakke Tiger Reserve) and common palm civet (0.43 in Pakke Tiger Reserve and 0.26 in Namdapha) in North-east India was lower than estimates from the present study. The large Indian civet (RAI = 1.66) and the common palm civet (RAI = 0.85) were frequently encountered in Cambodia as well (Gray and Phan 2011). Camera-trapping in the Sarawak, Malaysia revealed regular usage of *Acacia mangium* plantation by the common palm civet (Belden et al., 2007) and interestingly it was not photo-trapped in the semi-evergreen forest in the present study. Even earlier studies recorded its high capture frequency only in drier deciduous forests at lower elevations (< 800 m) and never in the rainforests (Mudappa et al., 2007) which could also explain why its detection probability was higher in deciduous than dry thorn forest in the present study. Small Indian civet and *Herpestes spp*. were more frequent in rainforest fragments than in the relatively undisturbed, large, contiguous rainforest tracts of Kalakad–Mundanthurai Tiger Reserve (Mudappa et al., 2007, Ramesh et al., 2012).

Earlier studies on brown palm civets have shown this species thriving in rainforests and evergreen forests of south India (Mudappa 1998), which was evident in the present study as it was frequently photocaptured in the semi-evergreen forest. A survey by Nixon et al., (2010) throughout Kerala and Karnataka, reported the occurrence of brown palm civet only in evergreen forests, common palm civet was almost entirely recorded in deciduous forests and plantations, all images of jungle cat were obtained from deciduous forests and plantations, leopard cat from evergreen forests and small Indian civet was photo-trapped in almost all vegetation types at all survey sites. The small Indian civet seemed to be a generalist with high occupancy in Mudumalai also indicating no variation in occupancy estimates across the three habitats. Small Indian civet, palm civets, small cats, and mongooses were recorded as the most abundant species in Bhadra Wildlife Sanctuary based on track plot evidences (Bali et al., 2007). With a live trapping effort of 400 days, Kumar and Umapathy (1999) reported low success rates for the small Indian civet (0.25%) and grey mongoose (0.5%) in the Nilgiri Biosphere Reserve. Detection probability of small

Indian civet was higher in dry thorn and semi-evergreen forests than deciduous forest due to high fruit availability in these habitats which in comparison the deciduous habitats are poorly rich in fruit diversity.

Although ruddy mongoose was not camera trapped in the semi-evergreen forest in the present study, I had direct observations of this species in this habitat. The detection probability of ruddy mongoose was higher in deciduous forest than in dry thorn forest since they are usually known to prefer relatively dry deciduous forests. Mongooses in India are also primarily adapted to drier forests, as indicated by greater abundance of their scats in such forests (Kumar and Umapathy 1999). However, the endemics, brown mongoose is absent from the drier parts of the Western Ghats. It was not recorded in our study area, although it has been sighted by local people in the night around coffee/tea estates towards the western region of Mudumalai, adjoining Wayanad Wildlife Sanctuary. Stripe-necked mongoose has been reported from wet, semi-evergreen and dry deciduous forests especially near water bodies in the Nilgiri Biosphere Reserve.

Ruddy mongoose, grey mongoose and jungle cat can tolerate or even exist in high abundance near moderately disturbed areas which also explains their high abundance in the thorn forest than deciduous forest in this study. Grey mongoose RAI in dry thorn forest was higher in the dry season than wet season because they probably thrive in open habitats with low canopy cover. During the present study I also noticed grey mongoose scats more often over rocky substrates especially in open habitats. As estimated by trapping grids, population densities of small Indian mongoose *Herpestes javanicus* range from 0.5/ha on the Fijian islands (Gorman 1979) to as much as 24.7/ha in Hawaii (Seaman 1952), and tend to be denser in scrubland, grassland, mixed vegetation types, and ecotones than in forested areas (Pimentel 1955, Nellis and Everard 1983, Hoagland et al., 1989).

High abundance of jungle cat in open scrub habitat has been documented by Mukherjee (1998) and Gupta (2011). Interestingly jungle cat occupancy was found to be higher in the wet season than dry season in the thorn forest probably due to their preference towards moist areas. On the contrary, jungle cat spatial distribution in the dry season could have been clumped towards water sources resulting in its low site occupancy. The higher occupancy of stripe-necked mongoose in dry than wet season in the deciduous forest could relate to their high movement rates as a result of food scarcity since mongooses generally feed on invertebrates especially burrowing insects which are usually active in the wet season. This could prompt the animal to increase it foraging activity resulting in high occupancy.

Past studies that compared abundance through scat encounter rates across wide geographical areas in Western Gats illustrated that lesser carnivores were more abundant in Silent Valley National Park followed by New Amarambalam, and least in Mudumalai Tiger Reserve and Wayanad Wildlife Sanctuary (Kumar and Yoganand 1999). They also reported that civet scats were most abundant in moist forests of Silent Valley National Park and New Amarambalam Reserved Forests while mongoose scats were more common in the in the relatively drier Wayanad Wildlife Sanctuary, Mudumalai Tiger Reserve and Sigur Reserved Forests. The differences in detection probability and occupancy of some species between habitats and seasons might result due to differences in the major vegetation, topographical attributes and spatial distribution of food. Different species will have different microhabitat preferences; therefore there is no reason to expect different species to have similar capture probability. Small carnivores represent suitable model species to test whether species with similar/dissimilar ecological niches have similar detection probabilities within the same habitat.

In south-east Asia the lower density of civets in the logged forest indicates that selective logging may have a negative impact and estimates suggest high densities ($31.5/km^2$) for civet species in undisturbed primary forest (Heydon and Bulloh 1996). Colon (2002) found that the population density of Malay civets in a logged forest was lower than those in unlogged forest and suggested that this may be because of lower fruit availability in the logged forest.

Estimates of small carnivores from available studies were mostly photographic indices that are only an index of abundance. Much work on small carnivores have used night walks along established trails to estimate encounter rates or densities, however in many areas, these may be time consuming and labour-intensive and preclude wider spatial coverage. The three habitats considered in this study, occurred in large, continuous areas. For those species with relatively small home ranges (mongooses and palm civets) most individuals would travel within the three habitats and may not between habitat zones for daily activities. Habitat use for these species is equal to difference in abundance in the three habitat types and I would expect seasonal variation in species. The high detection probability of small Indian civet in dry thorn and semi-evergreen forest in the dry season was probably due to high availability of fruits. I observed counterintuitive cases of relatively high site occupancy but low detectability for some species like the brown palm civet, stripe necked mongoose, jungle cat and small Indian civet. In this study, occupancy rate under both the models and abundances of some species were expected to be higher than those estimated. However, low estimates with relatively high standard errors can be expected when the number of sites are extremely low – such as in this study. Nevertheless this study will serve as benchmark database against which future estimates from the same area/habitats/landscape can be compared and refined. This study shows that the models can indeed be used to estimate abundances for a set of species for which conventional methods cannot be used. To understand small carnivore community ecology and status, future monitoring should consider comparative methods to assess abundance of primarily arboreal species (e.g. palm civets) with field methodological modifications.

3.4.2 *Body weight and abundance correlates*

The use of a simple Relative Abundance Index (RAI) based on camera-trap encounter rates for ecological studies is controversial particularly when comparing between species as a large number of variables (e.g. body size, average group-size, behaviour) are likely to affect trapping rates and detection probability and thus confound the relationship with actual abundance (Carbone et al., 2001, Jennelle et al., 2002, Treves et al., 2010). However, there is increasing evidence for a linear relationship between RAI and abundance estimated through more rigorous methodologies (Rovero and Marshall 2009). Camera trapping rate proved to be a useful index of abundance in small carnivore species from this study. However further comparative studies over different habitats, forest types at the landscape scale may aid in standardizing this technique for small carnivores. It is intuitive that camera trapping rate should be related to abundance. As density increases, the chance of encounters between individuals and cameras would be expected to increase. The likelihood of the observed relationship between camera trapping rates and density (from line transect) was applicable in forest ungulates (Rovero and Marshall 2009). This method has the potential for temporal comparison of populations and may facilitate to standardize and reduce costs of multi-species monitoring programmes. However different camera trap models would have different detection probabilities that could affect trapping rates which should be considered for future monitoring. Occupancy/trapping index with a refined approach has the potential to be applied on small ranging nocturnal species. Smaller species are more patchily distributed than large species which could explain the linear relationship between body size and RAI.

The strong positive relation between body weights and home range size was expected since a larger body weight facilitates the animal to travel longer distances. Larger species must collect more energy to supply their requirements than smaller species. Large energy demand will necessitate a large area for food gathering, unless food exists in superabundance. It is obvious that the size of a mammal will greatly affect the maximum area that can be covered thereby influencing the home range size. The type of food of a species also appears to be the determining factor for the size of home range; in the case of grey and ruddy mongoose that are thought to be insectivorous, would not have to travel long distances since their requirements could be conveniently fulfilled within their home range. The brown palm civet being strictly frugivorous and arboreal thrives in evergreen forests wherein the habitat itself has great potential in supplying fruits. Their strict arboreal nature allows them to utilize a three dimensional habitat structure due to increased access to strata above ground. Most civets are primarily frugivorous where fruit availability acts as a major predictor of their occurrence however in the case of fruit scarcity they might have to travel to fruit abundant patches and this factor could hamper home range sizes. The small Indian civet and common palm civet are known to have larger home ranges than the brown palm civet (Rabinowitz 1991, Joshi et al., 1995, Kumar and Umapathy 1999). Small cats on the contrary have larger home ranges than herpestids and viverrids. They are relatively more territorial and mostly the males would have overlapping ranges with several females and comparatively larger home ranges than females. Although there are no small cat home range estimates from India, I observed similar body sized estimates from available studies (Manfredi et al., 2006, Pereira et al., 2011). A wide range of home-range sizes for Malay civets has been documented in Buton (24–189 ha) and Borneo (27–283 ha) (Macdonald and Wise 1979, Nozaki et al., 1994, Colon 2002). A wide range of home-range sizes has been found for common palm civets at different study sites in Nepal and Thailand (Rabinowitz 1991, Joshi et al., 1995, Grassman 1998), and this was thought to be because of differences in habitat productivity (Joshi et al., 1995, Grassman 1998).

The weak relationship between body size and occupancy could be a result of individual range movements being greater for significantly larger species (>20 kg) than for small species.

3.4.3 *Jungle cat density and limitations*

Camera trapping provides a statistically robust estimate grounded in mark/recapture analysis, which can be used to determine jungle cat densities within a short period of time. Estimating carnivore density with ad hoc boundary strip methods can lead to overestimation and/or increased uncertainty as they do not incorporate spatial variation. Camera-trapping studies in conjunction with spatially explicit capture–recapture models provide a framework for estimating density of rare and elusive species. This framework includes the estimation of detection probabilities, which is critical in achieving unbiased estimates of abundance and density. In most studies the Mh model is chosen as a relatively more appropriate model showing the existence of heterogeneity in capture probabilities between individuals given the social structure of jungle cats and unequal access to camera traps. The density estimates of jungle cat from the present study are similar to estimates of small cats from other studies. The density of the Geoffrey cat ranged from 1.2 (0.3) to 2.9(1.4) individuals/km^2 in central Argentina (Periera et al., 2011); European wildcats, 0.1 to 0.5 individuals per km^2 (Anile et al., 2009). Density estimates for the jungle cat were extremely low and comparable with the density of Neotropical small cats; Pampas cat (0.74–0.79 individual/km^2 Gardner et al., 2010) and the Andean cat (0.07–0.12 individual/km^2). Jungle cat estimates from conventional methods are comparable with Andean cat densities (0.08 ± 0.05 to 0.17 ± 0.10) individual/km^2. In a review of lowland Neotropical cats the highest population densities of jaguarondi (*Puma yagouaroundi*), margay (*Leopardus wiedii*), little spotted cat (*L. tigrinus*), and Geoffroy’s cat ranged from 0.20 to 0.42 individual/km^2, and that of ocelots (*L. pardalis*), usually the most abundant species, reached 1 individual/km^2 in very productive ecosystems. Detection probabilities achieved from the present study were higher compared to Pampas cat, 0.02 (Gardner et al., 2010) and Andean cat, 0.07 (Reppucci et al., 2011). The jungle cat has morphological affinities (relatively short tail, long legs, long tufted and pointed ears) to African cats, such as serval

(*Leptailurus serval*) and caracal (*Caracal caracal*), which may indicate a preference for open habitats (as opposed to closed canopy forests). In India, jungle cats are often known to exist in high populations in agricultural ecosystems or near human habitations and open forests which also explains why I captured many individuals of jungle cats specifically in the thorn forest. Although the density is low there seems to be a satisfactory breeding population.

Mudumalai is a relatively large contiguous forest with undisturbed natural vegetation and high prey base. The eastern part of Mudumalai is connected to the Sigur and Singara Reserved forest which are also strongholds of lowland open scrub/thorn forests that has great potential in having high populations of jungle cats and other lesser cat species. The appropriate buffer value will depend on the characteristics of the sampling array layout (size, shape, and trap spacing) and the unknown home ranges of the sampled animals that may differ in size, shape, overlap, and proportion contained within the sampling area. Given the uncertainties of using 1/2MMDM and MMDM to buffer *N* in density estimation, carnivore studies often use both values, reporting two density estimates (Trolle et al., 2007). This is unsatisfying for conservation organizations attempting to identify populations and species at risk, as 1/2MMDM densities are almost twice than that of using MMDM. This study shows that as with other carnivore species, photographic capture-recapture methods are applicable for estimating the abundance of small felid populations, as they are a fast, effective and noninvasive method for establishing the status of populations. The deployment of remote cameras offers a helpful means to obtain additional information. Camera trapping makes it possible to establish the limits of the distribution area, to obtain information about the population sex ratio, and occasionally, to detect individuals with poor body condition or which are injured. This permits rapid response action for these individuals, if necessary. Estimates of the MMDM are constrained by the size of the sampling grid, as camera traps do not capture any movements beyond it. Thus, this approach likely underestimates movements and hence overestimates density. The issue of underestimating animal movement is important and applies to virtually all camera trapping surveys which usually sample small areas.

I successfully identified jungle cats by their body pelage patterns (flank, fore-limb, hind-limb) and body size. The most useful photos for individual identification were those showing animals with one entirely visible flank, complete tail, and atleast clear fore-limbs. Considering these results, I recommend to conduct such surveys for at least a month with an inter-trap distance of 600 m to achieve rigorous movement rates of jungle cats and high detection probabilities. Based on the spatial distribution of jungle cat encounters in the present study, I draw the conclusion that their movements were not fully addressed due to the low sample size and, consequently, density across the sampled area was much lower. With the increased amount of information on jungle cat movement within the sampled area due to the large scale of the sampling grid and by explicitly modeling movements, the order of magnitude of density estimates is likely to be more realistic. For such purposes, SECR models have the additional advantage over the traditional approach in that they provide a unified and formalized approach to estimate density, thus rendering them comparable. For surveys with small samples it would be much appropriate if I investigate further whether the low capture probability is an artifact of low detection probability at the assigned camera locations or truly represents low abundance of the target species within the study area.

It is India's most common small cat, inhabiting mostly arid, semiarid scrublands and grasslands, including areas under livestock pressure and croplands. It is rare in eastern India's thick forests than in the west's relatively more open habitats. The major threats to the jungle cat include habitat destruction, population fragmentation, poaching and hybridization with domestic cats (Duckworth et al., 2008a). The jungle cat belongs to the domestic cat lineage with strong morphological affinities to certain scrub dwelling African cats in appearance and distribution (Pocock 1939, Johnson et al., 2006). It shows strong geographical variation in body size and morphology (Mukherjee and Groves 2007). Mukherjee et al., (2010) explored the importance and power of molecular techniques to compare their population genetic structure across India yet there is lack of adequate data on its basic ecological aspects (Mukherjee 1998). A recent study by Gupta (2011) documented the abundance of jungle cat in an occupancy framework.

Although the survey area is under tremendous pressure from anthropogenic activities, jungle cats seem to be tolerant to some degree of habitat alteration however through long-term monitoring it is necessary to understand their spatial responses towards the degree of human disturbance. Several authors (Johnson and Franklin 1991, Cuellar et al., 2006) noted that Geoffroy's cats use small parts of their territory for a relatively short period of time (up to 3 months) and then moved to other areas or even abandoned their home ranges. This behaviour was also evident for radio-collared Geoffroy's cats in Lihue´ Calel (Pereira et al., 2006, Pereira 2009), and suggests the existence of an unstable home-range behaviour and little site fidelity by this felid, at least under severe environmental conditions. Recent studies have demonstrated that spacing between cameras, survey area size and scant information on true home-range size can influence the effective sampled area, resulting in an erroneous population density estimate. My estimations of jungle cat density are low although I do not have any comparable estimates; however this was expected as higher population persists in purely arid and semi-arid zones of the country in North-western India due to the preference by the species. As in the rest of jungle cat distribution, rigorous population estimates do not exist. Establishing a repeated and consistent monitoring program employing camera-trapping could be an essential tool to detect population and/or range changes and for properly assessing the effectiveness of conservation investments targeting the jungle cat. Future developments despite improved methodologies and equipment remain difficult to study. SECR models are an analytical step forward from traditional approaches to estimate densities of small felids and other elusive species, since they overcome the problem of interpreting abundance and make use of the full information obtained by photographic data, including auxiliary spatial information. Nevertheless these require large financial and logistical efforts in order to obtain sufficient data for reliable inference.

The use of these models for a suite of species that may not be uniquely identifiable provides a solution to monitoring populations of cryptic and nocturnal species in a systematic and statistically sound manner. However, its use is subject to certain caveats. Namely, that estimates are likely to improve (lower standard errors) when animal-specific detection probabilities are improved with the inclusion of covariates, and also when the number of repeated visits are increased and more sites (patches, cells) are surveyed. Therefore, it may be possible to obtain a reliable estimate of abundance, index of abundance or occupancy, if sampling methods are better tailored to species biology and is designed to increase animal specific detection probability. When home range sizes or movement rates can be established, densities may also be estimated. It was expected that small carnivores may have low animal-specific detectability (about 0.1 to 0.3 '*r*'). Royle and Nichols (2003) suggest that reliable estimates can be arrived at even at sample sizes of 100 cells and low '*r*', provided that the number of replicates is high. This method appears to provide a novel approach to monitor the distribution and abundance of species whose elusive traits have so far, prevented non-invasive abundance estimation using conventional techniques. It can potentially be a reliable method for the monitoring of multiple species from the same field survey, thus being especially useful for community level monitoring of animal populations and biodiversity. Uncertainty can be reduced by improved sampling techniques. The use of a systematic, likelihood-based method that also estimates detection probabilities can provide reliable estimates of at least an index of abundance if not abundance itself, and prove to be a robust tool in community and biodiversity monitoring. It is thus possible for us to now survey and monitor a range of rare, elusive species that hitherto, could not be surveyed, with unprecedented rigor. In the next chapter I discuss about the covariates influencing detection probability of these species in the three habitat types.

The results of this study showed that the occupancy-based Royle and Nichols (2003) and Repeated count model can provide a generally useful tool for the estimation of abundance of non-uniquely identifiable and cryptic species. Parameter estimates with high standard errors (as in this study) have high uncertain estimates, and therefore reliable estimates of absolute abundance may not be obtained. However, it is still possible to arrive at an index of abundance that takes detection probability into account. Such an index is comparable across temporal scales for instance, and can provide important insights into population trends.

Most small ranging carnivore species were not widely dispersed across sampling locations, but clustered in a few locations. Much of the camera trap data collected across Asia has been the result of opportunistic sampling. Mark/recapture and patch occupancy provide extremely useful tools for the detection of trends in wildlife population abundance and species presence/absence. The main advantages they offer are explicit treatment of detection probabilities, error assessments, and estimation of confidence intervals (Mackenzie et al., 2006). I recommend that in order to fully document diversity of small carnivores camera-trapping will need to be augmented by other methods such as spotlight surveys to detect small carnivores such as primarily arboreal civets, and rare species. This survey confirms that Mudumalai Tiger Reserve harbours nine small carnivore species of global conservation concern and is one of the most important areas for maintaining the diversity in carnivore-community in south India.

Chapter 4

DIETARY PATTERN OF SMALL CARNIVORES

4.1 Introduction

Ecologists have often acknowledged differences in body-size, feeding behavior and morphology among sympatric carnivores enabling differential food exploitation to potentially reduce competition (Rosenzweig 1966). Diet of small carnivores includes a wide range of animals, plants, insects, crustaceans, vertebrates from birds to large mammal carrion and fruits. Although they must respond at some level to site productivity, the tremendous range of food items they eat is produced in such a wide range of environment, and so difficult to measure, that the apparent food abundance is not highly predictive of the distribution or abundance of most species. A central theme in ecology is how animals respond to variation in food abundance and distribution. For some species, resource fluctuation may drive seasonal movements or regulate populations (Wiens 1976). Other species may simply switch their diet in response to scarcity of a given prey type (Martin et al., 1951, Whelan et al., 2000). A principal reason is the difficulty in accurately quantifying food abundance. Inconsistencies in the temporal or spatial coupling of plants and frugivores are likely to affect the evolutionary potential of the interaction (García and Ortíz-Pulido 2004). Some predators switch between primary and alternative prey items as the availability of food resources change. Although temporal switching of prey is well documented, spatial switching (e.g. foraging in different habitats) remains controversial (Prugh 2005). Generally, prey switching may be a consequence of the choice of prey items within a habitat or of the choice of habitat with different types of prey. This diet switching has been shown for the red fox (*Vulpes vulpes*—Ferrari and Weber 1995, Kjellander and Nordstro¨m 2003), martens (*Martes americana*—Thompson and Colgan 1990) and masked palm civet (*Paguma larvata*) (Zhou et al., 2008). One way of understanding community organizations, could be to measure overlap in resource use among the different species in a community guild (Krebs 1989). Higher food overlap, however, does not necessarily mean competition. Species can still coexist if: 1) food is abundant, 2) prey is partitioned by size (Rosenzweig 1966) or differential use (Mills 1984), and/or diversity is greater in the diet of one species' than it is in the other (White et al., 1995).

Factors affecting food choice by predators generally differ across landscapes, and this knowledge is essential to determine the ability of predators to deal with different prey compositions and habitat characteristics. For several reasons, small mammals are a highly profitable prey for carnivores. Small mammals usually constitute the main prey for small cats. Patterns of rodent selection by felids address whether prey choice may be related to the availability, morphology, and distribution of different rodent species. Cat's diet through scat analyses along with field estimates of rodent abundances from trapping is essential to investigate predation according to the availability (Bisceglia et al., 2011). In recent years, globally, much attention has been stressed on the ecology of free-roaming cats/feral cats due to predation pressure over native threatened prey species and social and sanitary problems arising from their presence and over-abundance (Genovesi et al., 1995). The Geoffrey cat has been described as an opportunistic predator (Canepuccia et al., 2007) feeding mainly upon introduced hare (*Lepus europaeus*) and small rodents (Johnson and Franklin 1991, Vuillermoz and Sapoznikow 1998). The main component of the diet of both pampas cat and Andean mountain cat was rodents (71% and 82%, respectively), followed by birds (27.5% and 18%) (Napolitano et al., 2008).

The important role of viverrids, as consumers and dispersers of fleshy-fruit plants is arguably a characteristic feature of their specialized habitats, thus contrasting with tropical environments where apparently along with monkeys, rodents, bats and large herbivores predominate in the mammal frugivore assemblages. Common palm civet is an omnivore that feeds on fruits and meat (Joshi et al., 1995). In forested areas, it is primarily frugivorous, feeding on berries and pulpy fruits and is an effective seed disperser (Rabinowitz 1991, Nakashima et al., 2010). It is often considered a pest because of its raiding of coffee plantations, other fruit crops and poultry, even though the coffee beans recovered from its faeces are used to make a high-value special coffee (Prater 1971). Wang et al., (1976) studied the diet of *Viverricula indica* in northern China, and found that it ate a lot of rodents and plants. Rabinowitz (1991) studied the diet of several civet species in Thailand. Diet studies have shown that seasonal variation in fruit consumption is quite common in frugivorous carnivores (Ferrari and Weber 1995, Martinoli et al., 2001). Most of the earlier studies only recorded the overall frequency of occurrence of various food items in the scats or stomach content. Rarely can one find enough information to compare the diet of these species and seasonal variation, especially where they are sympatric (Rabinowitz 1991). However, explicit and quantitative covariation with fruit supply (i.e. spatio-temporal variation in fruit abundance is linked to changes in the diet of carnivores) is yet to be addressed in most carnivores and other fruit eating mammals (Moegenburg and Levey 2003).

On the other hand, mongooses comprise a carnivore group exhibiting considerable diversity in ecology and behaviour (Rood 1986). It has been reported to feed on a variety of poisonous arthropods, harmful insects and rodents. In addition they have also been reported to feed on insects, crabs, frogs, reptiles, bird, small mammals, scorpions and other invertebrates. The small Indian mongoose (*Herpestes javanicus*) preys upon a variety of invertebrates, arthropods, small reptiles and mammalian species in Pakistan (Mahmood et al., 2011). Dwarf mongoose mainly feed on insects like termites, locusts, beetles, grubs, larvae and spiders (Osborn 1998). Several field studies have revealed the small Indian mongoose to be primarily an insectivore, though it also feeds opportunistically on small vertebrates (Cavallini and Nel 1995). An early field study on the amount and type of food eaten by the small Indian mongoose on the island of Trinidad depended largely on the opportunistic food available. Most herpestids are predators, feeding on a wide range of animals including small mammals and birds (including bird eggs), reptiles (especially snakes), a wide variety of insects, and crabs. Some species also include vegetable material in their diets, feeding on tubers, fruits, and berries (Feldhamer et al., 1999). Diet of the small Indian mongoose in the Fiji Islands consists of a wide variety of foods, both invertebrates (crabs and insects) and vertebrates (amphibians, reptiles, birds, and mammals) (Gorman 1975). Very few studies in India documented the feeding ecology of sympatric small carnivores (Joshi et al., 1995, Mukherjee et al., 2004, Mudappa et al., 2010, Gupta 2011).

The objectives of my research were: 1) to study the interspecific and intraspecific seasonal differences in feeding habits of small cats, civet and mongoose 2) to examine the changes in preferences for major food items preyed upon in relation to their biomass; 3) to analyze the trophic niche separation and overlap between these sympatric predators based on diet composition.

4.2 Materials and methods

The study was conducted from October 2009 to May 2011.

4.2.1 *Fruiting seasonality and biomass estimation*

A total of 19 plant species were monitored for the study in the three habitat types. Ten samples of each species were tagged which included 10 trees (*Cassia fistula*, *Cordia obliqua*, *Grewia tiliifolia*, *Ficus spp*, *Bischovia javanica*, *Symplocos cochinchinensis*, *Bridelia crenulata*, *Syzigium cumini*, *Dillenia pentagyna* and *Diospyros montana*), six shrubs (*Zizyphus oenophilia*, *Lantana camara*, *Antidesma zeylanicum*, *Coffee spp*, *Zizyphus mauritiana*, and *Grewia hirsuta*) two lianas (*Embelia gardneriana* and *Gnetum ula*) and one climber (*Piper nigrum*). These tagged individuals were monitored twice a month for presence/absence of fruits from October 2009 to May 2011. For fruit biomass estimation fruit production of the tagged trees was monitored every two weeks. Total fruit biomass per tree was estimated by weighing the total number of fruits on the maximum fruiting branch and then multiplying this value by the number of similar sized fruiting branch. The number of ripe fruits on representative branches was counted on every visit. I estimated the availability in g/m^2 of fruits for each month during the study period.

4.2.2 *Rodent and shrew abundance estimation*

Web trapping is preferable to other techniques in estimating population density of small mammals (Otis et al., 1978, Wilson and Anderson 1985). Few studies on rodent abundance are available in India (Chandrasekar-Rao and Sunquist 1996, Shanker and Sukumar 1998). Density of rodents and shrews were estimated using a trapping web design (Anderson et al., 1983). Rodent trapping was conducted in 72 locations to assess seasonal changes in relative abundance. Each location was operated by a single trapping web. Totally 72 trapping webs (26 in dry thorn forest, 25 in deciduous and 21 in semi-evergreen forest) using 32 Sherman traps (H. B. Sherman Traps, Inc., Tallahassee, Florida) were laid around each camera location in the reserve. Trapping was conducted for two years at each location once in dry and wet seasons amounting to a total effort of 4416 trap-nights in the dry season and 3072 trap-nights in the wet season. Trapping in the semi-evergreen forest could be conducted only in the dry season due to inaccessibility and logistic constraints. Each trapping web consisted of eight radial lines 20 m each at cardinal directions. Each trapping line had 4 traps; placed at 5-m intervals from the center (5, 10, 15, and 20 m). The traps were baited with fried coconut pieces. The circular area of each web was 0.12 ha. Traps were set in the evening and checked the next day morning. All rodents and shrews captured were sexed, weighed, measured and released at the trap site. Small mammals were identified using appropriate field guides (Prater 1971, Menon 2003). Unidentified mammals were collected and preserved in ethanol for identification by experts in the laboratory at the Zoological Survey of India. Data for each trapping period were pooled for two years for statistical analyses. Rodent and shrew density was estimated using program Distance 6.0 (Thomas et al., 2009). Analysis was done with fitting different detection functions to the observed data. The best model was selected on the basis of the lowest Akaike Information Criteria (AIC) values (Burnham et al., 1980, Buckland et al., 2001). Each dataset was analyzed by four models (uniform, half-normal, and hazard and negative exponential) with three possible model adjustments (cosine, polynomial, and hermite). Binomial model was fitted for the species which was trapped 5 to 15 times. Pooled species estimated strip width (ESW) was used to derive density wherever species were trapped less than five times. Suitable modifications in right truncation were made so as to ensure a reliable fit of key functions and adjustment terms to the data so as to arrive at density estimate. To get overall density in dry and wet season, sightings were pooled together for both the years.

4.2.3 *Bird and hare density estimation*

To monitor the seasonal variation in ground-dwelling birds and hare the intensive study area was divided into 1 km x 1 km grids and in each grid, a line transect (1 km) was laid (n = 69, total length = 69 km). Each transect was walked thrice in dry (Feb-May) and wet seasons (Sept-Dec) for two years (2009-2011). The trails in semi-evergreen forest could be sampled only in the dry season due to inaccessibility in the wet season. The total effort amounted to 702 km. All transects were walked in the early morning between 6.30 am to 8.30 am after sunrise. For every sighting, time, species, group/flock size, animal bearing (using a hand held compass) and the angular sighting distance (using a laser range finder) were recorded. Hare and bird species density was estimated using program Distance 6.0 (Thomas et al., 2009). Analysis was done with fitting different detection functions to the observed data for estimation of densities. Each dataset was analyzed by four models (uniform, half-normal, and hazard and negative exponential) with three possible model adjustments (cosine, polynomial, and hermite). The best model was selected on the basis of the lowest Akaike Information Criteria (AIC) values (Burnham et al., 1980). Binomial model was fitted for the species which was sighted 5 to 15 times. Pooled species estimated strip width (ESW) was used to derive density wherever species was sighted less than five times. Suitable modifications in right truncation were made so as to ensure a reliable fit of key functions and adjustment terms to the data so as to arrive at density estimate. Density was estimated separately for dry and wet season in each habitat. To get overall density in dry and wet season, sightings were pooled together for both the years. Densities were compared between habitats and seasons using Student's t-test (Zar 1999).

4.2.4 Diet composition

Sample collection and identification of faeces

Diet of small carnivores was studied through faecal analyses, a widely used method (Mudappa 2001, Mukherjee et al., 2004, Zhou et al., 2008). I established 36 trails (3 to 5 km) where each trail was surveyed systematically twice a month. During the first visit to each trail all detected faeces were removed to ensure that only fresh feces were collected during later visits. Faeces were also collected along roads, trails, near fruiting trees and termite mounds opportunistically. Faeces were identified from their characteristic shape, size, nearby tracks, foraging signs, and frequent defecation sites. Small cat faeces were identified based on their characteristic shape with twists and turns having pointed ends with a diameter (≤ 2 cm in diameter). Based on descriptions from past studies on civets and field knowledge, faeces were identified from their characteristic appearance; rounded ends, usually defecated along fallen logs, rocks, and lianas (Bartels 1964). Civet faeces were readily distinguished from bears because of their smaller dimensions and unique defecation characteristics. Civetries were also located for fresh droppings especially around fruiting trees and over-hanging branches along streams, the banks of small streams and tree-fall gaps (Nakashima et al., 2010). Mongoose faeces were collected along trails, over termite mounds and dried stream beds from their characteristic bulky appearance, and cylindrical shape with staggered ends (≤ 3 cm in diameter). Collectively, these procedures suggest that the probability of misidentification of civet, mongoose and small cat feces was low.

Food identification and estimating biomass for small carnivores

After collection, faeces were placed in a plastic bag, tagged and air-dried. Faeces were washed in water through a nylon mesh sieve to separate food remains and later sun-dried. Dietary components were identified to species/taxa using based on pertinent anatomical elements such as hair, mandibles, tooth rows, scales, feathers, wings, elytra, fruit cuticles, and seeds. Fruit seeds were identified with the help of experts from the herbarium section at the Wildlife Institute of India and a reference collection was made in the field. Hair remains were identified with the help of reference slides at the Research Laboratory of Wildlife Institute of India. To obtain estimates of rodent consumption by the carnivores, I first identified rodent species from the jaws found in faeces by comparing the mandibles with known specimens collected from the field, along with photographs and descriptions from Roberts (1977) and Mukherjee (1998). I then calculated the number of individuals of each rodent species consumed by counting jaw parts (jaws were usually present as 2 halves of upper and 2 halves of lower jaws, hence 4 parts would make 1 individual). However in some, all parts were not present, and in these cases the presence of even a single portion or fragment of jaw with dentition was recorded as 1 individual.

I calculated biomass of different prey types consumed per predator by multiplying average prey biomass with number of prey items consumed by small carnivores. Average prey biomasses for rodents and insectivore were obtained from trapping data. Wherever possible rodents were identified to the species level, otherwise they were categorized as unidentified rodents (weight calculated as the average body mass of all rodent species). Fresh fruit biomass consumed by study species for each sample was calculated by multiplying average wet weight of fruits with total number of seeds consumed. Body mass of hare, fish, reptiles, arthropods were collected from available literature (Menon 2003, Renjithkumar et al., 2011).

I estimated the diversity of each predator's diet using the Shannon diversity index (Magurran 2004), randomizing the original order of scat samples (1000 iterations) using the software EstimateS (Colwell 2006). Following this I assessed dietary diversity against sample size to determine whether the sample size was adequate to describe the diet of each predator.

I expressed diet composition in 3 complementary ways: percentage of the fresh biomass consumed (PB), percentage of occurrence (PO), and relative frequency of occurrence (RFO).

The PB from the total fresh mass of all food items (Loveridge and Macdonald 2003) was calculated as: consumed biomass of species or taxonomic group x 100/total biomass consumed.

The PO of prey items in the diet (Genovesi et al., 1995) was calculated as:

number of occurrences of species or taxonomic group x 100/total number of fecal samples.

The RFO (Loveridge and Macdonald 2003) was defined as: number of occurrences of species or taxonomic group x 100/total occurrences of all food types in all samples.

All three measures must be treated cautiously because feaces collected in series can contain common contents from a single large meal, or similar locally common small items, but small items can be overestimated (Atkinson et al., 2002).

I used Wilcoxon paired sample test (Wilcoxon 1945) to determine differences in observed frequencies of occurrence between food items and to determine differences in frequency occurrence and percent biomass between habitats. Diet composition (relative frequency of occurrence) of major food items (prey items were pooled) was compared between habitats by χ^2 contingency table analysis (Zar 1999). Mann-Whitney *U* test was applied to check for seasonal differences between species. Important prey types were then used to investigate monthly variation in diet.

Shannon's (Pielou 1966) diversity index (H') and Levins' (1968) index ($B = 1/\sum pi^2$) was calculated where pi is the percent biomass of a given item, provide useful indicators of food diversity and niche breadth and were used to examine temporal variation. For each sample, H' and B' was calculated using the proportion of biomass estimate of each species present in the sample. Levin's index was standardized to a scale of 0–1 following Hurlbert (1978), $B_s = (B - 1)/(n - 1)$ where *n* is the total number of food species recorded.

To assess similarity of food composition between small cat, civet and mongoose, the Pianka's niche overlap index was calculated (Pianka 1973) for dry and wet season. The index distributes between 0 and 1, the similarity is higher as the index is close to 1. Pearson's correlation was calculated between percent fruit biomass available in the environment and percent fruit biomass in civet feaces. I used Jacob's index (*D*) to determine food selectivity of small carnivores by the formula $D = (ri\text{-}pi)/(ri+pi\text{-}2ripi)$; where r_i = proportion of biomass of the given ($i^{th)}$ food category/taxon/species in the diet and p_i = proportion of biomass of the given (i^{th}) food category/taxon/species in the available prey community (Jacob 1974). Selectivity varies from -1.0 to +1.0. The Bonferroni simultaneous-confidence-interval approach of Marcum and Loftsgarden (1980) was applied to assess which food item was selected preferentially with ≤ 0.05 significance. I used SPSS 16 (SPSS 2007) statistical package to process data. All statistical tests were evaluated ≤ 0.05 significance level unless otherwise stated.

4.3 Results

4.3.1 *Fruit biomass availability and seasonality*

The 19 plant species were examined in detail for fruit availability, as they were identified as important species based on their percent occurrence (≥ 1%) in civet faeces (Table 9). Biomass and fruit availability could not be recorded for *Caryota urens* as it did not occur inside the reserve but appeared in the nearby tea/coffee plantations. These 19 species contributed to 61.53% of frequency occurrence in the diet of civet. Of these plants, *Grewia tiliifolia*, *Grewia hirsuta* and *Lantana camara* produced fruits throughout the year. Some species like *Zizyphus mauritiana*, *Cassia fistula*, *Coffee spp*, *Zizyphus oenophilia* showed bimodal fruiting pattern, producing fruits from Feb-May and Oct-Dec. *Bischovia javanica*, *Piper nigrum*, *Diospyros montana*, *Symplocos cochinchinensis*, *Dillenia pentagyna*, *Embelia gardeneriana*, *Ficus spp*, *Gnetum ula* and *Cordia obliqua*, produced fruits exclusively from Feb-April indicating a short fruiting period. Species that fruited exclusively during the wet season were *Syzigium cumini* and *Bridelia crenulata*. *Antidesma zeylanicum* began fruiting at the end of the second wet season in December and continued fruiting till May. Number of species fruiting were maximum during the dry season (Feb-May) as compared to wet season (July-Dec)(Fig 10).

Fig 9. Monthly percent fruit biomass of 19 plant species monitored in Mudumalai Tiger Reserve (2009-2011).

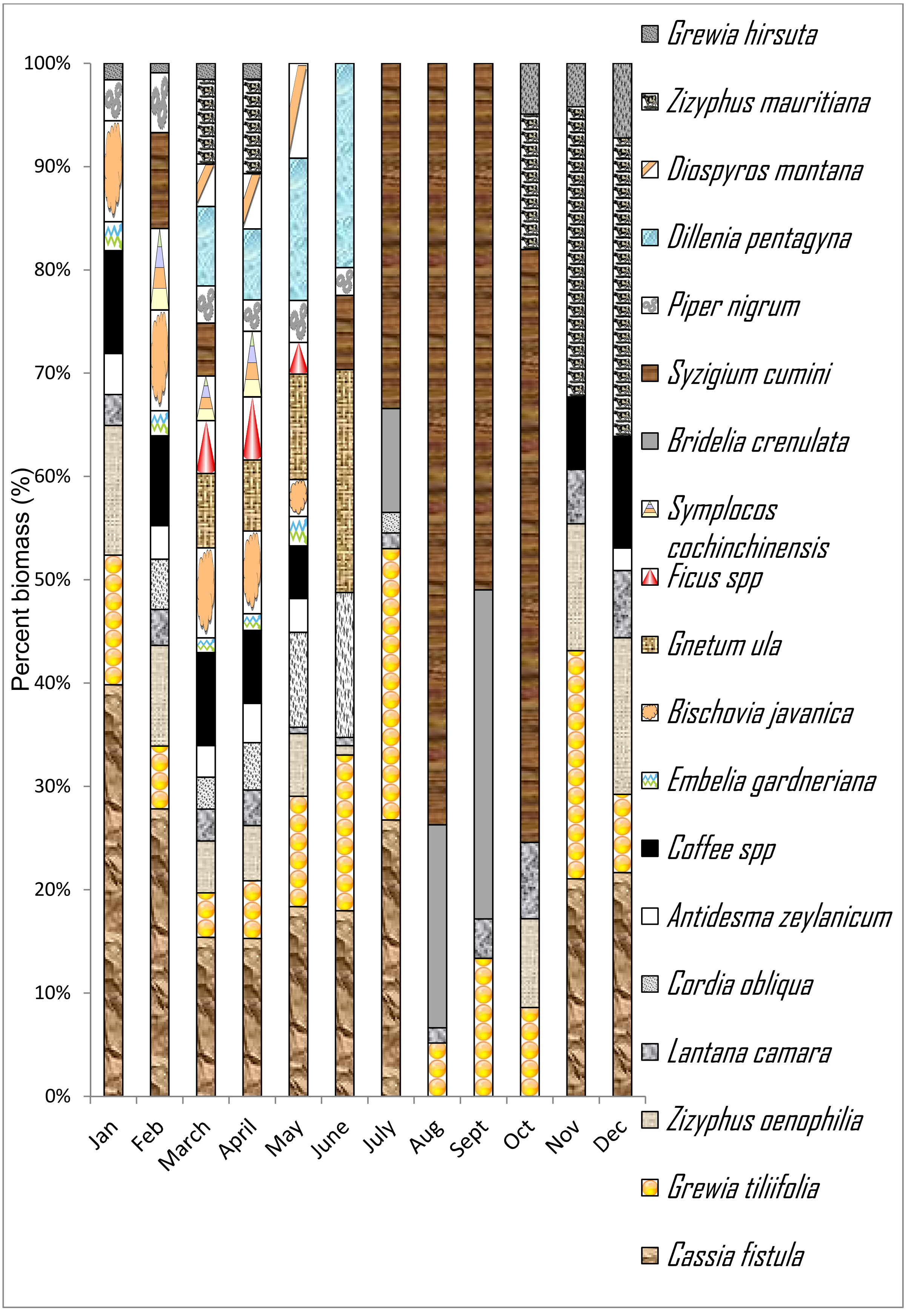
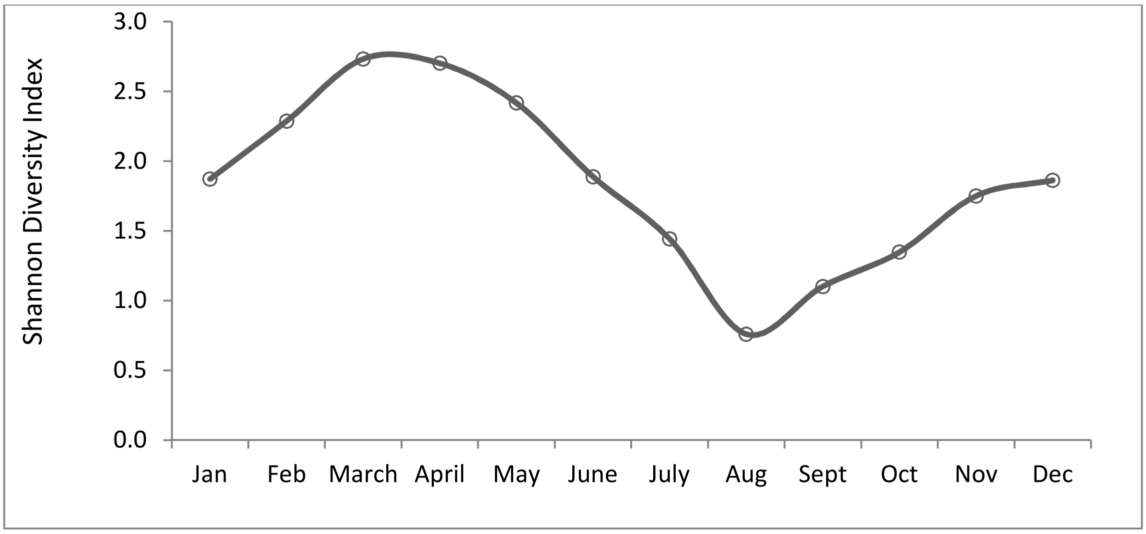


Fig. 10. Monthly fruit diversity recorded in Mudumalai Tiger Reserve (2009-2011).



4.3.2 *Rodent and insectivore abundance*

The two year trapping sessions revealed the presence of eight species of rodents and an insectivore. Overall there was no difference in small mammal abundance between dry and wet seasons (t test, t = 0.67, df = 8, P = 0.26) (Table 11). Collectively, abundance was higher in the dry season (95.25 individuals/ha) than wet season (65.98 individuals/ha). There was no difference in rodent abundance between dry and wet season in the thorn (t test, t = -0.36, df = 7, P = 0.36) and deciduous forest (t test, t = 0.44, df = 5, P = 0.33) (Table 10).

Dry season

Although abundance was higher in deciduous habitat, it was not significantly higher than thorn (t test, t =-1.89, df = 4, P = 0.06) and semi-evergreen forest (t test, t = 1.49, df = 2, P = 0.14). In the dry season *Mus platythrix* was the most abundant species (19.2 ± 5.8 individuals/ha) in the thorn forest and deciduous forest (44.88 ± 10.27 individuals/ha). *Millardia meltada* was the least abundant species (0.66 ± 0.06 individuals/ha) found in the thorn forest while *Rattus rattus rattus* was the least abundant species in deciduous forest (1.76 ± 0.42 individuals/ha). *Rattus rattus wrougtoni* was the most abundant species in the semi-evergreen forest (72.08 ± 6.44 individuals/ha) while *Suncus montanus* was the least abundant species (5.96 ± 1.96

individuals/ha). *Rattus rattus rattus* and *Mus musculus* were captured only once in the dry thorn forest while *Cremnomys blanfordi* was captured once in the semi-evergreen forest and hence I could not achieve its reliable density estimates for these species within a particular habitat. Across the reserve, *Mus platythrix* was the most abundant species (41.91 ± 9.09 individuals/ha) while *Millardia meltada* and *Tatera indica* were the least abundant (0.36 ± 0.08 individuals/ha respectively).

Wet season

There was no difference in abundance between thorn and deciduous forest (t test, t = -0.56, df = 6, P = 0.29). In the thorn forest, *Mus booduga* was the most abundant species (38 ± 13.21 individuals/ha) while *Millardia meltada* was the least abundant species (1 ± 0.11 individuals/ha). In the deciduous forest *Cremnomys blanfordi* was the most abundant species (39.98 ± 5.71 individuals/ha) while *Millardia meltada* (0.74 ± 0.12 individuals/ha) was the least abundant species. Across the reserve *Cremnomys blanfordi* was found to be the most abundant species (18.57 ± 3.02 individuals/ha) while *Rattus rattus rattus* was the least abundant species (0.49 ± 0.27 individuals/ha). *Rattus rattus rattus* was captured only once in the thorn forest and hence reliable abundance estimates for this species could not be achieved within a particular habitat.

Table 10. Density of rodents and insectivore estimated during the dry and wet season in Mudumalai Tiger Reserve (2009-2011).

Habitat	Species	D ± SE (ind/ha)	No. of captures	% CV	95% Confidence Interval	ERD ± SE (m)	Model	P- value	AIC
DRY-DTF	*Funambulus palmarum*	6.83 ± 0.85	21	12.46	5.29- 8.82	14.58 ± 0.85	USP	0.21	64.10
DRY-DTF	*Tatera indica**	0.99 ± 0.09	3	9.53	0.81-1.22	14.42 ± 0.67	USP	0.40	46.83
DRY-DTF	*Rattus rattus wroughtoni*	7.51± 0.37	22	4.93	6.81- 8.29	14.23 ± 0.16	USP	0.41	65.33
DRY-DTF	*Cremnomys blanfordi*	4.88 ± 0.72	15	14.70	3.63-6.55	14.58 ± 0.76	USP	0.89	40.65
DRY-DTF	*Millardia meltada**	0.66 ± 0.06	2	9.54	0.54- 0.81	14.42 ± 0.67	USP	0.40	46.83
DRY-DTF	*Mus platythrix*	19.2 ± 5.8	13	30.47	10.05-36.69	6.84 ± 1.03	HNH	0.74	29.01
DRY-DTF	*Mus booduga*	7 ± 1.22	18	17.43	4.87-10.06	12.96 ± 1.1	USP	0.73	48.01
DRY-DTF	*Suncus montanus*	2.86 ± 0.44	9	15.46	2.03- 4.02	14.76 ± 1.07	USP	0.35	30.59
DRY-DEC	*Funambulus palmarum**	5.92 ± 2.23	8	37.70	2.50-13.99	9.27 ± 1.74	HNSP	0.23	63.4
DRY-DEC	*Rattus rattus wroughtoni*	13.83 ± 0.55	44	3.97	12.77- 14.98	14.23 ± 0.02	USP	0.51	124.47
DRY-DEC	*Rattus rattus rattus**	1.76 ± 0.42	3	23.68	1.08-2.86	10.41 ± 1.22	HNSP	0.17	63.4
DRY-DEC	*Cremnomys blanfordi*	38.85 ± 5.58	73	14.37	29.25-51.59	10.94 ± 0.70	HNC	0.19	205.87
DRY-DEC	*Mus platythrix*	44.88 ± 10.27	26	22.88	28.14- 71.57	5.83 ± 0.66	HNH	0.10	49.21
DRY-DEC	*Suncus montanus**	4.11 ± 0.98	7	23.78	2.52- 6.69	10.94 ± 1.22	HNSP	0.17	63.4
DRY-SEM	*Rattus rattus wroughtoni*	72.08 ± 6.44	50	22.81	45.87- 113.27	7.25 ± 0.80	HNSP	0.24	143.36
DRY-SEM	*Mus platythrix* *	7.15 ± 2.36	6	33.07	3.53- 14.49	7.97±1.30	HNC	0.63	28.81
DRY-SEM	*Suncus montanus* *	5.96 ± 1.96	5	33.01	2.94-12.07	7.97 ± 1.30	HNC	0.63	28.81
WET-DTF	*Funambulus palmarum*	4.48 ± 0.60	14	13.50	3.36- 5.98	14.69 ± 0.96	USP	0.43	42.12
WET-DTF	*Tatera indica**	3.69 ± 1.45	6	39.23	1.44- 9.46	14.41± 0.74	UH	0.08	16.64
WET-DTF	*Rattus rattus wroughtoni*	12.54 ± 1.79	36	14.25	9.43-16.69	14.09 ± 0.91	USP	0.96	97.66
WET-DTF	*Cremnomys blanfordi*	2.03 ± 0.16	11	8.16	1.72-2.39	19.36 ± 0.56	UC	0.43	25.22
WET-DTF	*Millardia meltada**	1 ± 0.11	3	10.56	0.79-1.26	14.41± 0.74	USP	0.62	16.64
WET-DTF	*Mus platythrix* *	1.33 ± 0.14	4	10.55	1.06-1.69	14.41± 0.74	USP	0.20	16.64
WET-DTF	*Mus booduga*	38 ± 13.21	22	34.76	18.84-76.62	6.32 ± 1.10	HNSP	0.71	33.32
WET-DTF	*Suncus montanus* *	1 ± 0.11	3	10.56	0.79-1.26	14.41± 0.74	USP	0.20	16.64
WET-DEC	*Funambulus palmarum*	3.25 ± 0.73	9	22.61	1.95-5.43	13.01±1.46	USP	0.27	27.76

WET-DEC	*Rattus rattus wroughtoni*	10.89 ± 1.51	35	13.85	8.24-14.39	14.02 ± 0.93	USP	0.48	99.83
WET-DEC	*Rattus rattus rattus* *	1.11 ± 0.17	3	15.68	0.80-1.54	12.84 ± 0.99	USP	0.82	50.02
WET-DEC	*Cremnomys blanfordi*	39.98 ± 5.71	63	14.29	30.12- 53.07	9.82 ± 0.66	HNH	0.42	171.77
WET-DEC	*Millardia meltada**	0.74 ± 0.12	2	15.68	0.53-1.03	12.84 ± 0.99	USP	0.82	50.02
WET-DEC	*Mus platythrix*	13.96 ± 3.7	18	26.51	8.08-24.12	8.88 ± 1.15	USP	0.95	24.92
WET-DEC	*Mus musculus* *	2.22 ± 0.36	6	16.17	1.59-3.11	12.84 ± 0.99	USP	0.82	50.02
WET-DEC	*Mus booduga* *	2.59 ± 0.41	7	15.82	1.86-3.61	12.84 ± 0.99	USP	0.82	50.02
WET-DEC	*Suncus montanus*	22.91± 9.75	14	42.57	9.49-55.30	6.12 ±1.3	HNC	0.48	21.72

DRY-DTF = Dry season in dry thorn forest; DRY-DEC = Dry season in deciduous forest ; DRY-SEM = Dry season in semi-evergreen forest; D ± SE = Density (individuals/ha) and associated standard error; ERD ± SE = Effective radial distance (m) with associated standard error; AIC = Akaike information criterion values; USP = Uniform simple polynomial; UH = Uniform hermite; UC = Uniform cosine; HNH = Half-normal hermite; HNSP = Half-normal simple polynomial; HNC = Half-normal cosine; * = Pooled species ERD was used to derive density.

Table 11. Overall density of rodents and insectivore estimated during the dry and wet season in Mudumalai Tiger Reserve (2009-2011).

Season	**Species**	**D ± SE (ind/ha)**	**No. of captures**	**% CV**	**95% Confidence Interval**	**ERD ± SE (m)**	**Model**	**P-value**	**AIC**
DRY	*Funambulus palmarum*	5.25 ± 1.08	29	20.60	3.46-7.97	11.29 ± 1.16	HNC	0.05	91.07
DRY	*Tatera indica**	0.36 ± 0.08	3	22.53	0.22-0.6	13.75 ± 1.55	USP	0.89	32.15
DRY	*Rattus rattus wroughtoni*	21.96 ± 2.26	116	10.31	17.92-26.92	11.04 ± 0.56	HNC	0.23	343.53
DRY	*Rattus rattus rattus **	0.49 ± 0.11	4	22.53	0.29-0.8	13.75 ± 1.55	USP	0.89	32.15
DRY	*Cremnomys blanfordi*	10.73 ± 0.92	89	8.62	9.05-12.72	13.83 ± 0.56	USP	0.44	244.12
DRY	*Millardia meltada**	0.36 ± 0.08	3	22.53	0.22-0.6	13.75 ± 1.55	USP	0.89	32.15
DRY	*Mus platythrix*	41.91 ± 9.09	48	21.70	27.21-64.55	5.14 ± 0.55	HNC	0.69	115.29
DRY	*Mus booduga*	2.67 ± 0.42	19	15.77	1.92-3.71	12.8 ± 1.00	HNC	0.49	53.78
DRY	*Suncus montanus*	2.53 ± 0.42	21	16.73	1.79-3.58	13.83 ± 1.15	HNC	0.15	63.90
WET	*Funambulus palmarum*	3.68 ± 0.16	23	4.49	3.36-4.04	14.24 ± 0.30	USP	0.09	68.58
WET	*Tatera indica**	1.69 ± 0.19	11	11.81	1.33- 2.15	14.55 ± 0.78	USP	0.29	66.13
WET	*Rattus rattus wroughtoni*	11.65 ± 1.10	71	9.45	9.66-14.06	14.07 ± 0.64	USP	0.56	195.5
WET	*Rattus rattus rattus**	0.49 ± 0.27	4	53.17	0.15-1.55	16.25 ± 4.3	HNH	0.97	25.65
WET	*Cremnomys blanfordi*	18.57 ± 3.02	74	16.27	13.46-25.62	11.38 ± 0.91	HNSP	0.20	205.53
WET	*Millardia meltada **	0.77 ± 0.08	5	10.82	0.61- 0.96	14.55 ± 0.78	USP	0.29	66.13
WET	*Mus platythrix*	5.71 ± 1.51	17	26.50	3.29- 9.91	9.83 ± 1.30	HNSP	0.76	32.48
WET	*Mus musculus**	0.92 ± 0.10	6	11.02	0.73- 1.16	14.55 ± 0.78	USP	0.29	66.13
WET	*Mus booduga*	14.16 ± 2.81	29	19.85	9.47- 21.18	8.15 ± 0.80	HNC	0.40	73.94
WET	*Suncus montanus*	8.34 ± 2.15	17	25.83	4.86-14.29	8.14 ± 1.05	HNSP	0.69	44.38

D ± SE = Density (individuals/ha) and associated standard error; ERD = Effective radial distance (m); HNC = Half-normal cosine; USP = Uniform simple polynomial; HNSP = Half-normal simple polynomial; HNH = Half-normal hermite; * = Pooled species ERD was used to derive density

4.3.3 *Bird and hare abundance*

Though bird density was higher during the dry season in thorn forest than deciduous forest it was not significant (t test, t = 1.33, df = 3, P = 0.14), and did not differ between thorn and semi-evergreen forest (t test, t = 1.52, df = 3, P = 0.16, Table 12). Though bird density was higher during the wet season in the thorn forest than deciduous forest, it was not significant (t test, t = 1.58, df = 3, P = 0.10). Bird density did not differ between seasons in the thorn forest (t test, t = 1.45, df = 3, P = 0.12) and deciduous forest (t test, t = -0.24, df = 3, P = 0.41, Table 12). Overall there was no difference in bird abundance in dry and wet season (t test, t = 1.46, df =4, P = 0.10, Table 13).

Table 12. Hare and bird density estimates during the dry and wet season in Mudumalai Tiger Reserve (2009-2011).

Habitat	Species	D ± SE (ind/km^2)	No. of observations	% CV	95% Confidence Interval	ESW ± SE (m)	Model	P- value	AIC
DRY-DTF	Black-naped hare	2.31 ± 0.56	19	24.38	1.42-3.74	19.99 ± 5.34	HNC	0.49	54.68
DRY-DTF	Peafowl	3.19 ± 1.62	18	50.61	1.23- 8.30	22.1 ± 3.23	HNSP	0.16	39.25
DRY-DTF	Grey jungle fowl	3.75 ± 1.62	10	43.17	1.61- 8.79	16.1 ± 2.69	HNSP	0.25	50.24
DRY-DTF	Red spur fowl	3.83 ± 2.92	12	60.61	1.56-14.84	13.29 ± 1.33	HNSP	0.06	32.44
DRY-DTF	Grey francolin	3.86 ± 1.46	14	38.02	1.85- 8.07	16.1 ± 5.28	HNSP	0.25	40.69
DRY-DTF	Quail	20.23 ± 7.74	18	38.3	9.54- 42.86	16.34 ± 4.23	UC	0.38	60.45
DRY-DEC	Black-naped hare	2.62 ± 0.99	10	37.90	1.26- 5.46	13.1 ± 1.37	HNC	0.15	27.37
DRY-DEC	Peafowl	1.93 ± 0.75	14	38.95	0.90- 4.11	22.32 ± 5.93	HNC	0.20	34.53
DRY-DEC	Grey jungle fowl	1.63 ± 0.57	12	35.15	0.82-3.22	19.16 ± 2.69	HNC	0.05	19.55
DRY-DEC	Red spur fowl*	2.90 ± 1.57	8	53.99	1.05-8.01	16.41 ± 1.31	HNSP	0.28	27.2
DRY-DEC	Quail*	1.47 ± 0.52	5	35.17	0.74-2.92	16.41 ± 1.31	HNSP	0.28	27.2
DRY-SEM	Peafowl*	1.14 ± 0.70	4	61.70	0.34-3.76	21.51 ± 2.42	HNC	0.08	30.62
DRY-SEM	Grey jungle fowl*	6.34 ± 1.65	25	26.1	3.8-10.59	21.51 ± 2.42	HNC	0.09	30.62
DRY-SEM	Red spur fowl*	0.57 ± 0.33	4	58.62	0.18-1.75	21.51 ± 2.42	HNC	0.09	30.62
DRY-SEM	Quail*	0.56 ± 0.32	3	60.23	1.12-11.56	21.51 ± 2.42	HNC	0.13	30.62
WET-DTF	Black-naped hare	4.87 ± 1.95	14	39.96	2.25-10.52	10.24 ± 1.22	HNC	0.26	31.35
WET-DTF	Peafowl	1.33 ± 0.39	16	29.08	0.75-2.35	15.31 ± 6.36	UH	0.19	53.43
WET-DTF	Grey jungle fowl	4.12 ± 1.59	12	38.60	1.95- 8.68	15.61 ± 2.19	HNSP	0.17	35.86
WET-DTF	Red spur fowl*	0.87 ± 0.5	5	58.04	0.29-2.57	18.89 ± 2.68	HNC	0.13	28.69
WET-DTF	Grey francolin*	2.35 ± 0.89	8	37.88	1.13- 4.89	18.89 ± 2.68	HNC	0.07	28.69
WET-DTF	Quail	6.68 ± 2.27	16	33.99	3.45-12.93	15.29 ± 1.82	HNSP	0.06	46.10
WET-DEC	Black-naped hare	4.06 ± 1.73	11	42.49	1.78- 9.25	8.91 ± 2.24	HNC	0.05	21.42
WET-DEC	Peafowl	1.65 ± 0.66	12	39.85	0.76- 3.58	19.99 ± 8.81	HNC	0.26	55.75

WET-DEC	Grey jungle fowl	1.84 ± 0.85	14	46.27	0.76-4.46	15 ± 9.12	HNC	0.06	52.17
WET-DEC	Quail*	0.58 ± 0.40	3	69.79	0.16-2.08	19.99 ± 8.81	HNC	0.26	55.75

DRY-DTF = Dry season in thorn forest; DRY-DEC = Dry season in deciduous forest; DRY-SEM = Dry season in semi-evergreen forest WET-DTF = Wet season in thorn forest; WET-DEC = Wet season in deciduous forest; D ± SE = Density (individuals/km^2) and associated standard error; ESW = Effective strip width (m); AIC = Akaike information criterion; HNC = Half-normal cosine; UC = Uniform cosine; UH = Uniform hermite; HNSP = Half-normal simple polynomial; * = Pooled species ESW was used to derive density

Table 13. Overall hare and bird density estimates in Mudumalai Tiger Reserve (2009-2011).

Season	**Species**	**D ± SE (ind/km^2)**	**No. of observations**	**% CV**	**95% Confidence Interval**	**ESW ± SE (m)**	**Model**	**P- value**	**AIC**
DRY	Black-naped hare	2.28 ± 0.64	29	27.99	1.32-3.93	20.85 ± 4.04	HH	0.06	94.57
DRY	Peafowl	1.18 ± 0.54	36	45.52	0.49-2.84	21.51 ± 2.76	HNC	0.14	100.45
DRY	Grey jungle fowl	6.86 ± 0.98	50	14.36	5.16-9.13	16.02 ± 3.49	HNC	0.53	87.08
DRY	Red spur fowl	3.44 ± 0.87	24	25.40	2.09-5.64	18.99 ± 3.13	HNC	0.34	55.93
DRY	Grey francolin	0.84 ± 0.15	14	38.19	0.59-1.20	18.98 ± 6.11	HNC	0.09	40.6
DRY	Quail	4.06 ± 1.33	27	32.81	2.16-7.64	18 ± 3.34	HNSP	0.24	86.38
WET	Black-naped hare	4.17 ± 1.15	25	27.72	2.44-7.16	10.29 ±1.01	HNSP	0.07	51.39
WET	Peafowl	1.70 ± 0.49	26	29.05	0.97-2.99	21.98 ± 1.16	HNC	0.4	35.27
WET	Grey jungle fowl	2.66 ± 0.82	27	30.81	1.46-4.84	14.46 ± 4.5	HNC	0.06	100.2
WET	Red spur fowl*	0.23 ± 0.12	4	53.04	0.08-0.63	10.45 ± 4.22	UC	0.26	24.18
WET	Grey francolin*	1.69 ± 0.61	8	36.31	0.84-3.4	10.45 ± 4.22	UC	0.26	24.18
WET	Quail*	2.81 ± 0.79	19	28.22	1.62-4.87	10.45 ± 4.22	UC	0.26	24.18

D ± SE = Density (individuals/km^2) and associated standard error; ESW = Effective strip width (m); AIC = Akaike information criterion; HH = Half-normal hermite; HNC = Half-normal cosine; UC = Uniform cosine; HNSP = Half-normal simple polynomial; * = Pooled species ESW was used to derive density.

4.3.4 *Adequacy of sample size*

A total of 355 faeces of small cats were collected from dry thorn (n = 258) and deciduous (n = 97) forests. The number of faeces collected in each month ranged from four to 97 (Fig 11). A total of 467 food items were detected in faeces; the mean ± SD number of food items per fecal sample was 1.34 ± 0.79 (range = 1-4). Totally seven species of small mammals, bird, snake, calotes, skink, fish, crustacean, pilli millipede, dragonfly, dung beetle, scorpion, termite, cricket, reptile eggs, three species of fruits, and grass were recorded in small cat scats (Table 8). The number of species appeared to reach an asymptote at about 80 samples in dry thorn and deciduous forests (Fig 12).

Fig 11. Number of small cat faecal samples collected across months during the study period (2009-2011) in Mudumalai Tiger Reserve.

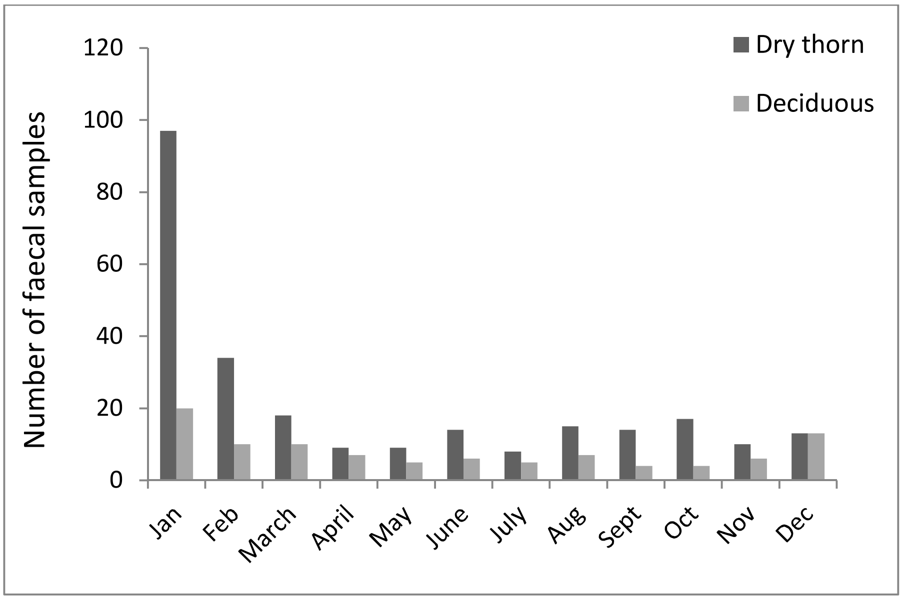
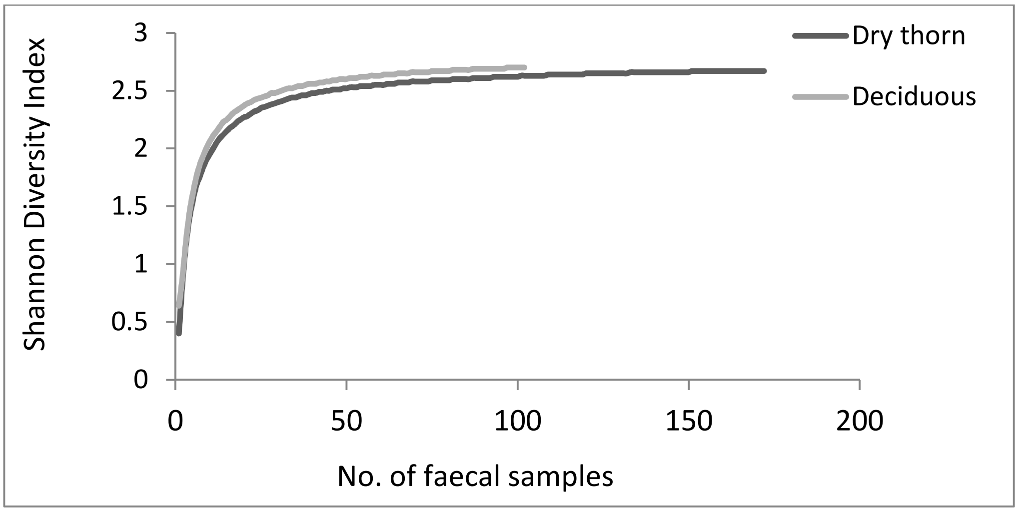


Fig 12. Cumulative dietary diversity indexed by the Shannon diversity index, for small cat against increasing number of faecal samples for the years 2009–2011 in Mudumalai Tiger Reserve.



A total of 831 civet faeces were collected during the two year study from three habitats; dry thorn (n = 205), deciduous (n = 320) and semi-evergreen (n = 306) forests. The number of faeces collected in each month varied between five and 127 (Fig 13). A total of 1431 food items was detected in civet feces; the mean ± SD number of food items per fecal sample were 1.59 ± 1.02 (range = 1-6). Totally 31 species of fruits, five species of small mammals, bird, snake, calotes, skink, mollusks, arthropods, herbs and grass were recorded in civet faeces (Table 9). The number of species appeared to reach an asymptote at about 80 samples in dry thorn, 55 in deciduous and 97 in semi-evergreen forests (Fig 14).

Fig 13. Civet faecal samples collected on a monthly basis during the study period in Mudumalai Tiger Reserve (2009-2011).

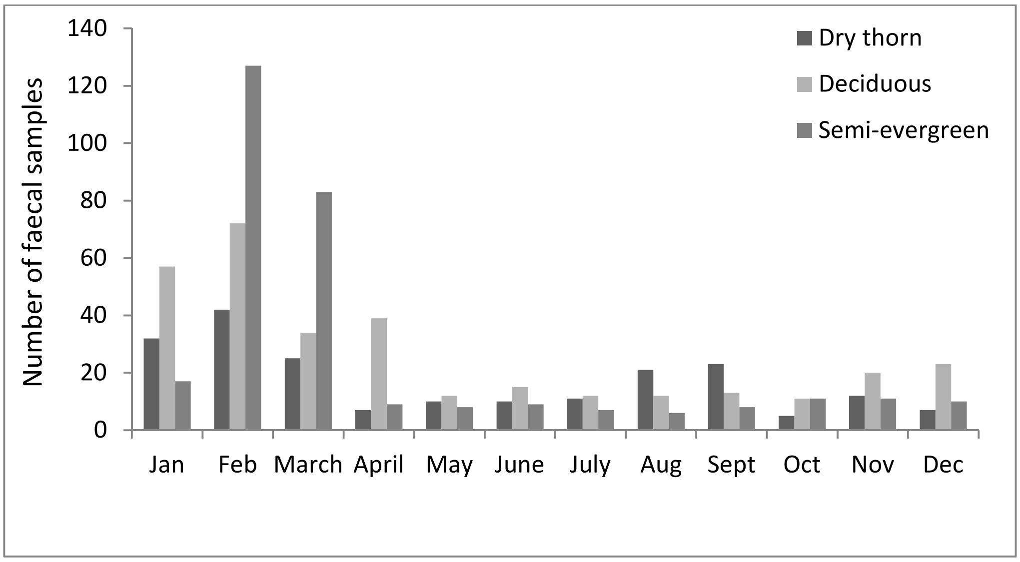
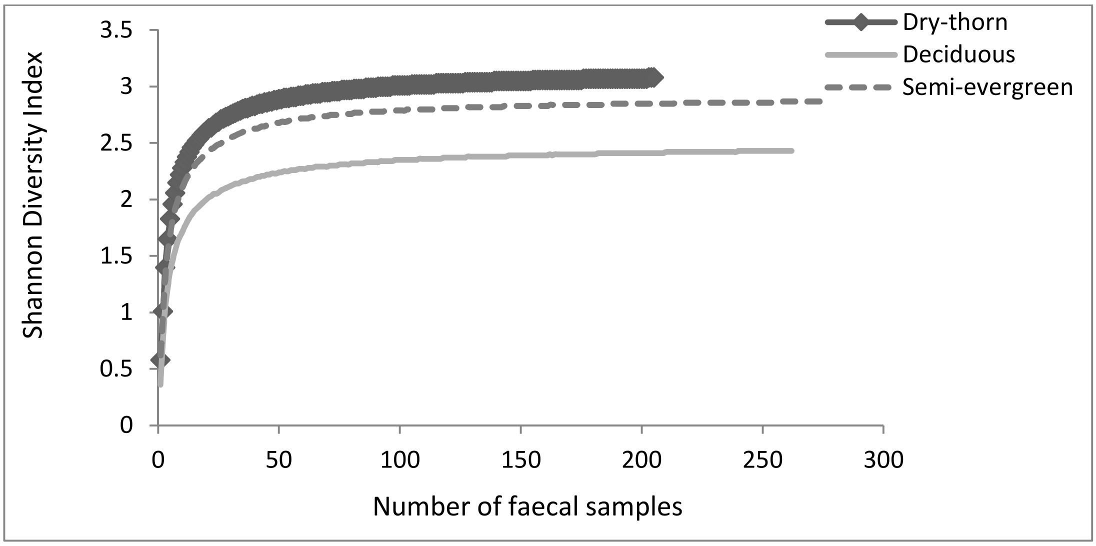


Fig 14. Cumulative dietary diversity indexed by the Shannon diversity, for civet against increasing number of faecal samples in Mudumalai Tiger Reserve (2009–2011).



A total of 708 mongoose faeces were collected during the two year study from three habitats; dry thorn (n = 281), deciduous (n = 333) and semi-evergreen (n = 94) forests. The number of faeces collected in each month varied between four and 85 (Fig 15). A total of 1318 food items were detected in mongoose feces; the mean ± SD number of food items per fecal sample was 1.89 ± 1.16 (range = 1-5). Totally eight species of fruits, seven species of small mammals, bird, snake, calotes, skink, fish, mollusks, arthropods, reptile eggs, and vegetative matter were recorded in mongoose faeces (Table 10). The number of species appeared to reach an asymptote at about 70 samples in dry thorn and deciduous forest while it almost stabilized in semi-evergreen forests probably due to the small sample size (Fig 16).

Fig 15. Mongoose faecal samples collected on a monthly basis during the study period in Mudumalai Tiger Reserve (2009-2011).

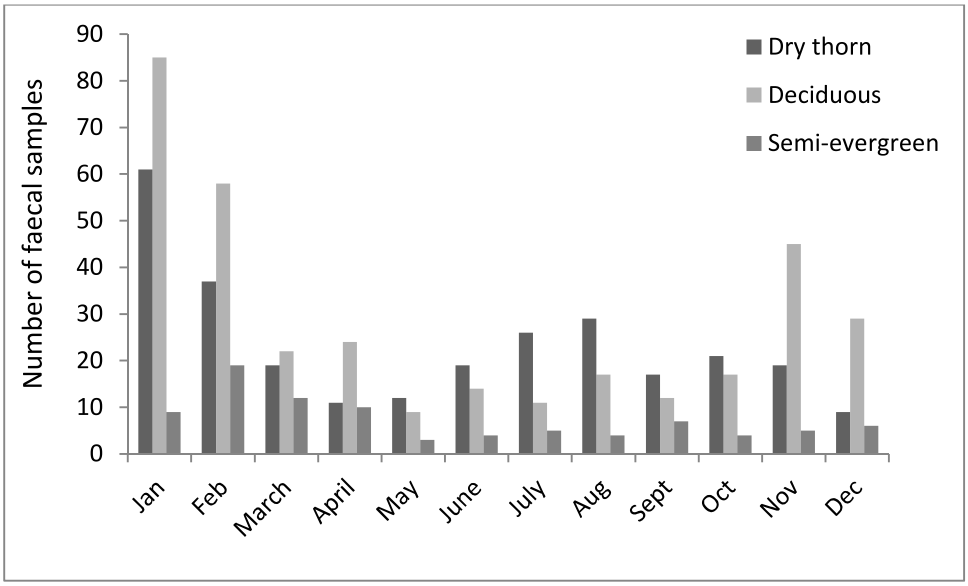
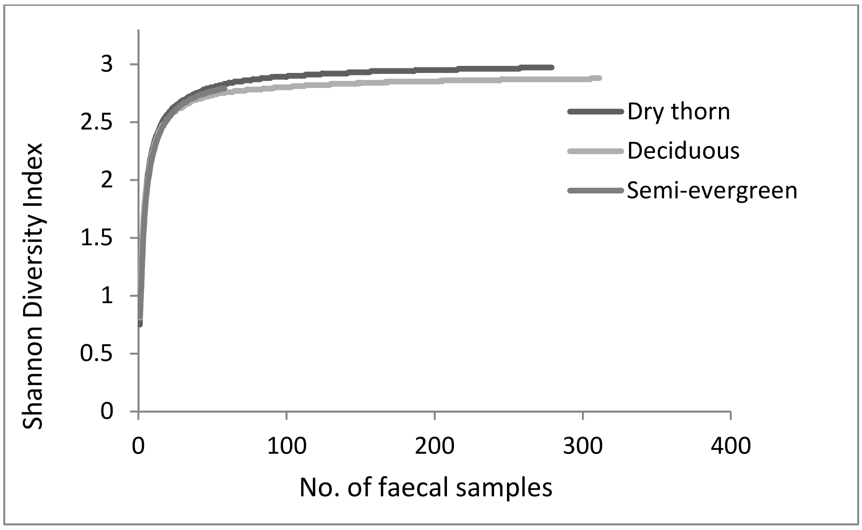


Fig 16. Cumulative dietary diversity indexed by the Shannon diversity index, for mongoose against increasing number of faecal samples for the years 2009–2011 in Mudumalai Tiger Reserve.



4.3.5 *Diet composition of small cat*

Animal prey included hare, rodents, shrew, birds, reptiles, crustaceans, pilli millipedes, scorpions, four orders of insects, reptile eggs, three species of fruits, grass and carrion (Table 14). Small mammals were the most common food items, accounting for more than 50.73% of the materials in fecal samples and 82.87% of the ingested biomass followed by invertebrates that accounted for about 21.29% of the materials in fecal samples and 1.66% of the ingested biomass. Bird represented 11.9% of the materials in fecal samples and 8.67% of ingested biomass. Reptiles formed 4.8% of materials and 6.54% of ingested biomass. Fish represented poorly in their diet forming 0.84% of the materials and 0.10% of ingested biomass while fruit represented 2.30% of the materials and 0.17% of ingested biomass.

Table 14. Diet composition of small cat based on fecal analysis in Mudumalai Tiger Reserve (2009-2011).

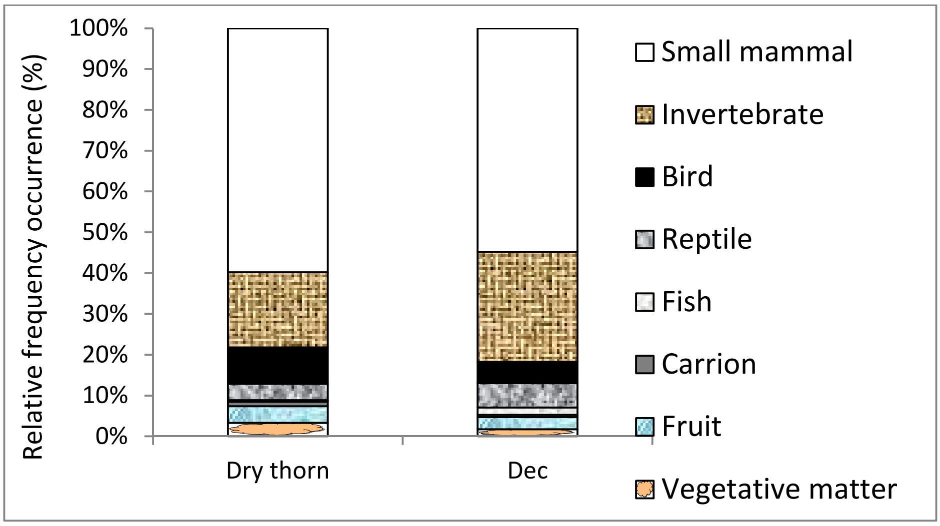
Food item	**weight (g)**	**n**	**PO**	**RFO**	**PB**
Small mammal		**243**	**68.45**	**50.73**	**82.87**
Lepus nigricollis	2000	9	2.54	1.88	13.41
Unidentified rodent$^{\$}$	70	47	13.24	9.81	16.34
Rattus rattus	97	22	6.20	4.59	10.60
Mus spp	5	73	20.56	15.24	1.81
Cremnomys blanfordi	101	52	14.65	10.86	26.09
Tatera indica	128	20	5.63	4.18	12.72
Millardia meltada	30	9	2.54	1.88	1.34
Suncus montanus	10	11	3.10	2.30	0.55
Carrion		**4**	**1.13**	**0.84**	
Rusa unicolor		1	0.28	0.21	
Axis axis		3	0.85	0.63	
Bird		**57**	**16.06**	**11.90**	**8.67**
Unidentified bird	30	50	14.08	10.44	7.45
Quail	35	7	1.97	1.46	1.22
Reptile		**19**	**5.35**	**4.80**	**6.54**
Calotes spp	10	11	3.10	2.30	0.55

Snake	200	6	1.69	1.25	5.96
Skink	3	2	0.56	0.42	0.03
Fish	**5**	**4**	**1.13**	**0.84**	**0.10**
Invertebrate		**102**	**28.73**	**21.29**	**1.66**
Crustacean	26	6	1.69	1.25	0.77
Pilli millipede	2	35	9.86	7.31	0.35
Dragonfly	0.5	3	0.85	0.63	0.01
Dung beetle	2	50	14.08	10.44	0.50
Scorpion	1	4	1.13	0.84	0.02
Termite	0.2	2	0.56	0.42	0.00
Cricket	0.3	1	0.28	0.21	0.00
Unknown insect	1	1	0.28	0.21	0.00
Reptile eggs	1.3	1	0.28	0.21	0.01
Fruit		**11**	**3.10**	**2.30**	**0.17**
Unidentified fruit	3	1	0.28	0.21	0.01
Lantana camara	1.5	1	0.28	0.21	0.01
Zizyphus oenophilia	3.5	7	1.97	1.46	0.12
Cordia spp.	3	2	0.56	0.42	0.03
Grass	**0**	**38**	**10.70**	**7.93**	**0.00**

n = sample size, PO = Percentage occurrence, RFO = relative frequency occurrence, PB = percentage biomass ingested, $^{\$}$ = Average weight calculated from live trapped rodents.

The frequency occurrence of major food items did not differ between habitats (Contingency table analysis, $\chi^2 = 6.26$, df = 7, P = 0.51, Fig 17).

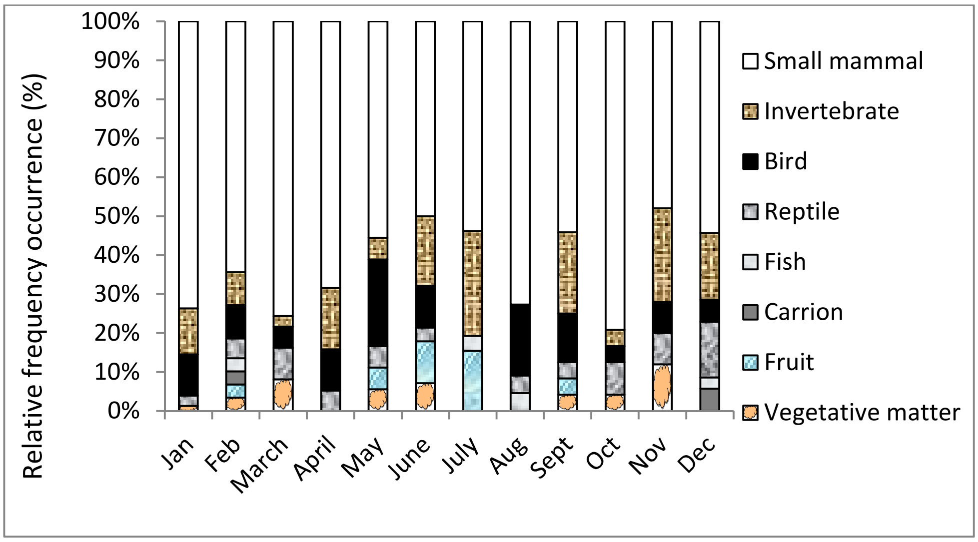
Fig 17. Relative frequency occurrence of food items of small cat in the dry thorn and deciduous forests (2009-2011).



Monthly relative frequency occurrence of small mammals was significantly higher than other food items (Wilcoxon paired sample test, z = -3.06, P = 0.002 respectively, Fig 18). Although invertebrate occurrence was higher than reptile, it was not significant (Wilcoxon paired sample test, z = -1.86, P = 0.06) but it was higher than fish (Wilcoxon paired sample test, z = -2.82, P = 0.005), carrion (Wilcoxin paired sample test, z = -2.93, P = 0.003) and fruit (Wilcoxon paired sample test, z = -2.8, P = 0.005). There was no difference between bird and invertebrate remains (Wilcoxon paired sample test, z = -0.86, P = 0.38).

Fig 18. Monthly variation in the main food types ingested by small cat (2009-2011); a) overall b) dry thorn c) deciduous forests.

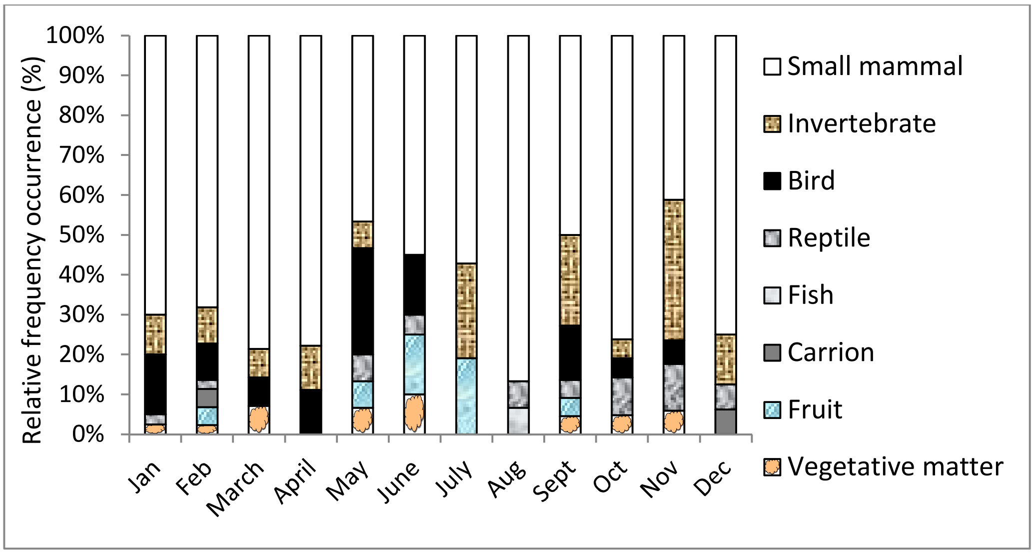
a)



Food types are expressed as relative percentage occurrence.

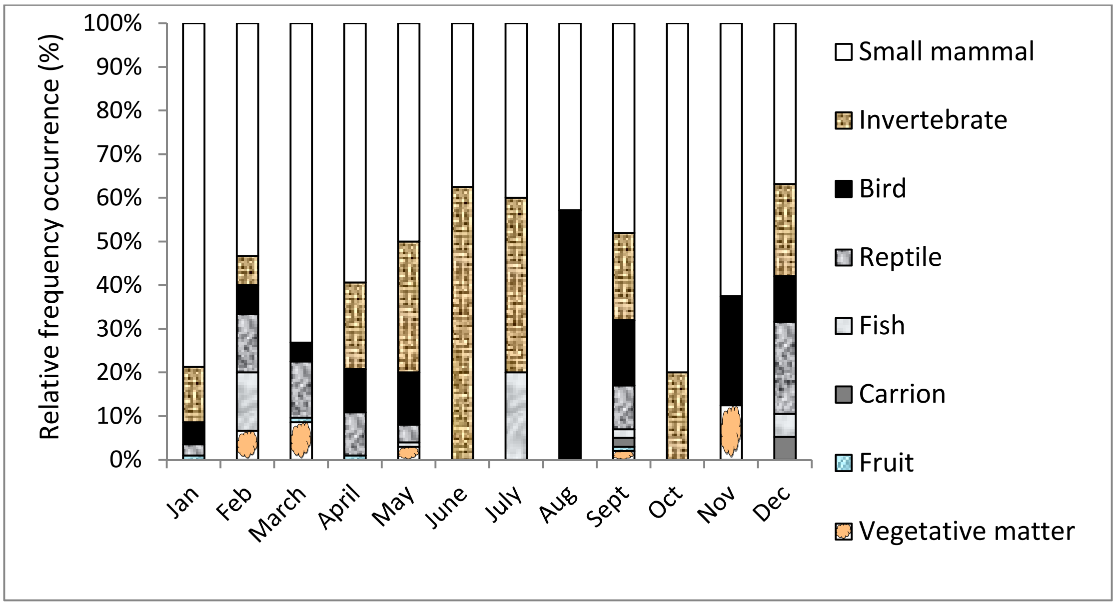
Small mammal and invertebrate remains occurred throughout the year in both forest types. However in the deciduous forest, invertebrate remains occurred highest in June and bird remains occurred highest in August. Fruits and carrion occurred rarely in the diet in both habitats (Fig 18b, c).

b)



Food types are expressed as relative percentage occurrence.

c)



Food types are expressed as relative percentage occurrence.

4.3.6 *Diet composition of civet*

Animal prey included hare, rodents, shrew, birds, reptiles, mollusks, crustaceans, pilli millipedes, millipedes, scorpion, at least five different orders of insects, 31 species of fruits and vegetative matter (Table 15). Fruits were the most common food items, accounting for more than 69.25% of the materials in fecal samples and 33.63% of the ingested biomass followed by invertebrates that accounted for about 16.77% of the materials in fecal samples and 5.91% of the ingested biomass. Small mammals, represented 6.85% of the materials in fecal samples and 28.79% of ingested biomass, respectively. Reptiles formed about 3.07% of materials and 14.93% of ingested biomass. Bird represented poorly in their diet forming 2.52% of the materials and 16.73% of ingested biomass.

From the 31 plant species (excluding unidentified species) consumed by civets, cultivated plants like *Coffee spp* and *Caryota urens*, were also found in the diet. Based upon the percent occurrence (≥ 1%) , the key plant species of the civet's diet were *Grewia tiliifolia*, *Cassia fistula, Zizyphus oenophilia*, *Lantana camara*, *Cordia spp*, *Antidesma zeylanicum*, *Coffee spp*, *Embelia gardneriana*, *Bischovia javanica*, *Gnetum ula*, *Ficus spp*, *Symplocos cochinchinensis*, *Bridelia crenulata*, *Syzigium cumini*, *Caryota urens*, *Piper nigrum*, *Dillenia pentagyna*, *Diospyros montana*, *Zizyphus* *mauritiana*, and *Grewia tiliifolia*. The proportion of small sized fruits (0.3-3.5 cm) was high in the diet except for *Cassia fistula* fruits. The fruits chosen were pulpy, fleshy, berries and drupes.

Table 15. Diet composition of civets based on fecal analysis in Mudumalai Tiger Reserve (2009-2011).

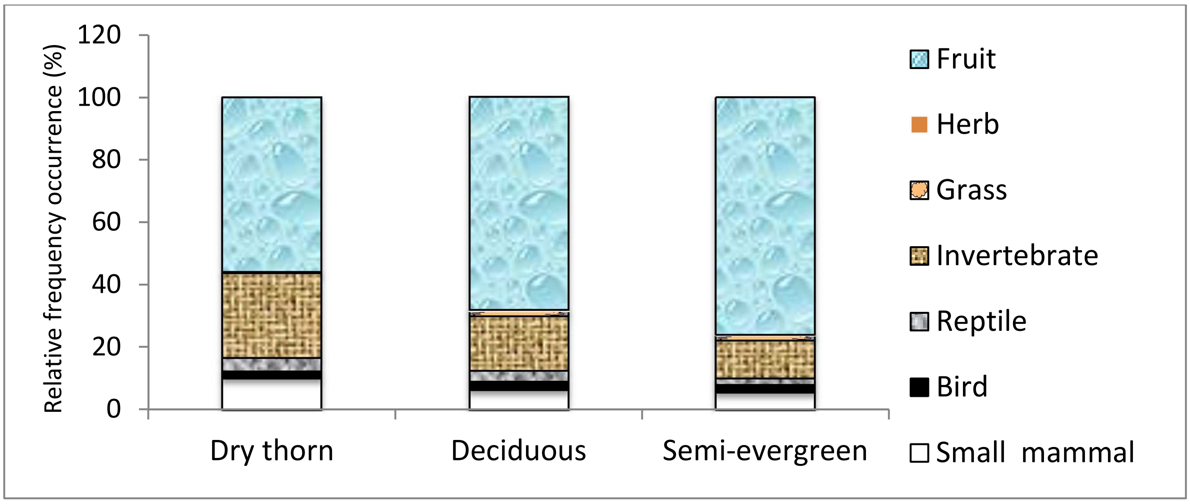
Food item	n	weight (g)	PO	RFO	PB
Small mammal	**98**		**11.79**	**6.85**	**28.79**
Lepus nigricollis	2	2000	0.24	0.14	2.48
Unidentified rodent$^{\$}$	50	70	6.02	3.49	14.46
Cremnomys blanfordi	11	101	1.32	0.77	4.59
Rattus spp.	17	97	2.05	1.19	6.81
Mus spp.	14	5	1.68	0.98	0.29
Suncus montanus	4	10	0.48	0.28	0.17
Bird	**36**		**4.33**	**2.52**	**16.73**
Unidentified bird	35	30	4.21	2.45	4.34
Pavo cristatus	1	3000	0.12	0.07	12.39
Reptile	**44**		**5.29**	**3.07**	**14.93**
Calotes spp.	19	10	2.29	1.33	0.78
Snake	17	200	2.05	1.19	14.04
Skink	8	3	0.96	0.56	0.10
Invertebrates	**240**		**28.88**	**16.77**	**5.91**
Mollusc	10	3.5	1.20	0.70	0.14
Crustacean	41	26	4.93	2.87	4.40
Pilli millipede	65	2	7.82	4.54	0.54
Dung beetle	65	2	7.82	4.54	0.54
Red ant	1	0.1	0.12	0.07	0.00
Scorpion	3	1	0.36	0.21	0.01
Termite	4	0.2	0.48	0.28	0.00
Grasshopper	4	0.4	0.48	0.28	0.01
Unknown insect	10	1	1.20	0.70	0.04
Millipede	37	1.5	4.45	2.59	0.23
Fruit	**991**		**119.25**	**69.25**	**33.63**
Grewia tiliifolia	353	5.24	42.48	24.67	7.64
Cassia fistula	155	30	18.65	10.83	19.21

Zizyphus oenophilia	66	3.5	7.94	4.61	0.95
Lantana camara	35	1.5	4.21	2.45	0.22
Cordia obliqua	34	3	4.09	2.38	0.42
Antidesma zeylanicum	30	2	3.61	2.10	0.25
Cordia spp	28	3	3.37	1.96	0.35
Coffee spp	27	2.5	3.25	1.89	0.28
Embelia gardneriana	20	0.7	2.41	1.40	0.06
Bischovia javanica	20	3.5	2.41	1.40	0.29
Gnetum ula	23	11	2.77	1.61	1.05
Ficus spp	19	6	2.29	1.33	0.47
Symplocos cochinchinensis	16	1.2	1.93	1.12	0.08
Bridelia crenulata	15	0.4	1.81	1.05	0.02
Syzigium cumini	14	3.8	1.68	0.98	0.22
Caryota urens	12	7	1.44	0.84	0.35
Piper nigrum	12	1.4	1.44	0.84	0.07
Dillenia pentagyna	12	4	1.44	0.84	0.20
Diospyros montana	10	15	1.20	0.70	0.62
Zizyphus mauritiana	10	4	1.20	0.70	0.17
Grewia hirsuta	9	0.7	1.08	0.63	0.03
Mimusops elengi	7	5	0.84	0.49	0.14
Elaeocarpus variabilis	7	3.5	0.84	0.49	0.10
Tamarandus indicus	6	1.3	0.72	0.42	0.03
Flacourtia indica	5	1.5	0.60	0.35	0.03
Erythroxylum monogynum	4	0.4	0.48	0.28	0.01
Phoenix iourerii	2	0.6	0.24	0.14	0.00
Breynia retusa	1	0.4	0.12	0.07	0.00
Cucurbitaceae	1	0	0.12	0.07	0.00
Polyalthia cerasoides	1	1	0.12	0.07	0.00
Naringi crenulata	1	2	0.12	0.07	0.01
Unidentified fruit	30	3	3.61	2.10	0.37
Unknown plant	6	0	0.72	0.42	
Vegetative matter	**22**	**0**	**2.65**	**1.54**	

n = sample size, PO = Percentage occurrence, RFO = relative frequency occurrence, PB = percentage biomass ingested, $^{\$}$ = Average weight calculated from live trapped rodents.

There was no difference in relative frequency occurrence of food items between dry thorn and deciduous (Contingency table analysis, $\chi^2 = 5.53$, df = 6, P = 0.47), between deciduous and semi-evergreen (Contingency table analysis, $\chi^2 = 1.7$, df = 6, P = 0.94), while there was significant difference between dry thorn and semi-evergreen forest (Contingency table analysis, $\chi^2 = 11.94$, df = 5, P = 0.03, Fig 19).

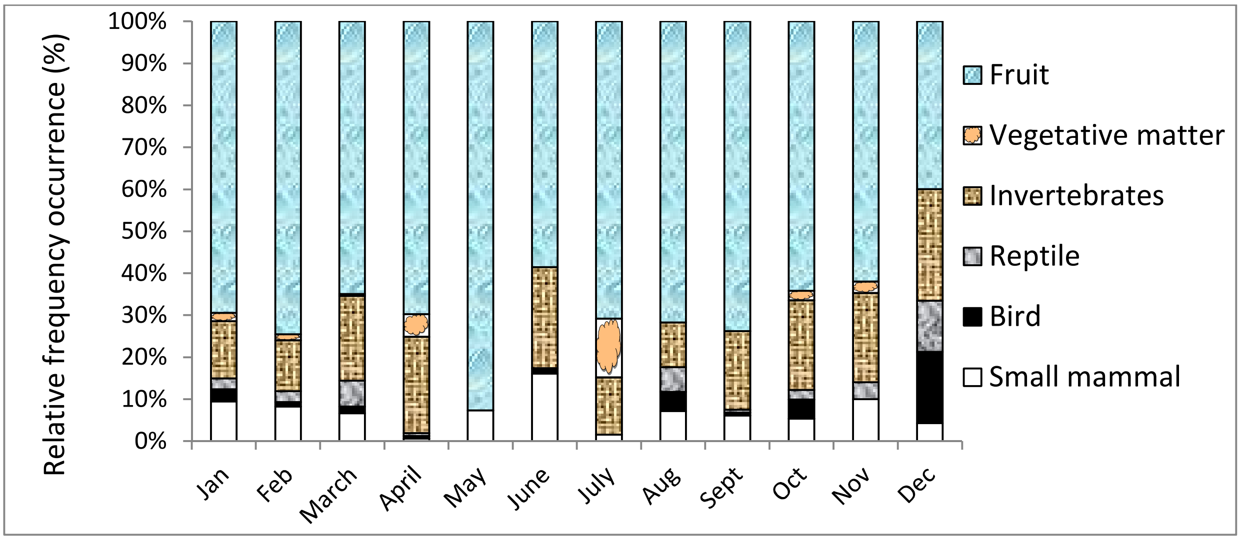
Fig 19.Variation in the relative frequency occurrence of food items of the civet in the dry thorn, deciduous and semi-evergreen forests (2009-2011).



Civet diet was characterized by marked variation over time for most of the food items (Fig 20 a, b, c, d). Overall, fruits, small mammals and invertebrates were eaten throughout the year (Fig 20a), however fruits occurred highest in May and lowest in December. Bird remains were found maximum in December. Fruits were consumed significantly higher than other food items (Wilcoxon paired sample test, z = -3.06, P = 0.002 respectively, Fig 20a). Invertebrates occurred higher than reptiles, birds (Wilcoxon paired sample test, z = -2.93, P = 0.003 respectively), vegetative remains (Wilcoxon paired sample test, z = -2.84, P = 0.004), and small mammals (Wilcoxon paired sample test, z = -2.75, P = 0.006).

Fig 20. Monthly variation in food items ingested by civet (2009-2011); a) overall b) dry thorn c) deciduous d) semi-evergreen forests.

a)

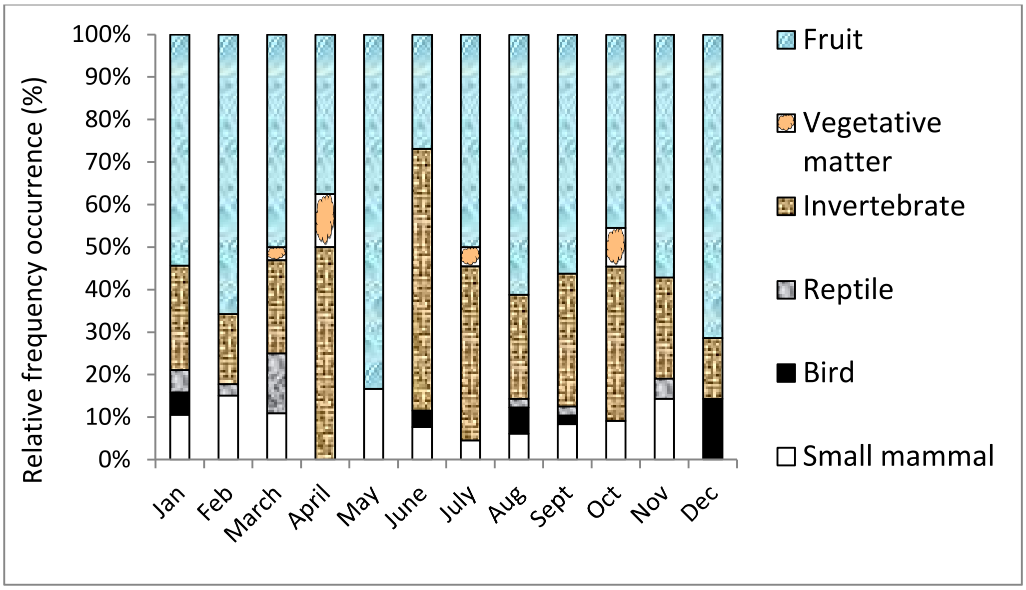


Food types are expressed as relative percentage occurrence

Small mammal, fruit and invertebrate occurred throughout the year in dry thorn and deciduous forest. In the thorn forest, fruits were consumed maximum in May and December, small mammal occurrence was consistent across months, invertebrate remains were highest in June while bird remains were highest in December, and reptile remains were highest in March (Fig 20b). In the deciduous forest only fruits were consumed maximum from May to July, bird remains were highest in October and reptile remains occurred highest in August (Fig 20c). In the semi-evergreen forest fruits occurred in all the months except December, small mammal occurrence was highest in June, only fruits were consumed in August and October, bird and reptile remains occurred maximum in December and vegetative remains were maximum in July during high rainfall (Fig 20d).

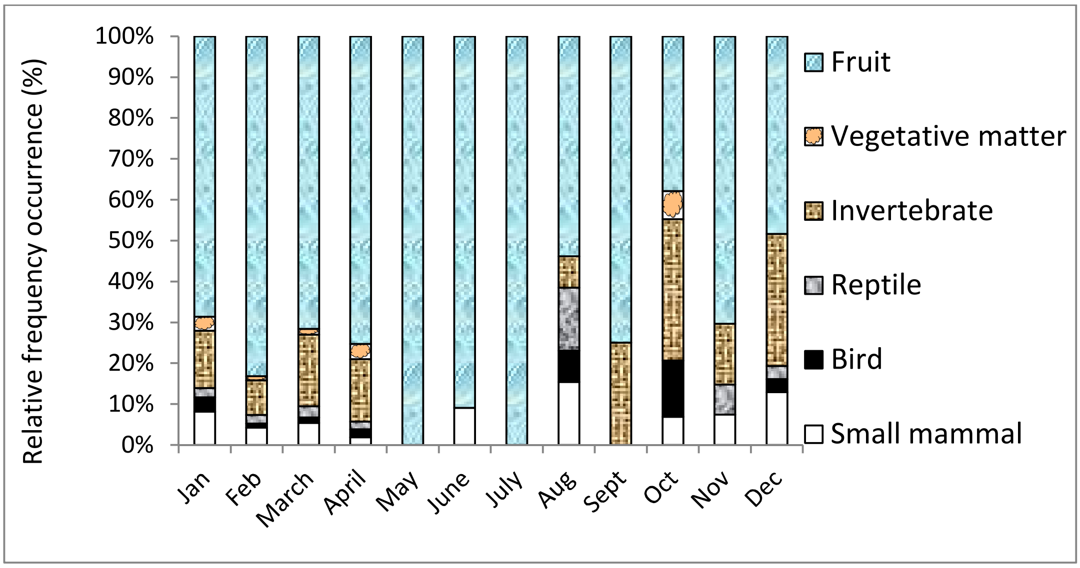
Monthly invertebrate remains occurred higher in deciduous forest than thorn forest (Wilcoxon paired sample test, z = -2.22, P = 0.02). Fruit occurrence was higher in deciduous forest than thorn forest (Wilcoxon paired sample test, z = -2.12, P = 0.02) while it was higher in semi-evergreen forest than thorn forest (Wilcoxon paired sample test, z = -2.12, P = 0.03).

b)



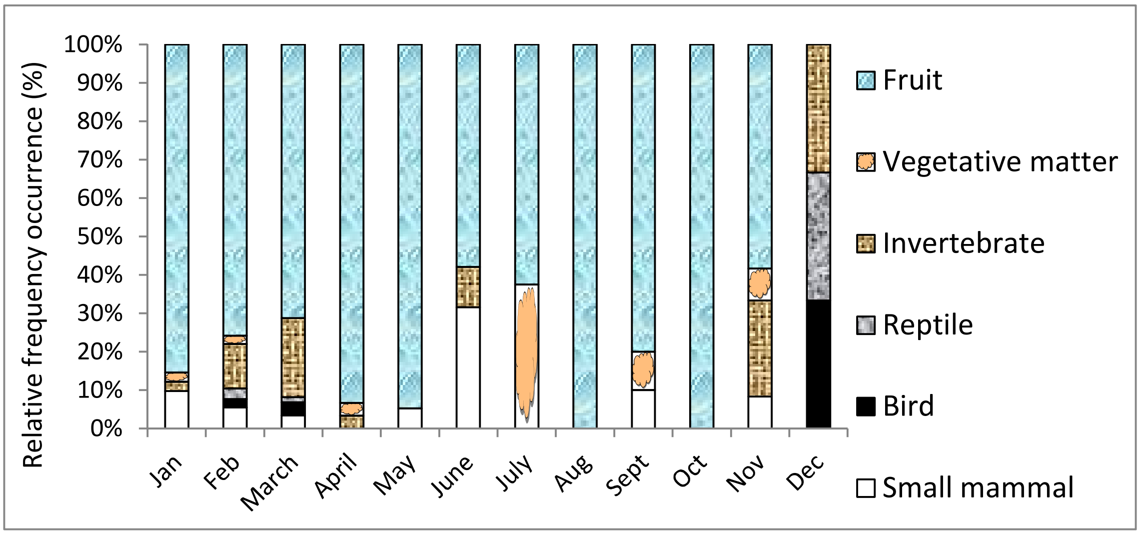
Food types are expressed as relative percentage occurrence

c)



Food types are expressed as relative percentage occurrence

d)



Food types are expressed as relative percentage occurrence

4.3.7 *Diet composition of mongoose*

Animal prey included hare, rodents, shrew, birds, reptiles, molluscs, crustaceans, pilli millipedes, millipedes, scorpion, at least five orders of insects, eight species of fruits, reptile egg and carrion (Table 16). Invertebrates were the most common food items, accounting for more than 48.49% of the materials in fecal samples and 8% of the ingested biomass followed by small mammal that accounted for about 18.29% of the materials in fecal samples and 56.38% of the ingested biomass. Fruit presented 16.01% of the materials and 2.63% of ingested biomass. Reptiles formed 8.12% of materials and 27.35% of ingested biomass while bird formed 3.56% of the materials in fecal samples and 5.22% of ingested biomass. Fish presented poorly in the diet accounting for 1.06% of the materials and 0.26% of ingested biomass.

Table 16. Diet composition of mongoose based on fecal analysis in Mudumalai Tiger Reserve (2009-2011).

Food item	**weight (g)**	**n**	**PO**	**RFO**	**PB**
Small mammal		**241**	**34.04**	**18.29**	**56.38**
Unidentified rodent$^{\$}$	70	84	11.86	6.37	21.60
Lepus nigricollis	2000	9	1.27	0.68	9.92
Mus spp	5	62	8.76	4.70	1.14
Cremnomys blanfordi	101	33	4.66	2.50	12.24

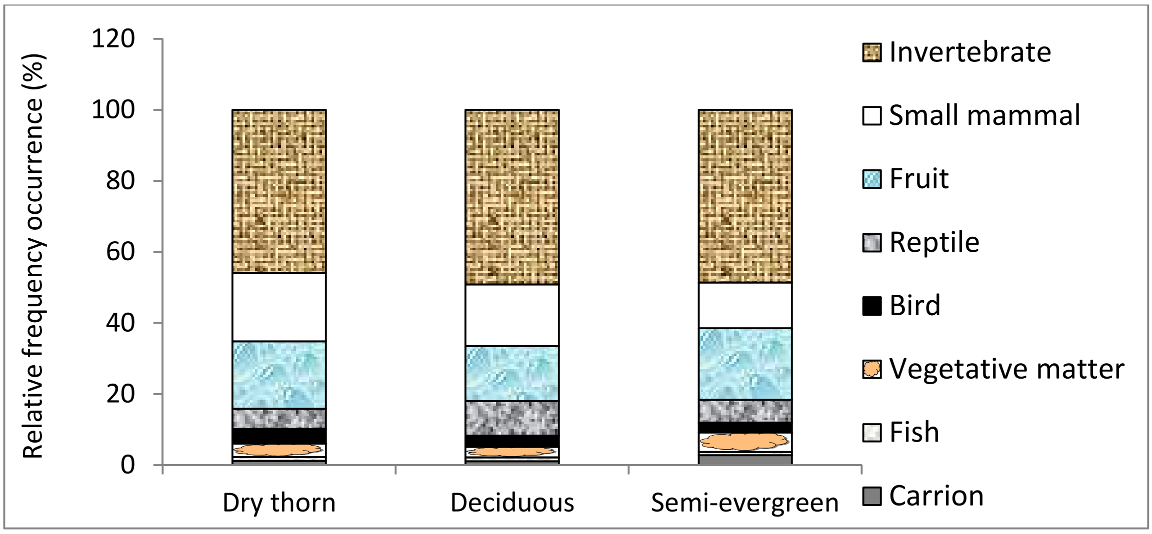
Rattus spp	97	28	3.95	2.12	9.98
Suncus montanus	10	22	3.11	1.67	0.81
Millardia meltada	30	2	0.28	0.15	0.22
Tatera indica	128	1	0.14	0.08	0.47
***Axis axis* (carrion)**	**0**	**15**	**2.12**	**1.14**	
Bird		**47**	**6.64**	**3.56**	**5.22**
Unidentified bird	30	45	6.36	3.41	4.96
Quail	35	2	0.28	0.15	0.26
Reptile		**107**	**15.11**	**8.12**	**27.35**
Calotes spp	10	61	8.62	4.63	2.24
Snake	200	34	4.80	2.58	24.98
Skink	3	12	1.69	0.91	0.13
Fish	**5**	**14**	**1.98**	**1.06**	**0.26**
Invertebrates		**639**	**90.25**	**48.49**	**8**
Mollusc	**3.5**	**8**	**1.13**	**0.61**	**0.10**
Crustacean	26	43	6.07	3.26	4.11
Pilli millipede	2	228	32.20	17.30	1.68
Millipede	1.5	35	4.94	2.66	0.19
Dragonfly	0.5	1	0.14	0.08	0.00
Dung beetle	2	233	32.91	17.68	1.71
Red ant	0.1	2	0.28	0.15	0.00
Scorpion	1	29	4.10	2.20	0.11
Cricket	0.3	9	1.27	0.68	0.01
Termite	0.2	24	3.39	1.82	0.02
Grasshopper	0.4	7	0.99	0.53	0.01
Moth	0.2	1	0.14	0.08	0.00
Unknown insect	1	19	2.68	1.44	0.07
Reptile egg	**1.3**	**1**	**0.14**	**0.08**	**0.00**
Fruit		**211**	**29.8**	**16.01**	**2.63**
Zizyphus oenophilia	3.5	65	9.18	4.93	0.84
Unidentified fruit	3	54	7.63	4.10	0.60
Grewia tiliifolia	5.24	38	5.37	2.88	0.73
Cordia obliqua	3	21	2.97	1.59	0.23
Lantana camara	1.5	17	2.40	1.29	0.09
Bridelia crenulata	0.4	8	1.13	0.61	0.01
Zizyphus mauritiana	4	3	0.42	0.23	0.04

Syzigium cumini	3.8	3	0.42	0.23	0.04
Ficus spp.	6	2	0.28	0.15	0.04
Vegetative matter		**43**	**6.07**	**3.26**	

n = sample size, PO = Percentage occurrence, RFO = relative frequency occurrence, PB = percentage biomass ingested, $^{\$}$ = Average weight calculated from live trapped rodents.

The frequency occurrence of major food items did not differ between dry thorn and deciduous forest (Contingency table analysis, $\chi^2 = 5.7$, df = 7, P = 0.57), between dry thorn and semi-evergreen (Contingency table analysis, $\chi^2 = 6.57$, df = 7, P = 0.47) and between deciduous and semi-evergreen forest (Contingency table analysis, $\chi^2 = 4.94$, df = 7, P = 0.55, Fig 21).

Figure 21. Relative frequency occurrence of food items of the mongoose in the dry thorn, deciduous and semi-evergreen forests.

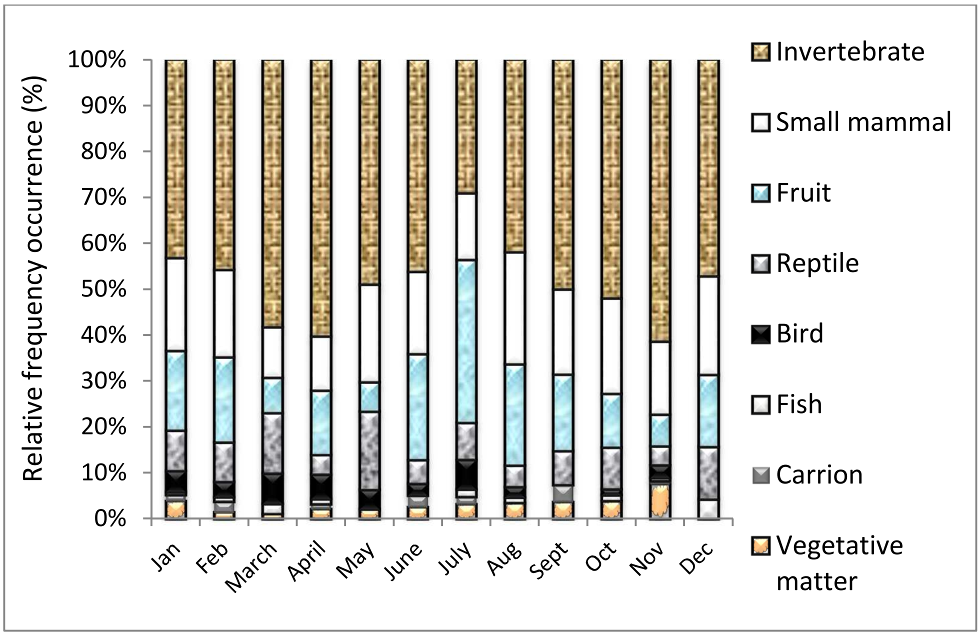


Invertebrates, small mammals and fruits were consumed throughout the year in the three habitats (Fig 22, b, c, d). Overall invertebrates were consumed significantly higher than small mammals, bird, reptile, fish, vegetative matter, carrion (Wilcoxon paired sample test, z = -3.06, P = 0.002 respectively) and fruit (Wilcoxon paired sample test, z = -2.98, P = 0.003). Small mammal remains occurred higher than birds, fish, vegetative matter, (Wilcoxon paired sample test, z = -3.06, P = 0.002) and reptile (Wilcoxon paired sample test, z = -2.98, P = 0.003, Fig 22a).

Monthly occurrence of bird remains was higher in thorn forest than semi-evergreen forest (Wilcoxon paired sample test, z = -2.29, P = 0.02). Reptile occurrence was higher in deciduous than thorn (Wilcoxon paired sample test, z = -2.12, P = 0.03) and semi-evergreen forest (Wilcoxon paired sample test, z = -2.62, P = 0.009).

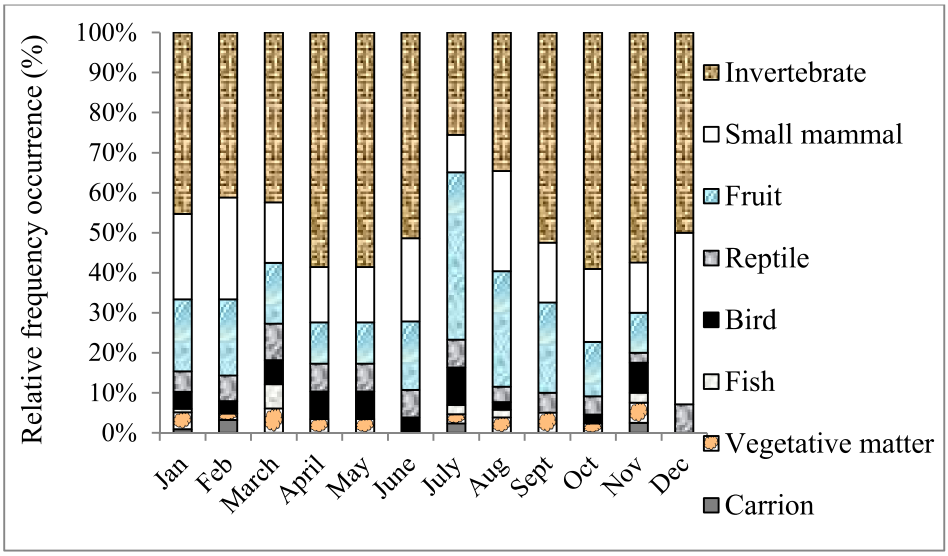
Fig 22. Monthly variations in the main food types ingested by mongoose; a) overall b) dry thorn c) deciduous d) semi-evergreen forests.

a)



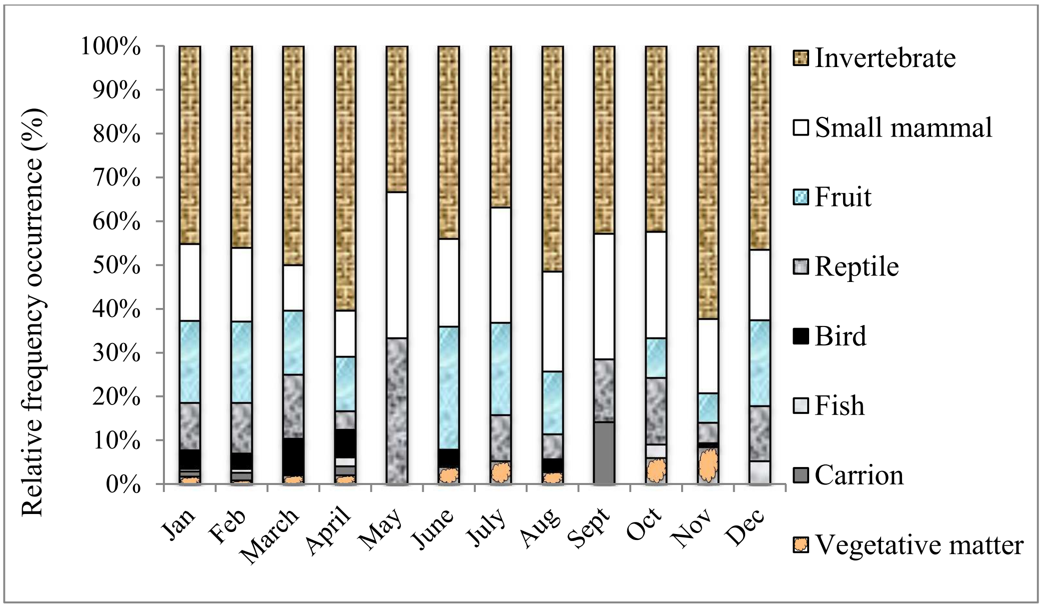
Food types are expressed as relative percentage occurrence

b)



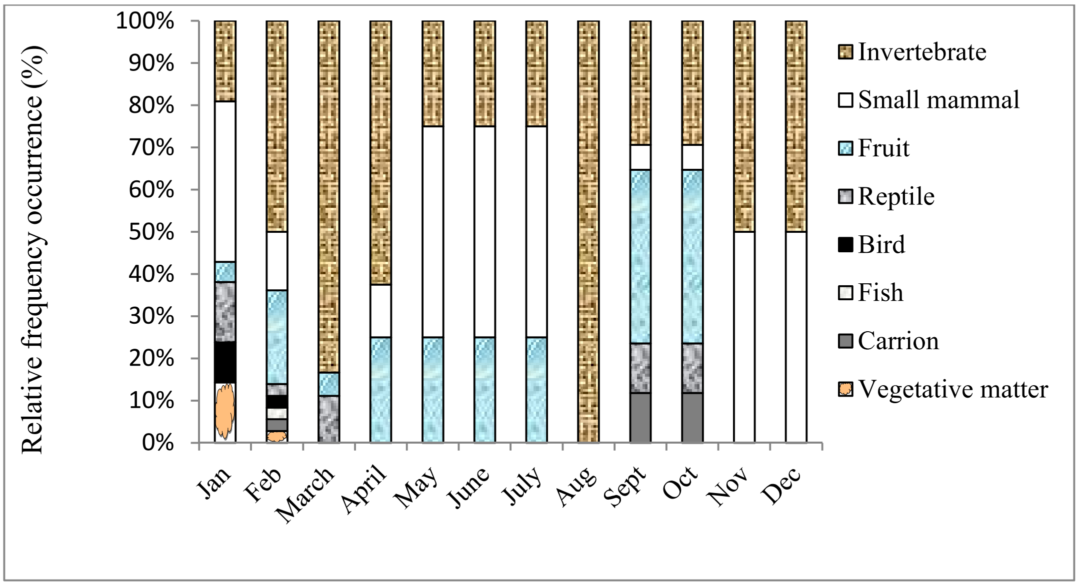
Food types are expressed as relative percentage occurrence

c)



Food types are expressed as relative percentage occurrence

d)



Food types are expressed as relative percentage occurrence

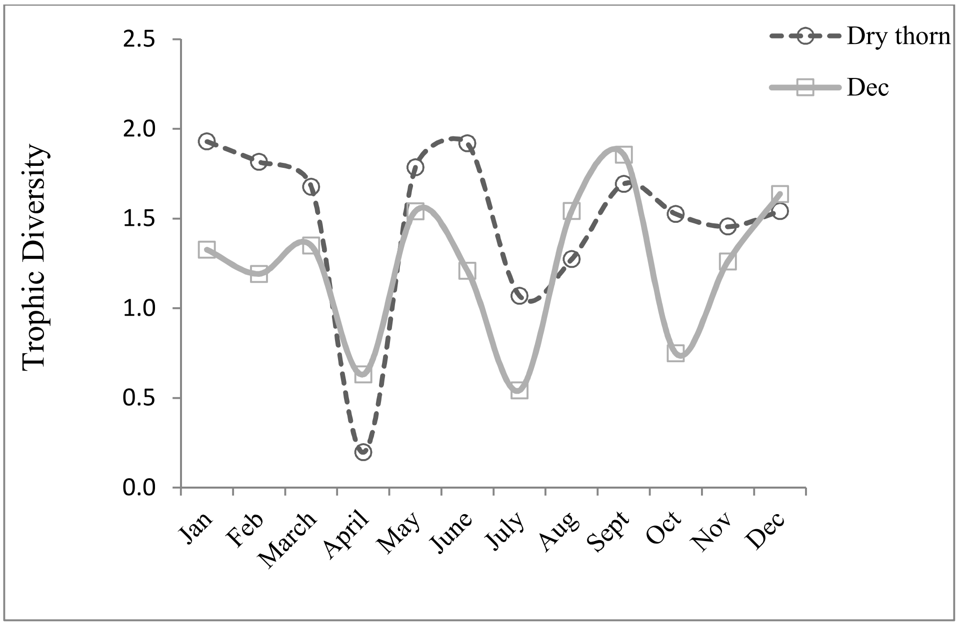
4.3.8 *Diet diversity*

Small cat

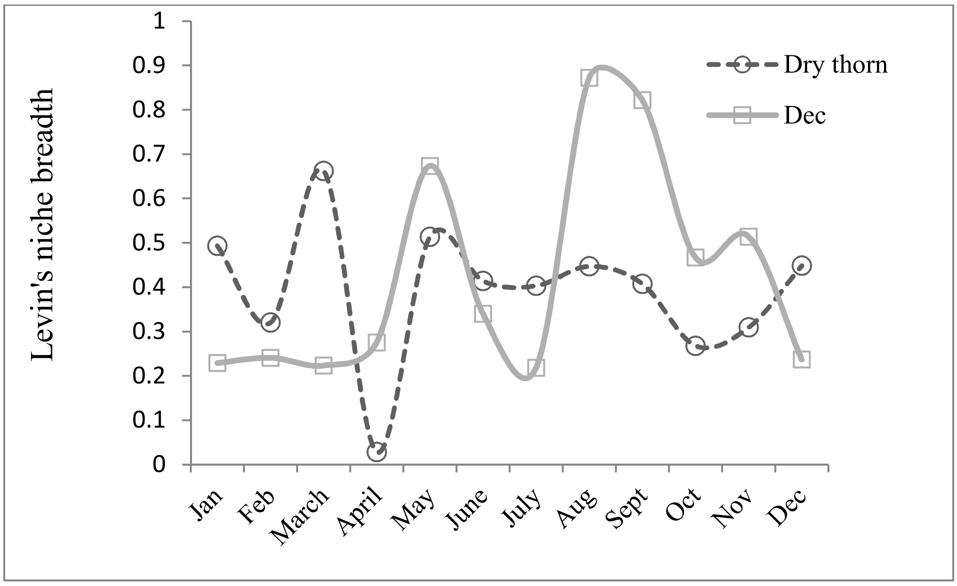
The estimated trophic diversity (Wilcoxon paired sample test, z = -1.88, P = 0.06, Fig 23a) and niche breadth (Wilcoxon paired sample test, z = -1.88, P = 0.06, Fig 23b) did not differ between habitats. Trophic diversity was lowest in April (H' < 0.5) in the thorn forest while in deciduous forest it was low in April, July and Oct. Monthly niche breadth (*B*) varied between 0.03 and 0.87 (mean = 0.4).

Fig 23. Temporal variation in diet diversity a) and niche breadth b) of small cat in Mudumalai Tiger Reserve from 2009-2011.

a)



b)



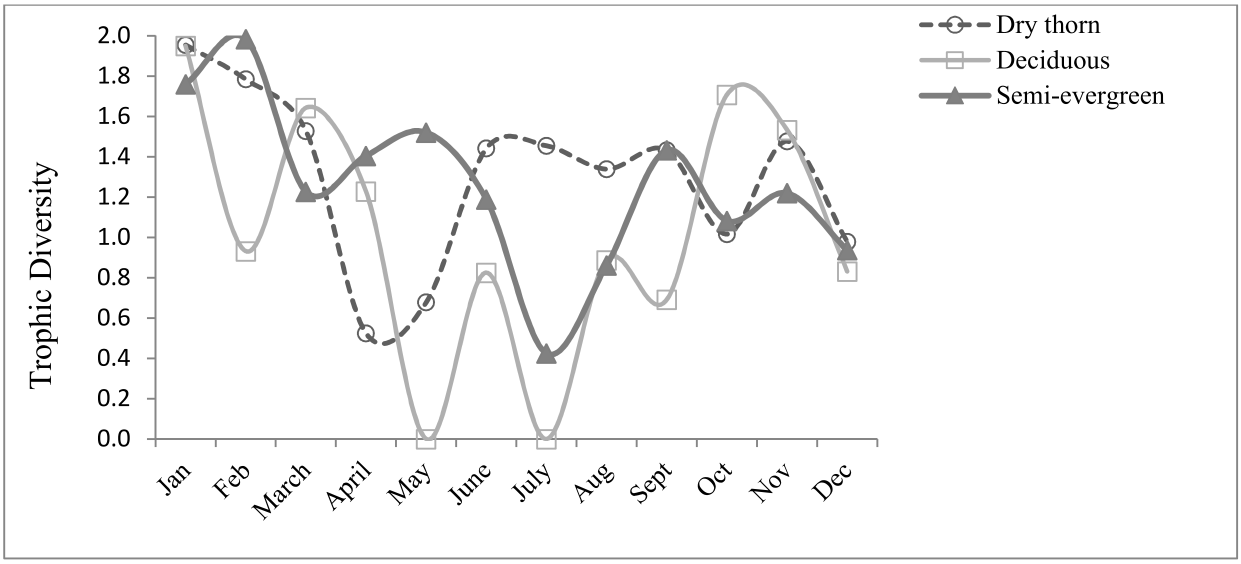
Civet

Trophic diversity did not differ between habitats (Fig 24a) while niche breadth did not differ between thorn and semi-evergreen forest (Wilcoxon paired sample test, z = -

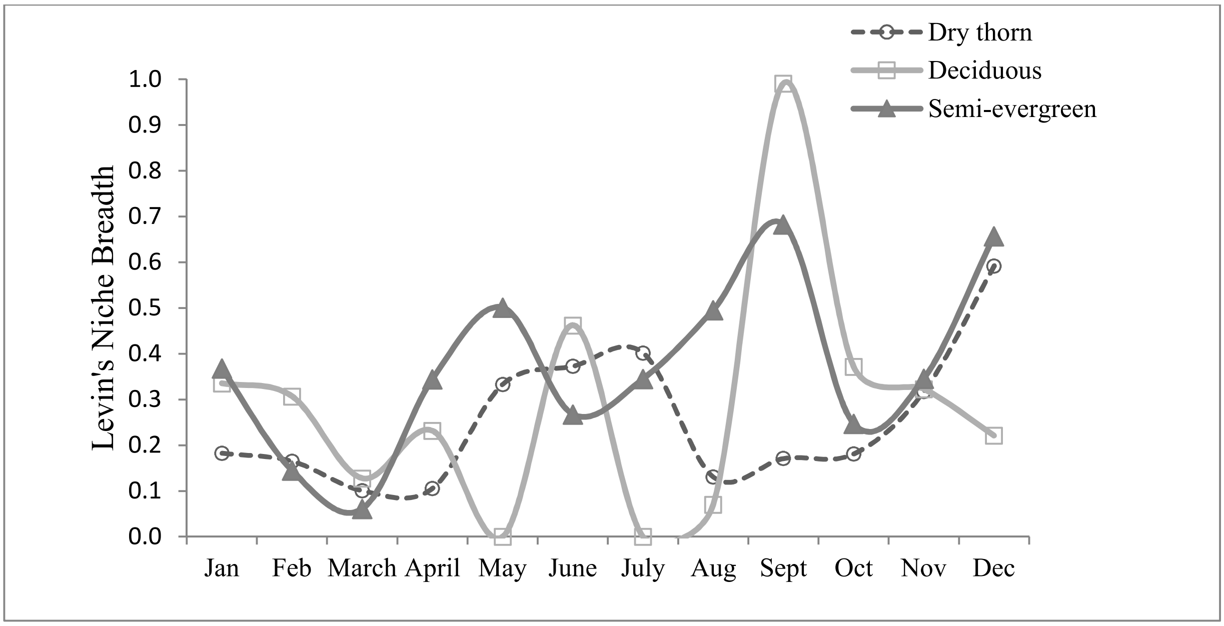
1.88, P = 0.06 Fig 24b). In general, trophic diversity was < 0.5 in April in thorn forest, May and July in deciduous forest and July in semi-evergreen forest. Monthly niche breadth (*B*) varied between 0.07 and 0.99 (mean = 0.3).

Fig 24. Temporal variation in diet diversity a) and niche breadth b) of civet in Mudumalai Tiger Reserve from 2009-2011.

a)



b)

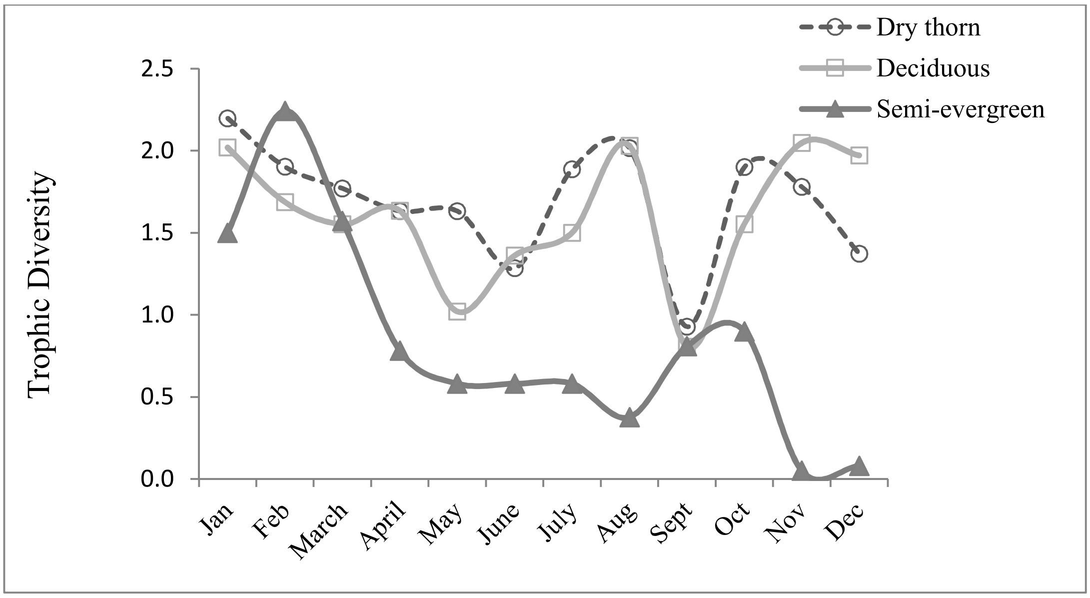


Mongoose

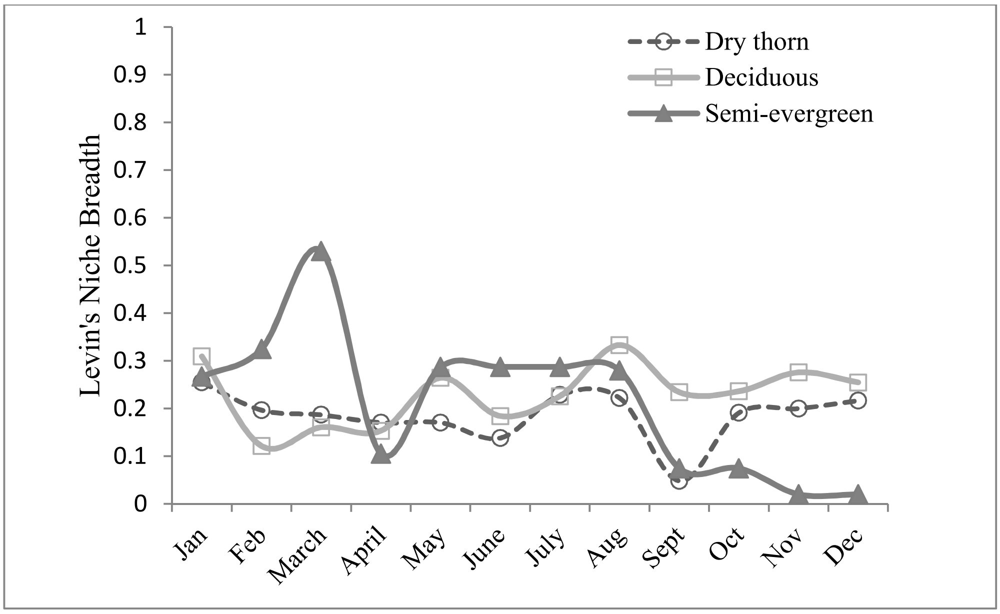
Trophic diversity was significantly higher in deciduous than semi-evergreen forest (Wilcoxon paired sample test, z = -2.43, P = 0.01, Fig 25a) and significantly higher in thorn forest than semi-evergreen forest (Wilcoxon paired sample test, z = -2.82, P = 0.005). Niche breadth was significantly higher in deciduous than thorn forest (Wilcoxon paired sample test, z = -1.96, P = 0.05, Fig 25b). In general, trophic diversity was < 0.5 in Aug, Nov and Dec in semi-evergreen forest, while in other habitats diversity was > 0.5 throughout the year (Fig. 25a). Monthly trophic niche breadth (*B*) varied between 0.02 and 0.53 (mean = 0.21).

Fig 25. Temporal variation in diet diversity a) and niche breadth b) of mongoose in Mudumalai Tiger Reserve from 2009-2011.

a)



b)



4.3.9 *Temporal dietary changes in small carnivores*

Small cat

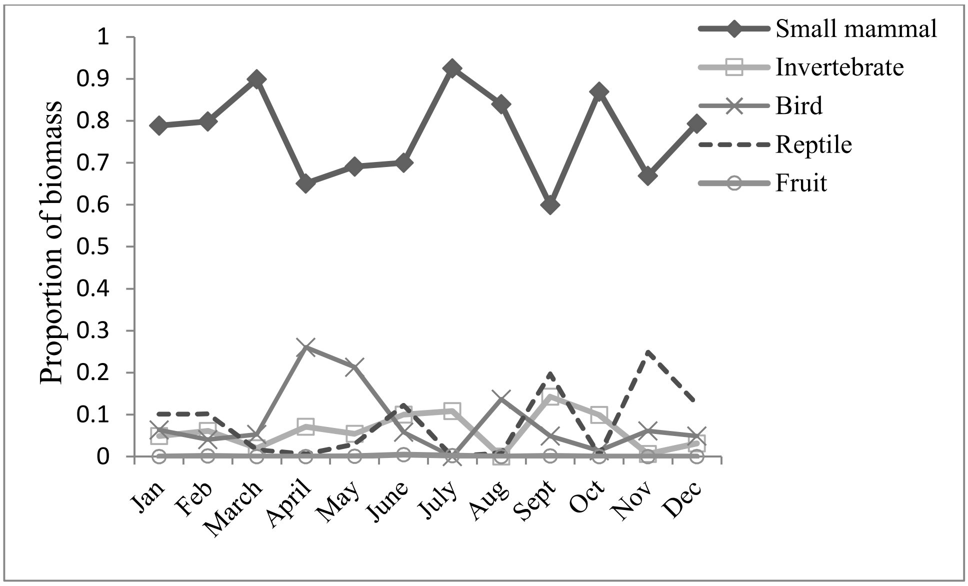
Small mammals were consumed significantly higher than invertebrates (Wilcoxon paired sample test, z = -3.06, P = 0.002), birds (Wilcoxon paired sample test, z = -3.06, P = 0.002), reptile (Wilcoxon paired sample test, z = -3.06, P = 0.002) and fruit (Wilcoxon paired sample test, z = -3.06, P = 0.002, Fig 26a). Invertebrates were consumed significantly higher than fruit (Wilcoxon paired sample test, z = -2.93, P = 0.003). Bird was consumed significantly higher than fruit (Wilcoxon paired sample test, z = -2.98, P = 0.003). Reptile was consumed significantly higher than fruit (Wilcoxon paired sample test, z = -2.98, P = 0.003).

In the thorn forest, small mammal was consumed significantly higher than invertebrates (Wilcoxon paired sample test, z = -3.06, P = 0.002), birds (Wilcoxon paired sample test, z = -3.06, P = 0.002), reptiles (Wilcoxon paired sample test, z = -2.98, P = 0.003) and fruit (Wilcoxon paired sample test, z = -3.06, P = 0.002, Fig 26b). Reptile was consumed significantly higher than fruit (Wilcoxon paired sample test, z = -2.6, P = 0.009). Invertebrates were consumed significantly higher than fruit (Wilcoxon paired sample test, z = -2.5, P = 0.01).

In the deciduous forest, small mammal was consumed significantly higher than invertebrates (Wilcoxon paired sample test, z = -3.06, P = 0.002), bird (Wilcoxon paired sample test, z = -2.98, P = 0.003) and reptile (Wilcoxon paired sample test, z = -3.06, P = 0.002, Fig 26c)

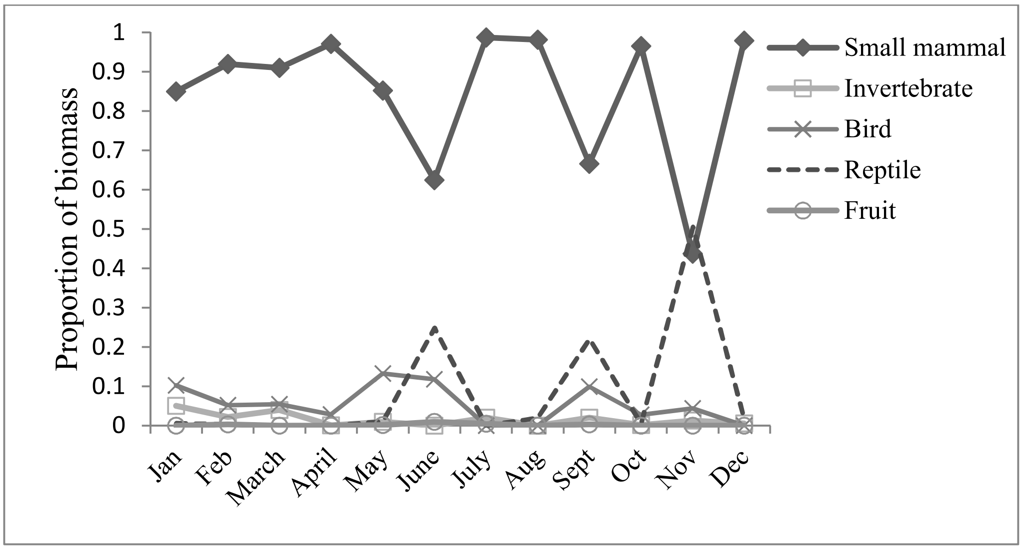
Fig 26. Temporal changes in the diet of small cat in Mudumalai Tiger Reserve (2009-2011). a) overall b) dry thorn c) and deciduous forests

a)



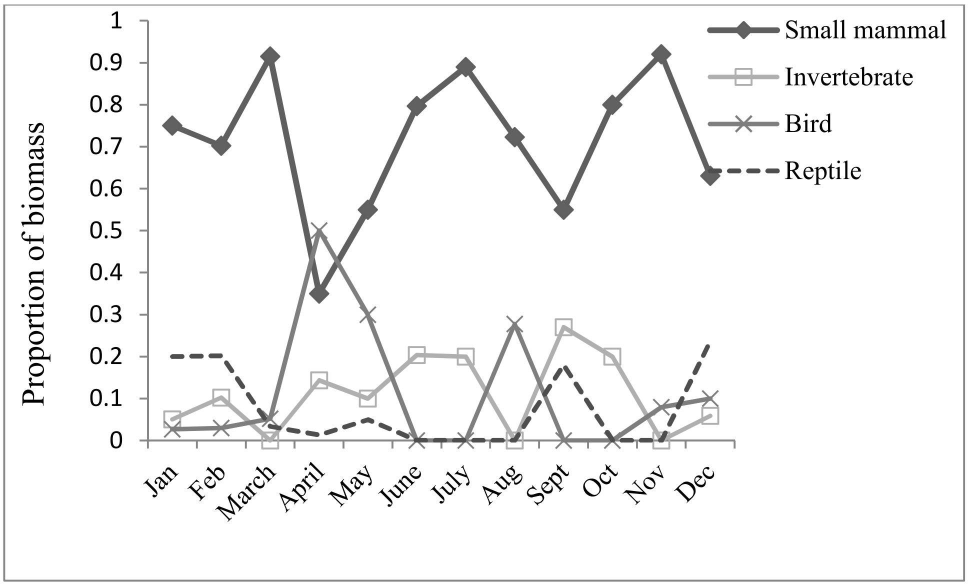
The proportion of prey biomass in fecal samples represented by small mammal, invertebrate, bird, reptile and fruit

b)



The proportion of prey biomass in fecal samples represented by small mammal, invertebrate, bird, reptile and fruit

c)



The proportion of prey biomass in fecal samples represented by small mammal, invertebrate, bird, reptile and fruit

Civet

Overall, there was no difference in the monthly consumption of small mammals and fruits (Fig 27a). Small mammal consumption was significantly higher than invertebrates (Wilcoxon paired sample test, z = -2.82, P = 0.005), birds, (Wilcoxon paired sample test, z = -2.66, P = 0.008) and reptiles (Wilcoxon paired sample test, z = -2.82, P = 0.005, Fig 27a). Fruit biomass was consumed significantly higher than invertebrates (Wilcoxon paired sample test, z = -3.06, P = 0.002), birds (Wilcoxon paired sample test, z = -2.66, P = 0.008) and reptiles (Wilcoxon paired sample test, z = -3.06, P = 0.002).

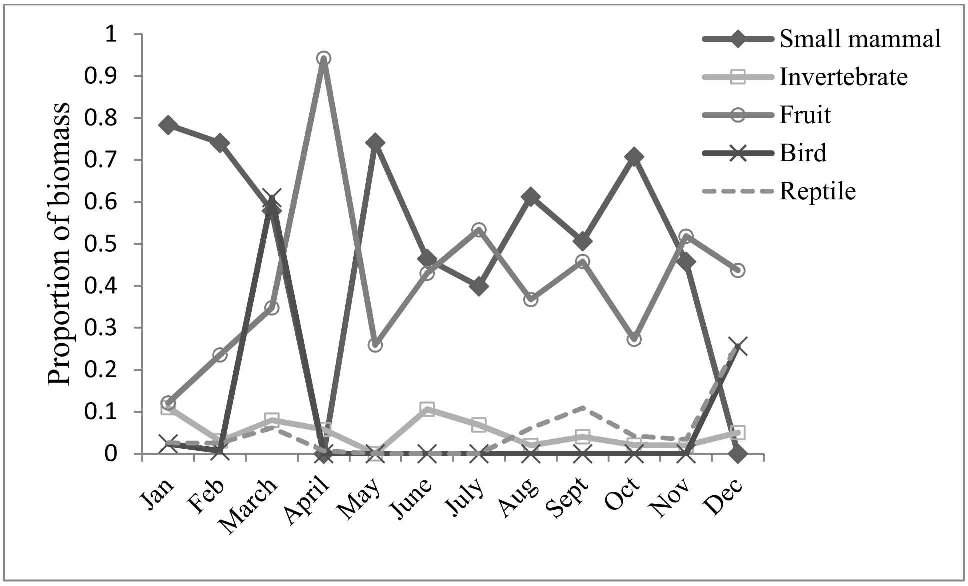
In the thorn forest, small mammal consumption was significantly higher than invertebrates (Wilcoxon paired sample test, z = -2.82, P = 0.005), birds (Wilcoxon paired sample test, z = -2.84, P = 0.004) and reptiles (Wilcoxon paired sample test, z = -2.8, P = 0.005, Fig 27b). However it did not differ from fruit biomass (Wilcoxon paired sample test, z = -0.71, P = 0.5). Fruit biomass was consumed significantly higher than invertebrates (Wilcoxon paired sample test, z = -3.06, P = 0.002), birds (Wilcoxon paired sample test, z =3.06, P = 0.002) and reptiles (3.06, P = 0.002). Invertebrates were consumed significantly higher than birds (Wilcoxon paired sample test, z = -2.93, P = 0.003) and reptiles (Wilcoxon paired sample test, z = -2.93, P = 0.003).

In deciduous forest, there was no difference between small mammal and fruit consumption (Fig 27c). Small mammal consumption was higher than invertebrates (Wilcoxon paired sample test, z = -2.49, P = 0.01) and reptiles (Wilcoxon paired sample test, z = -2.6, P = 0.009). Fruit consumption was significantly higher than invertebrates (Wilcoxon paired sample test, z = -3.06, P = 0.002) and reptiles (Wilcoxon paired sample test, z = -3.06, P = 0.002). Invertebrates were consumed significantly higher than reptiles (Wilcoxon paired sample test, z = -2.52, P = 0.01).

On the contrary in semi-evergreen forest, fruit consumption was significantly higher than small mammals (Wilcoxon paired sample test, z = -2.05, P = 0.04), bird (Wilcoxon paired sample test, z = -2.82, P = 0.005), invertebrate (Wilcoxon paired sample test, z = -2.24, P = 0.02) and reptile (Wilcoxon paired sample test, z = -2.75, P = 0.006, Fig 27d). There was no difference in other food items.

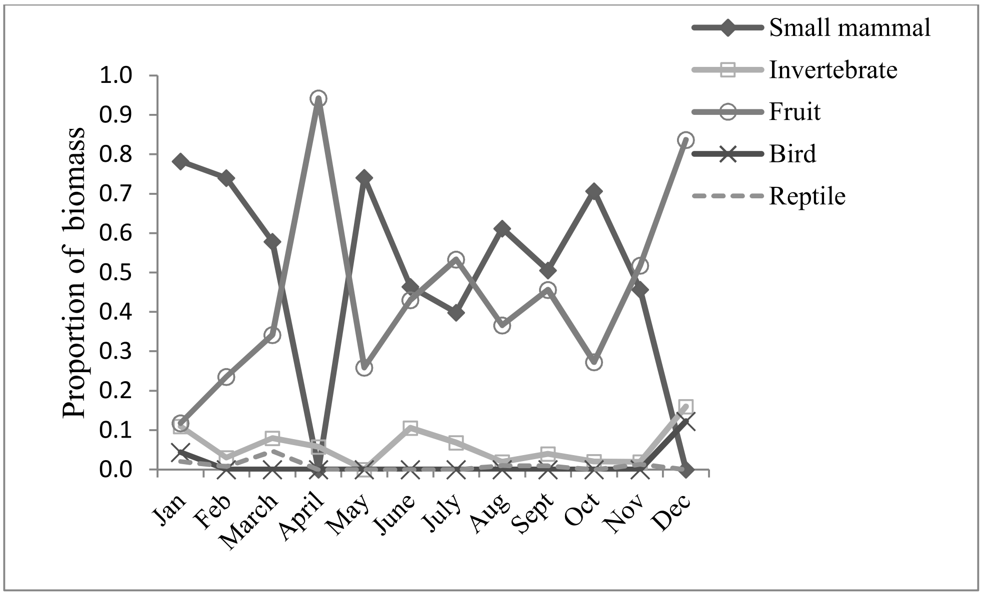
Fig 27. Temporal changes in the diet of civet in Mudumalai Tiger Reserve (2009-2011). a) overall b) dry thorn c) deciduous and d) semi-evergreen forests

a)



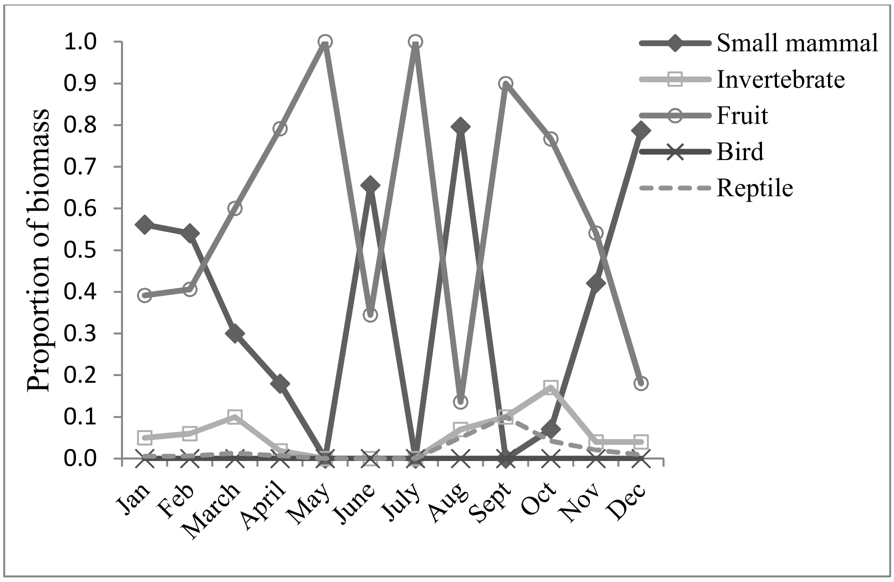
The proportion of prey biomass in fecal samples represented by small mammal, invertebrate, fruit, bird and reptile.

b)



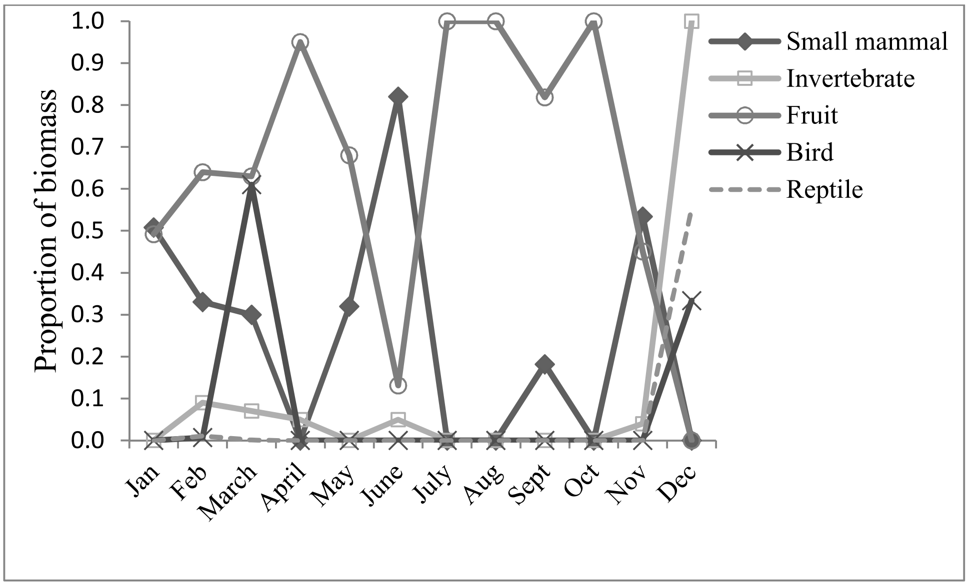
The proportion of prey biomass in fecal samples represented by small mammal, invertebrate, fruit, bird and reptile.

c)



The proportion of prey biomass in fecal samples represented by small mammal, invertebrate, fruit, bird and reptile.

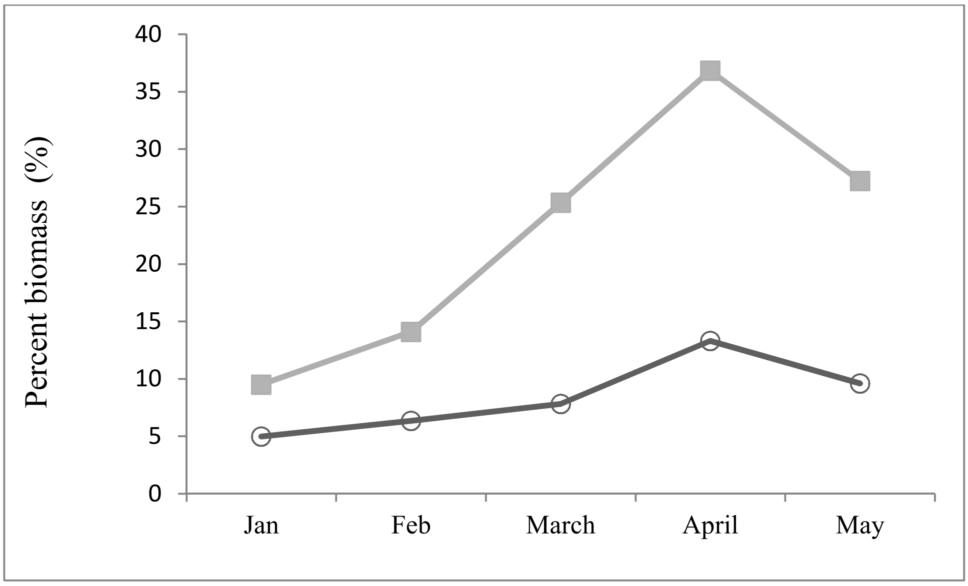
d)



The proportion of prey biomass in fecal samples represented by small mammal, invertebrate, fruit, bird and reptile.

The correlation between ingested percent biomass by civet and available biomass in the environment revealed a positive trend ($R^2 = 0.87$, $P = 0.02$, Fig 28) in the dry season. However fruit diversity in the environment correlated with fruit diversity in faeces ($R^2 = 0.55$, $P = 0.06$).

Fig 28. Overall the percent availability of fruit biomass observed in faeces of the civet (circles) as compared to fruit biomass in the environment (squares) in Mudumalai Tiger Reserve (2009-2011).



Mongoose

Overall, consumption of small mammal biomass was significantly higher than invertebrates (Wilcoxon paired sample test, z = -2.82, P = 0.005, Fig 29a), fruit (Wilcoxon paired sample test, z = -3.06, P = 0.002), bird (Wilcoxon paired sample test, z = -3.06, P = 0.002), reptile (z = -2.67, P = 0.008) and fish (Wilcoxon paired sample test, z = -3.06, P = 0.002, Fig 29a). Small mammal consumption was low in March and August during which they supplemented their diet largely with invertebrates and reptiles. Invertebrate was consumed marginally higher than birds (Wilcoxon paired sample test, z = -1.88, P = 0.06), significantly higher than reptile (Wilcoxon paired sample test, z = -3.06, P = 0.002) and fish (Wilcoxon paired sample test, z = -3.06, P = 0.002). Reptile was consumed significantly higher than fruits (Wilcoxon paired sample test, z = -2.9, P = 0.004), bird (Wilcoxon paired sample test, z = -3.06, P = 0.002) and fish (Wilcoxon paired sample test, z = -3.06, P = 0.002).

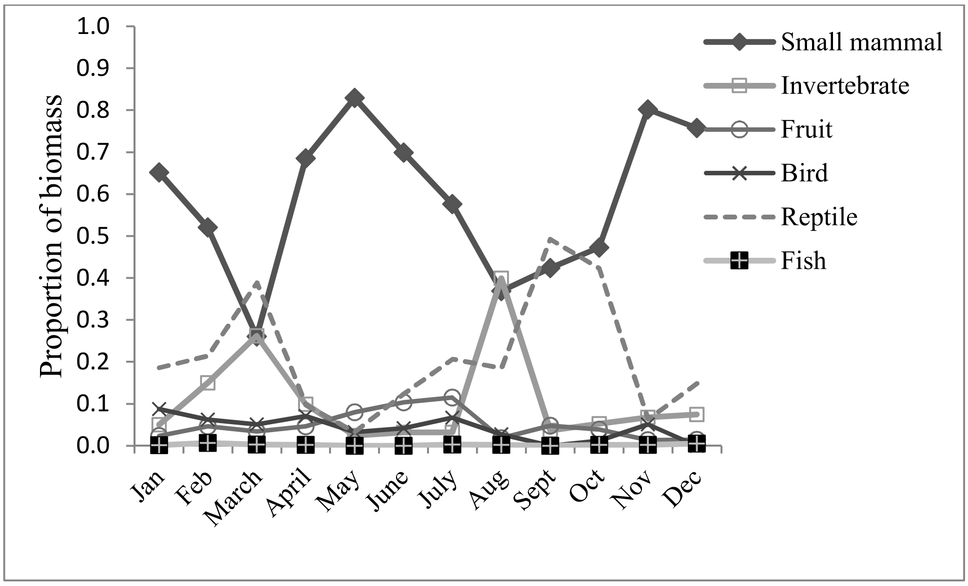
In thorn forest small mammals were consumed higher than invertebrates (Wilcoxon paired sample test, z = -3.06, P = 0.002), fruits (Wilcoxon paired sample test, z = -3.06, P = 0.002), bird (Wilcoxon paired sample test, z = -3.06, P = 0.002), reptile (Wilcoxon paired sample test, z = -2.9, P = 0.004) and fish (Wilcoxon paired sample test, z = -3.06, P = 0.002, Fig 29b). Small mammal consumption was low in March and July, during which they supplemented their diet with mainly reptiles and birds. Invertebrates were consumed more than fruits (Wilcoxon paired sample test, z = -2.82, P = 0.005) and fish (Wilcoxon paired sample test, z = -3.06, P = 0.002).

In deciduous forest small mammals were consumed significantly more than invertebrates (Wilcoxon paired sample test, z = -3.06, P = 0.002), fruits, (Wilcoxon paired sample test, z = -3.06, P = 0.002), bird (Wilcoxon paired sample test, z = = -3.06, P = 0.002) and fish (Wilcoxon paired sample test, z = -3.06, P = 0.002, Fig 29c). Two peaks in reptile consumption were observed in March and September. Reptile consumption was significantly higher than invertebrates (Wilcoxon paired sample test, z =-2.98, P = 0.003), fruit (Wilcoxon paired sample test, z =-2.98, P = 0.003), and fish (Wilcoxon paired sample test, z =-2.98, P = 0.003). Invertebrates were consumed significantly higher than fruits (Wilcoxon paired sample test, z = -2.82, P = 0.005), bird (Wilcoxon paired sample test, z = -2.12, P = 0.03) and fish (Wilcoxon paired sample test, z = -3.06, P = 0.002).

In semi-evergreen forest small mammals were consumed significantly higher than fruits (Wilcoxon paired sample test, z = -2.67, P = 0.007), birds (Wilcoxon paired sample test, z = -2.81, P = 0.005), reptile (Wilcoxon paired sample test, z = -2.23, P = 0.03) and fish (Wilcoxon paired sample test, z = -2.81, P = 0.005 Fig 29d). Invertebrates were consumed significantly higher than birds (Wilcoxon paired sample test, z = -2.36, P = 0.02) and fish (Wilcoxon paired sample test, z = -3.07, P = 0.002). Fruits were consumed weakly higher than birds (Wilcoxon paired sample test, z = -1.84, P = 0.06) but significantly higher than fish (Wilcoxon paired sample test, z = -2.68, P = 0.007).

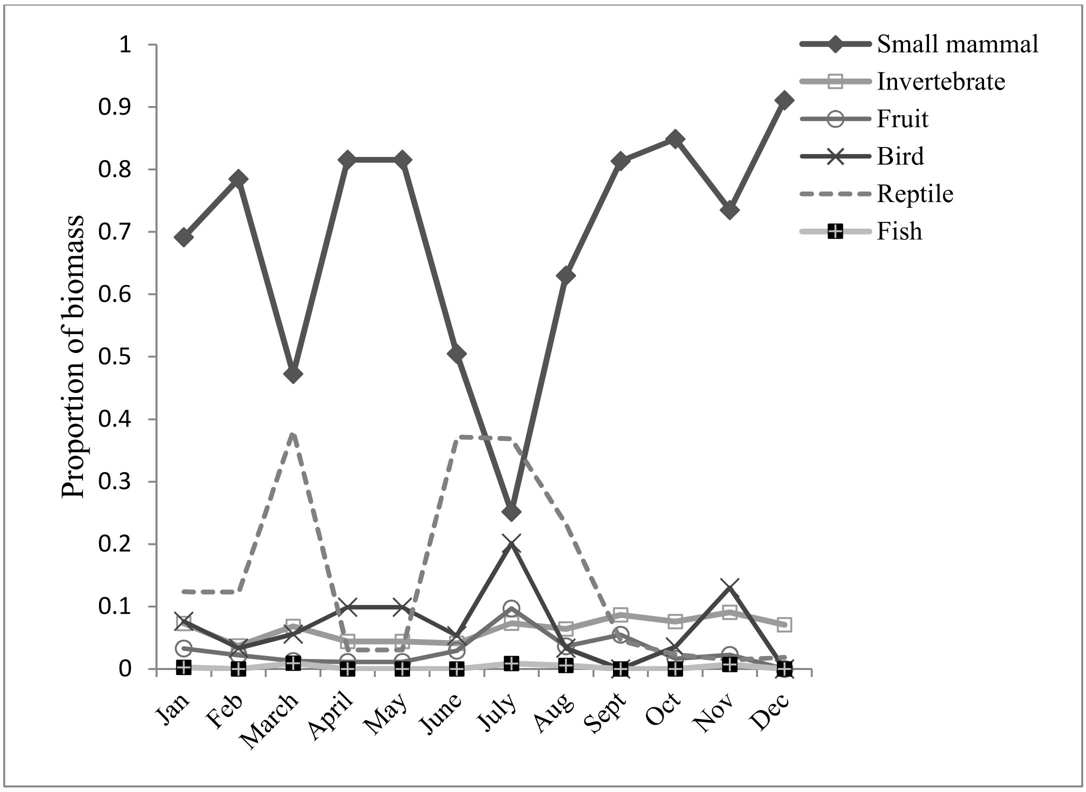
Fig 29. Temporal changes in the diet of mongoose in Mudumalai Tiger Reserve (2009-2011). a) overall b) dry thorn c) deciduous and d) semi-evergreen forests

a)



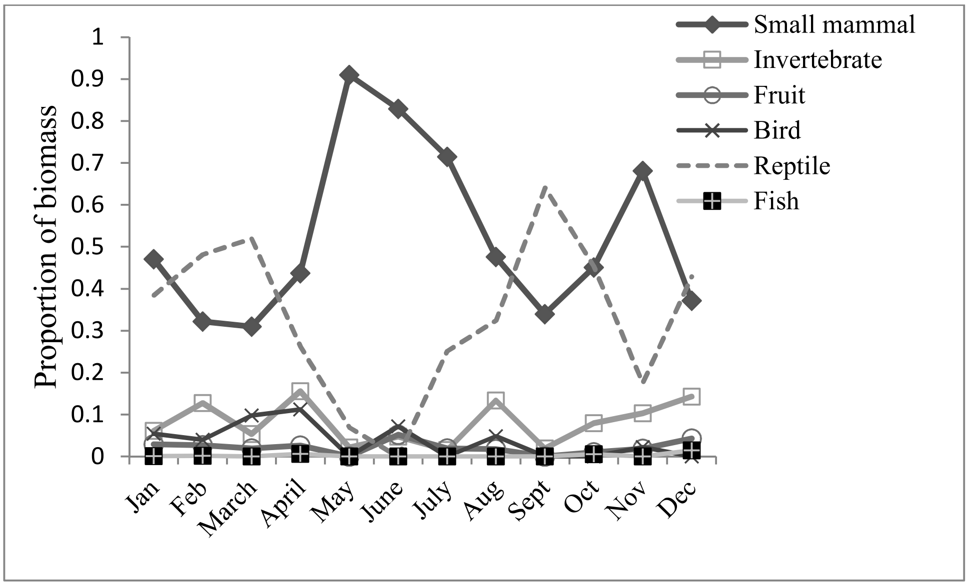
The proportion of prey biomass in fecal samples represented by small mammal, invertebrate, fruit, bird, reptile and fish

b)



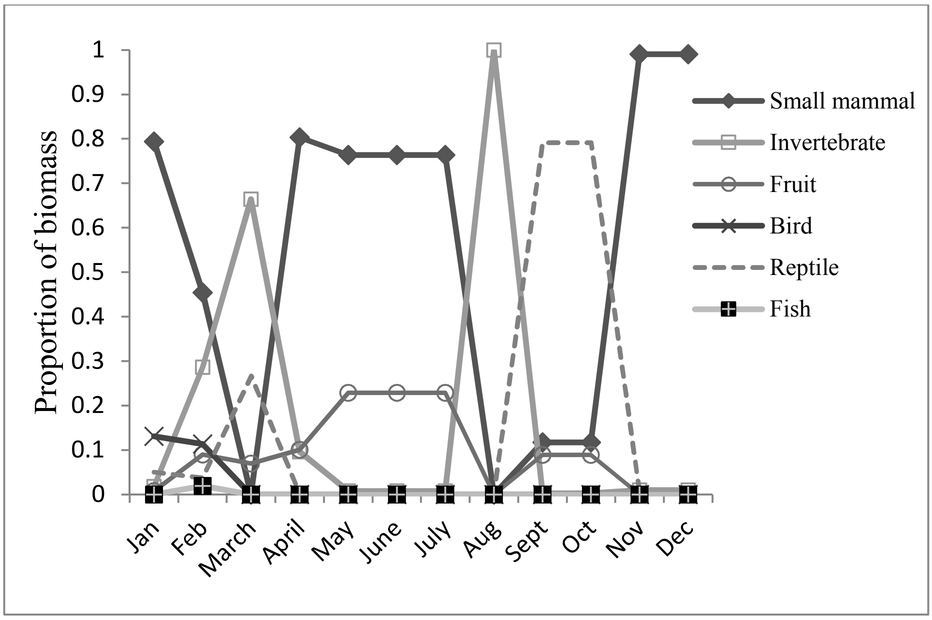
The proportion of prey biomass in fecal samples represented by small mammal, invertebrate, fruit, bird, reptile and fish

c)



The proportion of prey biomass in fecal samples represented by small mammal, invertebrate, fruit, bird, reptile and fish

d)



The proportion of prey biomass in fecal samples represented by small mammal, invertebrate, fruit, bird, reptile and fish

4.3.10 *Dietary differences between small carnivores*

Small mammal, bird, reptile, invertebrate and fruit occurrence did not vary between seasons (Mann–Whitney U test: $U = 26$, P = 0.56; $U = 1$, P = 0.69; $U = 3$, P = 0.66; $U = 30.5$, P = 0.91, $U = 1$, P = 0.69, Table 17) in the diet of small cats. Civets consumed birds in the dry season and not in the wet season. The occurrence of small mammal, reptile, invertebrate did not vary between seasons (Mann–Whitney U test: $U = 18$, P = 0.93; $U = 3$, P = 0.66; $U = 57$, P = 0.84), however fruit occurrence was higher in the dry season than wet season (Mann–Whitney U test: $U = 362.5$, P = 0.04) in the diet of civet (Table 17). Small mammal, bird, reptile, invertebrate and fruit occurrence (Mann–Whitney U test: $U = 31$, P = 0.95; $U = 1$, P = 0.69; $U = 3$, P = 0.66; $U = 76$, P = 0.68; $U = 42.5$, P = 0.59) did not vary between seasons in the diet of mongoose (Table 17).

***Dry season*:** Small cats consumed small mammals more frequently than civets (Mann–Whitney U test: $U = 1$, P = 0.001) and mongooses (Mann–Whitney U test: $U = 6$, P = 0.007). Mongoose consumed invertebrates more than small cats (Mann–Whitney U test: $U = 38.5$, P = 0.01) however it did not differ from civet (Mann–

Whitney U test: *U* = 55, P = 0.1). Fruit occurrence was higher in the diet of civet than small cat (Mann–Whitney U test: *U* = 71, P = <0.001) and mongoose (Mann–Whitney U test: *U* = 167, P = < 0.001). Reptile egg shells were recorded only in faeces of mongooses.

Wet season: Small mammal occurrence was higher in small cat than civet (Mann–Whitney U test: *U* = 9, P = 0.02). Mongoose consumed more invertebrates than small cat (Mann–Whitney U test: *U* = 45.5, P = 0.04). Civet consumed fruits more than small cat (Mann–Whitney U test: *U* = 253.5, P = < 0.001) and mongooses (Mann–Whitney U test: *U* = 383.5, P = 0.05, Table 17). Fish and carrion was recorded only in small cats and mongooses. Mongooses consumed pilli millipedes and dung beetles with greater frequency than other carnivores.

Table 17. Diet composition (relative frequency occurrence %) of small cat, civet and mongoose during dry sand wet season in Mudumalai Tiger Reserve (2009-2011).

Food item	**Small cat**		**Civet**		**Mongoose**	
	Dry (n =219)	**Wet (n = 136)**	**Dry (n = 574**	**Wet (n = 257)**	**Dry (n = 391)**	**Wet (n =320)**
Small mammal	**67.55**	**61.67**	**6.47**	**6.82**	**16.68**	**20.2**
Lepus nigricollis	6.19	0.83	0.08	-	1.02	0.68
Unknown rodent	7.26	9.41	3.94	3.73	7.84	5.23
Rattus spp	8.38	4.83	1.03	0.99	2.11	2.18
Tatera indica	11.79	11.61	-	-	0.07	-
Cremnomys blanfordi	9.77	10.77	0.82	-	2.40	3.25
Millardia meltada	2.13	0.83	-	-	0.10	-
Mus spp	16.82	21.97	0.37	1.21	2.15	7.07
Suncus montanus	5.21	1.40	0.24	0.88	1.00	1.79
Carrion	**0.68**	**1.14**	-	-	**0.95**	**0.88**
Rusa unicolor	-	-	-	-	0.07	-
Axis axis	0.68	1.14	-	-	0.89	0.88
Bird	**11.43**	**9.71**	**1.25**	**-**	**4.82**	**1.28**
Unknown bird	7.30	6.63	-	-	4.18	1.28
Quail	4.13	3.08	-	-	0.64	-
Reptile	**5.33**	**7.87**	**2.38**	**5.09**	**10.38**	**7.34**
Calotes spp	3.06	2.62	1.11	0.80	6.05	4.68

Snake	0.34	3.00	0.83	3.36	2.60	2.43
Skink	1.93	2.24	0.43	0.94	1.73	0.23
Fish	**0.68**	**1.48**	-	-	**0.98**	**1.49**
Invertebrates	**8.87**	**13.23**	**13.78**	**23.27**	**51.25**	**50.47**
Crustacean	1.73	1.14	4.05	5.10	3.35	3.18
Mollusc	-	-	0.86	0.18	1.14	0.83
Pilli millipede	3.63	4.41	3.14	9.17	18.12	18.85
Dragonfly	0.26	-	-	-	-	0.26
Dung beetle	1.14	5.47	3.19	5.32	17.42	20.02
Scorpion	1.05	1.40	0.11	0.61	2.54	2.26
Termite	-	0.80	0.28	-	2.12	1.98
Grasshopper	1.05	-	-	1.47	1.06	0.14
Unknown insect	-	-	0.71	0.80	1.56	1.04
Red ant	-	-	0.08	-	0.07	0.37
Cricket	-	-	-	-	1.37	-
Moth	-	-	-	-	-	0.14
Millipede	-	-	1.36	0.61	2.51	1.40
Reptile egg	-	-	-	-	0.07	-
Fruit	**1.79**	**0.83**	**74.27**	**63.48**	**12.79**	**14.61**
Grewia tiliifolia	-	-	31.24	26.46	3.29	1.56
Cassia fistula	-	-	11.35	7.20	-	-
Lantana camara	1.11	-	2.04	3.17	1.18	1.85
Zizyphus oenophilia	0.68	-	4.84	5.71	3.96	3.25
Zizyphus mauritiana	-	-	0.88	0.28	0.10	0.23
Unknown fruit	-	-	3.40	11.20	3.02	3.01
Antidesma zeylanicum	-	-	4.32	-	-	-
Bischovia javanica	-	-	4.88	-	-	-
Syzigium cumini	-	-	-	0.18	-	0.60
Gnetum ula	-	-	2.53	-	-	-
Caryota urens	-	-	1.26	1.10	-	-
Cordia obliqua	-	0.83	0.33	0.88	0.90	2.67
Coffee spp	-	-	1.09	-	-	-
Cordia spp	-	-	0.94	1.97	-	-
Erythroxylum monogynum	-	-	0.39	-	-	-
Naringi crenulata	-	-	0.11	-	-	-
Bridelia crenulata	-	-	0.26	0.37	0.23	1.30

Piper nigrum	-	-	0.27	-	-	-
Embelia gardneriana	-	-	0.85	1.05	-	-
Ficus spp	-	-	0.38	-	-	0.14
Diospyros montana	-	-	0.15	1.54	-	-
Symplocos cochinchinensis	-	-	0.28	-	-	-
Flacourtia indica	-	-	0.04	-	-	-
Phoenix iourerii	-	-	0.25	-	-	-
Cucurbitaceae	-	-	-	0.18	-	-
Mimusops elengi	-	-	0.09	0.18	-	-
Tamarandus indicus	-	-	0.70	-	-	-
Breynia retusa	-	-	0.01	-	-	-
Polyalthia cerasoides	-	-	0.12	-	-	-
Elaeocarpus variabilis	-	-	0.30	-	-	-
Grewia hirsuta	-	-	0.42	0.33	-	-
Unknown plant			0.57	1.68	0.13	-
Vegetative matter	**3.67**	**4.07**	**1.86**	**1.34**	**2.08**	**3.73**
Herb	-	-	0.36	-	-	-
Grass	3.67	4.07	1.50	1.34	2.08	3.73

4.3.11 *Trophic niche overlap and niche breadth*

Overall, small mammals contributed maximum to the diet of small cat, invertebrates in mongoose and fruit in civets (Fig 30). Civet and mongoose consumed small sized food items (1-10 g) in equal proportion and comparatively higher than small cats (Fig 31), on the other hand large sized prey (> 30 g) were consumed higher by small cats than civet and mongoose.

Fig 30. Overall diet composition (relative frequency occurrence %) of small cat, civet and mongoose in Mudumalai Tiger Reserve (2009-2011).

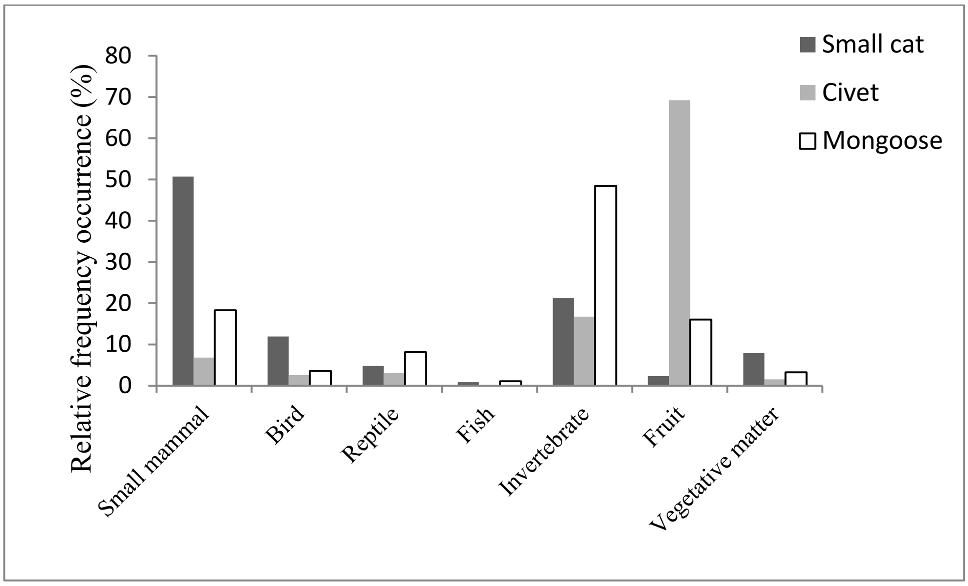
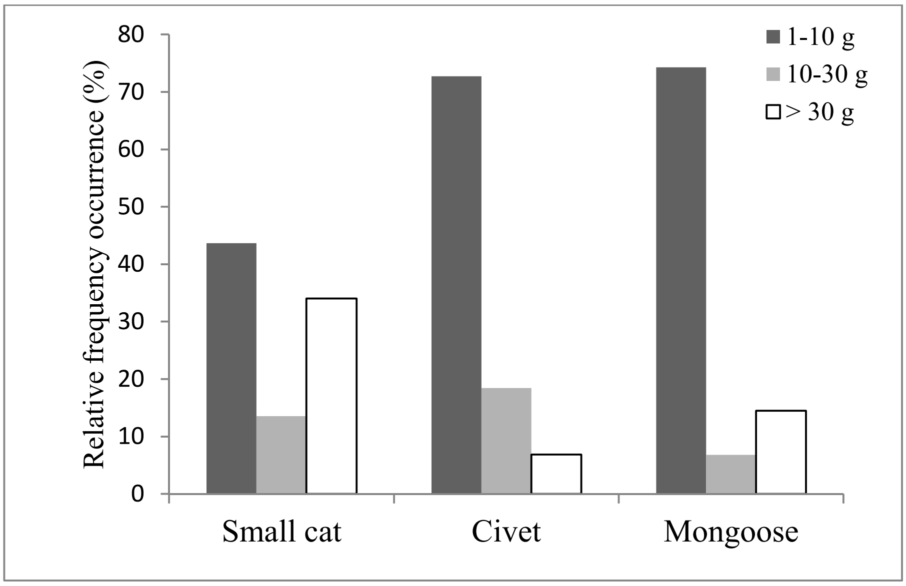


Fig 31. Prey size categories selected by small cats, civet and mongooses (2009-2011).

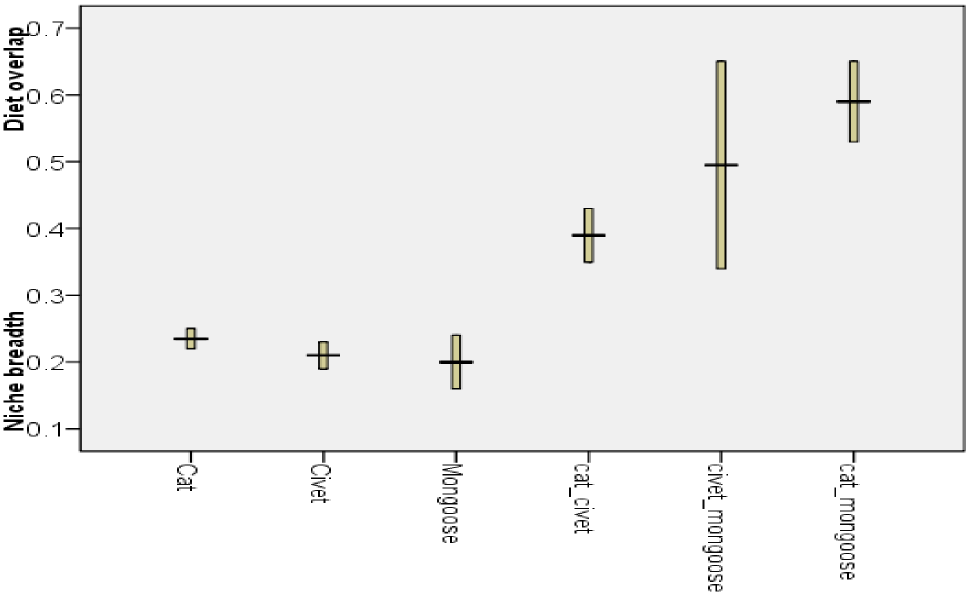


Niche overlap between civet and mongoose and between cat and mongoose was high. (Table 18). Niche overlap between civet and mongoose was higher in the dry season than wet season however there was no difference in other species pairs. Niche breadth was highest in mongoose followed by civet and small cat. Overall niche overlap was higher between cat and mongoose than other species pairs (Fig 32).

Table 18. Trophic niche overlap and niche breadth of small carnivores in Mudumalai Tiger Reserve (2009-2011).

	Niche overlap		
Season	**Small cat/Civet**	**Civet/Mongoose**	**Small cat/Mongoose**
Dry	0.43	0.65	0.65
Wet	0.35	0.34	0.53
	Niche breadth		
	Small cat	**Civet**	**Mongoose**
Dry	0.25	0.19	0.24
Wet	0.22	0.23	0.16

Fig 32. Trophic niche breadth and niche overlap of small carnivores in Mudumalai Tiger Reserve (2009-2011).



4.3.12 *Food preference by small carnivores*

Small cat

In the dry season, small cats selected *Tatera indica*, *Millardia meltada* and *Suncus montanus* positively and significantly more than their availability compared to their relative abundance in the wild, while *Rattus spp* was significantly underrepresented

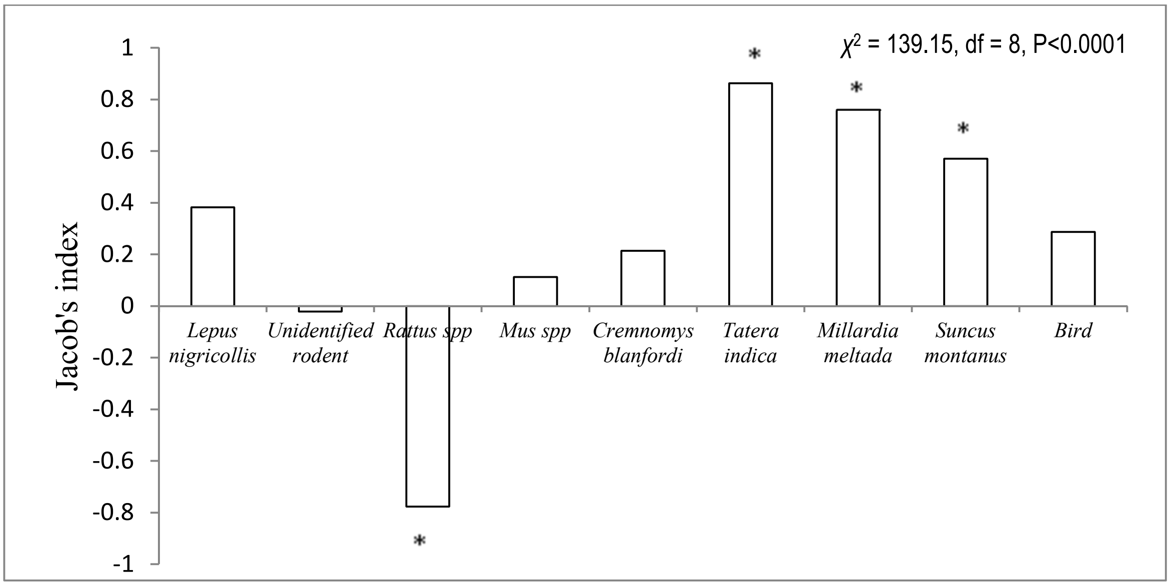
(Fig 33a, Table 19). *Mus spp*., unidentified rodent and *Cremnomys blanfordi* were consumed in proportion to their availability in the dry season. In the wet season small cats selected *Tatera indica* and *Millardia meltada* positively and significantly more than its availability in the environment while *Lepus nigricollis* and *Suncus spp* were underrepresented significantly in the wet season (Fig 33b, Table 19). *Mus spp* were consumed in proportion to its availability in the wet season.

Table 19. Food selection by small cat based on the Bonferroni confidence intervals. Positive (prey species consumed more than expected) or negative (consumed less than expected) selection are considered at the 0.05 level of significance.

Food item	Observed proportion$^{\$}$	Expected proportion$^{\$}$	Bonferroni 95% Confidence intervals$^{\$}$	Selection$^{\#}$	Observed proportion$^{\#}$	Expected proportion$^{\#}$	Bonferroni 95% Confidence Intervals$^{\#}$	Selection$^{\#}$
Lepus nigricollis	0.19	0.09	0.016, 0.18	1	0.04	0.21	-0.24, -0.093	3
Unidentified rodent	0.08	0.08	-0.07, 0.06	2	0.16	0.07	0.016, 0.162	1
Rattus rattus	0.15	0.53	-0.53, -0.3	3	0.09	0.20	-0.18, -0.03	3
Mus spp	0.04	0.03	-0.035, 0.05	2	0.05	0.07	-0.07, 0.037	2
Cremnomys blanfordi	0.22	0.14	-0.025, 0.15	2	0.15	0.31	-0.26, -0.066	3
Tatera indica	0.07	0.01	0.021, 0.11	1	0.26	0.03	0.15, 0.301	1
Millardia meltada	0.01	0.001	-0.01, 0.03	1	0.02	0.004	-0.01, 0.035	1
Suncus montanus	0.01	0.003	-0.012, 0.03	1	0.002	0.01	-0.03, 0.01	3
Bird	0.11	0.06	-0.02, 0.11	2	0.04	0.09	-0.09, 0.002	3

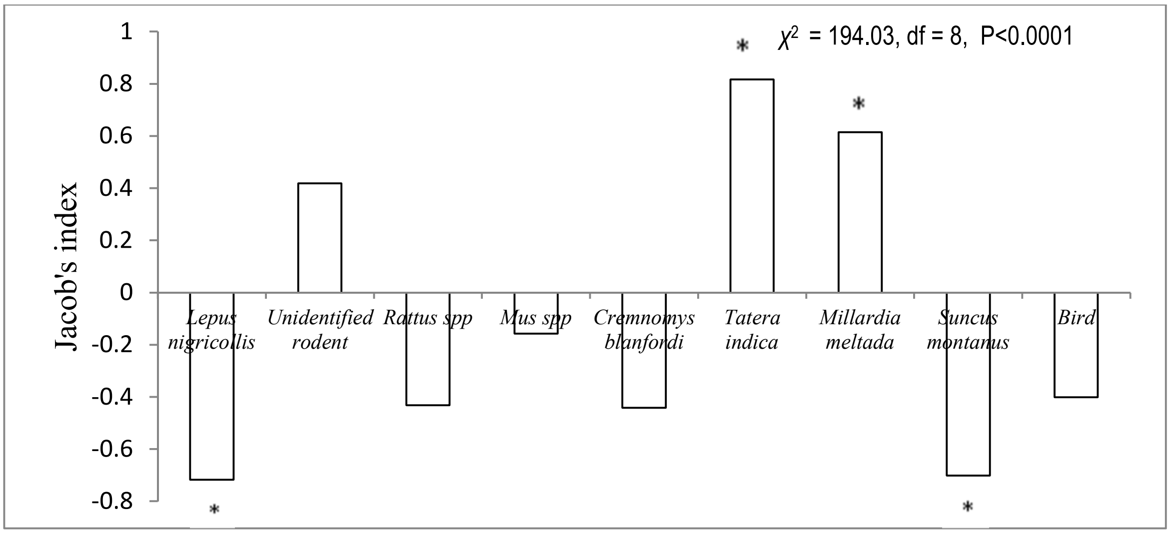
$^{\$}$ = Dry season, $^{\#}$ = Wet season; Selection; 1 = used greater than expected; 2 = proportional use; 3 = used less than expected

Fig 33a. Consumption of major prey items by small cats in the dry season derived from Jacob's index (2009-2011).



Chi-square test demonstrated that there was a highly significant difference between the expected utilization of the food items and the observed frequency of use; * = significant positive or negative selection according to Bonferroni simultaneous confidence intervals at P ≤ 0.05

b. Consumption of major prey items by small cats in the wet season derived from Jacob's index (2009-2011).



Chi-square test demonstrated that there was a highly significant difference between the expected utilization of the food items and the observed frequency of use; * = significant positive or negative selection according to Bonferroni simultaneous confidence intervals at P ≤ 0.05

Civet

Civets selected unidentified rodent, *Grewia tiliifolia* and *Cassia fistula* positively and significantly more than its availability in the wild while other food items were underrepresented in the dry season (Fig 34a, Table 20). *Cremnomys blanfordi*, *Suncus montanus*, *Zizyphus oenophilia*, *Cordia obliqua* and *Antidesma zeylanicum* were consumed in proportion to their availability in the dry season. *Cordia obliqua* was utilized equal to its proportion in the environment in the dry season. In the wet season, civets selected unidentified rodent, *Grewia tiliifolia*, *Lantana camara*, *Zizyphus oenophilia*, *Cassia fistula*, and *Diospyros montana* positively and significantly more than its availability in the environment (Fig 34b, Table 20). *Bridelia crenulata* was consumed in proportion to its availability in the wet season. *Syzigium cumini* and *Zizyphus mauritiana* were significantly underrepresented in their diet in the wet season.

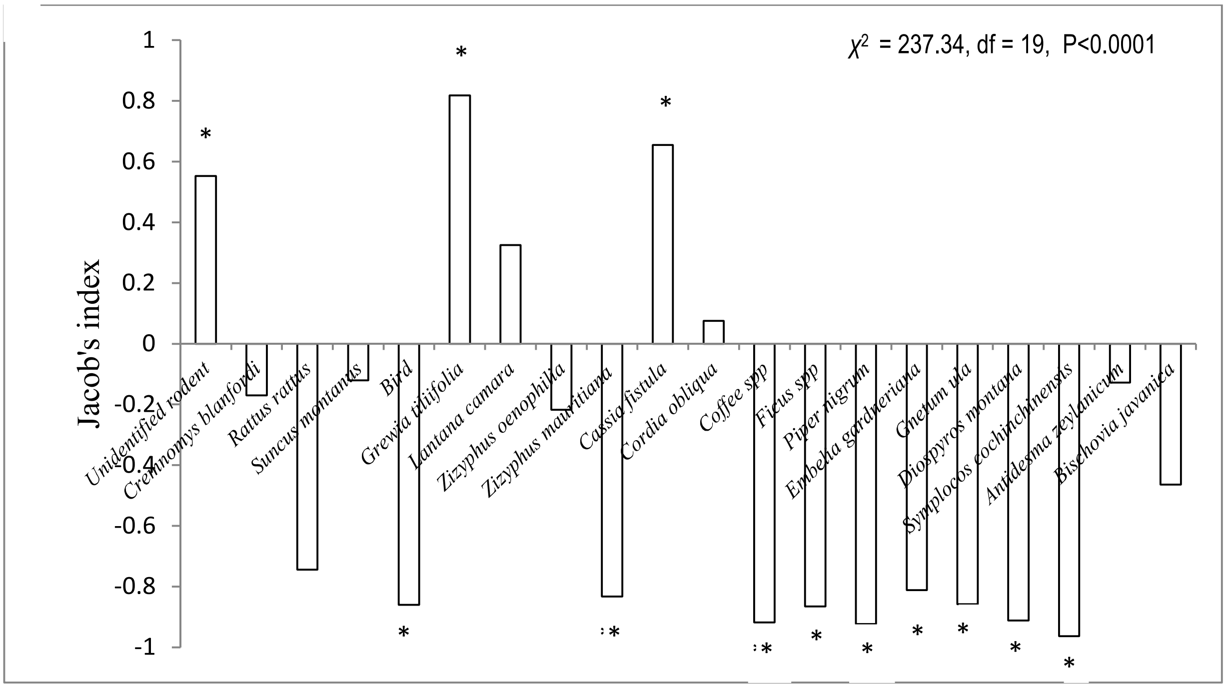
Table 20. Food selection by civet based on the Bonferroni confidence intervals. Positive (prey species consumed more than expected) or negative (consumed less than expected) selection are considered at the 0.05 level of significance.

Food item	Observed proportion$^{\$}$	Expected proportion$^{\$}$	Bonferroni 95% Confidence intervals$^{\$}$	Selection$^{\$}$	Observed proportion$^{\#}$	Expected proportion$^{\#}$	Bonferroni 95% Confidence intervals$^{\#}$	Selection$^{\#}$
Unidentified rodent	0.19	0.06	0.19, -0.07	1	0.22	0.1	0.03, 0.21	1
Cremnomys blanfordi	0.08	0.11	-0.07, 0.04	2	-	-	-	-
Rattus spp	0.09	0.4	-0.22, -0.09	3	0.06	0.22	-0.24, -0.08	3
Mus spp	-	-	-	-	0.006	0.017	-0.04, 0.015	3
Suncus montanus	0.002	0.003	-0.12, 0.12	2	0.015	0.008	-0.02, 0.03	1
Bird	0.004	0.05	-0.13, 0.08	3	0.04	0.1	-0.12, -0.005	3
Grewia tiliifolia	0.17	0.02	-0.06, 0.21	1	0.2	0.014	0.016, 0.34	1
Lantana camara	0.01	0.006	-0.02, 0.03	1	0.03	0.006	-0.01, 0.057	1
Zizyphus oenophilia	0.012	0.02	-0.03, 0.02	3	0.036	0.009	-0.008, 0.06	1
Zizyphus mauritiana	0.002	0.02	-0.02, 0.01	3	0.001	0.02	-0.04, 0.006	3
Cassia fistula	0.27	0.07	-0.09, 0.29	1	0.2	0.012	-0.057, 0.0003	1
Cordia obliqua	0.02	0.02	-0.03, 0.04	2	0.01	0.004	0.1, 0.25	1
Syzigium cumini	-	-	-	-	0.001	0.03	-0.005, 0.02	3

Coffee spp	0.001	0.03	-0.06, 0.03	3	-	-	-	-
Bridelia crenulata	-	-	-	-	0.016	0.01	-0.014, 0.03	2
Ficus spp	0.001	0.02	-0.06, 0.04	3	-	-	-	-
Piper nigrum	0.001	0.015	-0.06, 0.04	3	-	-	-	-
Embelia gardneriana	0.001	0.008	-0.05, 0.05	3	0.001	0.003	-0.01,0.01	3
Gnetum ula	0.0004	0.03	-0.08, 0.06	3	-	-	-	-
Diospyros montana	0.001	0.02	-0.06, 0.04	3	0.03	0.01	0.015,0.026	1
Symplocos cochinchinensis	0.0003	0.02	-0.02, 0.01	3	-	-	-	-
Antidesma zeylanicum	0.01	0.013	-0.03, 0.02	2	-	-	-	-
Bischovia javanica	0.011	0.03	-0.03, 0.02	3	-	-	-	-

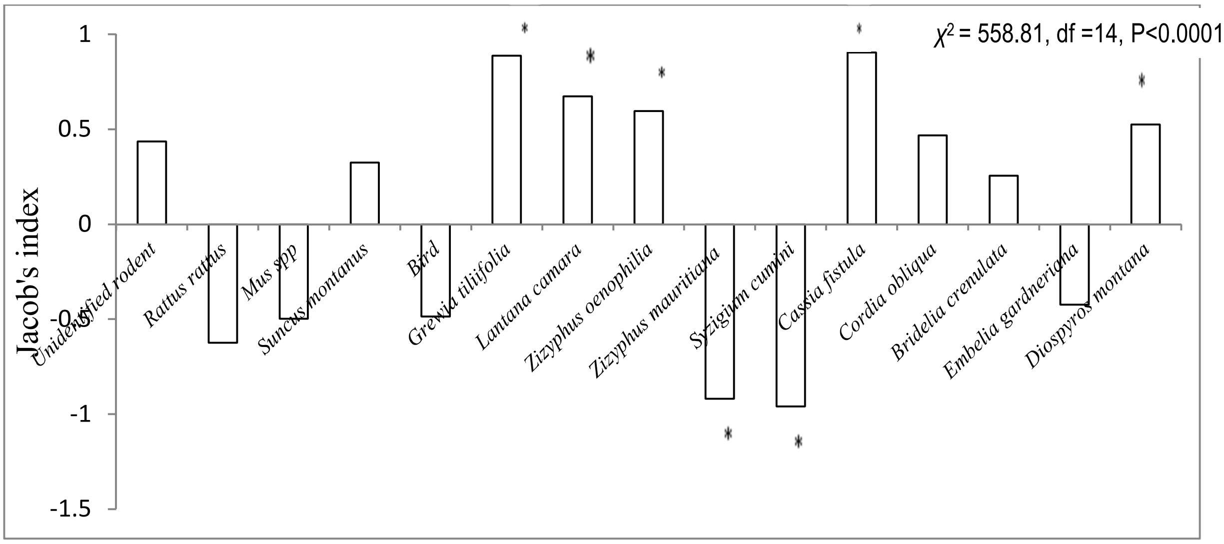
$^{\$}$ = Dry season, $^{\#}$ = Wet season; Selection; 1 = used greater than expected; 2 = proportional use; 3 = used less than expected

Fig 34a. Consumption of major food items by civets in the dry season derived from Jacob's index (2009-2011).



Chi-square test demonstrated that there was a highly significant difference between the expected utilization of the food items and the observed frequency of use; * = significant positive or negative selection according to Bonferroni simultaneous confidence intervals at P ≤ 0.05

b. Consumption of major food items by civets in the wet season derived from Jacob's electivity index (2009-2011).



Chi-square test demonstrated that there was a highly significant difference between the expected utilization of the food items and the observed frequency of use; * = significant positive or negative selection according to Bonferroni simultaneous confidence intervals at P ≤ 0.05

Mongoose

Mongoose selected unidentified rodent and *Millardia meltada* positively and significantly more than its availability in the dry season. *Rattus spp* and *Lantana camara* were significantly underrepresented in the diet during dry season. *Grewia tiliifolia* and *Cordia obliqua* were consumed equal to its proportion in the environment (Fig 35a, Table 21). In the wet season also mongoose selected *Millardia meltada* and *Suncus montanus* positively and significantly more than its availability in the wild while other food items were underrepresented (Fig 35b, Table 21).

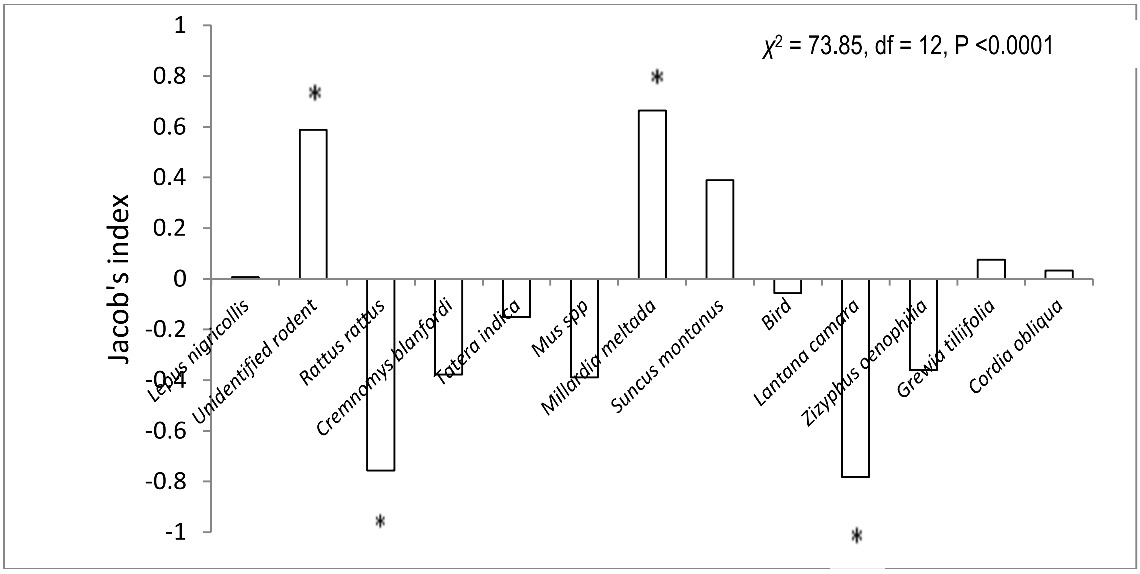
Table 21. Food selection by mongoose based on the Bonferroni confidence intervals. Positive (prey species consumed more than expected) or negative (consumed less than expected) selection are considered at the 0.05 level of significance.

Food item	Observed proportion[$]	Expected proportion[$]	Bonferroni 95% Confidence intervals[$]	Selection[$]	Observed proportion[#]	Expected proportion[#]	Bonferroni 95% Confidence intervals[#]	Selection[#]
Lepus nigricollis	0.1	0.09	-0.07, 0.07	2	0.11	0.19	-0.16, 0.002	3
Unidentified rodent	0.25	0.08	0.085, 0.26	1	0.14	0.07	-0.0001, 0.14	1
Rattus rattus	0.092	0.42	-0.42, -0.24	3	0.05	0.18	-0.205, -0.06	3
Mus spp	0.058	0.12	-0.15, -0.003	3	0.01	0.06	-0.098, -0.01	3
Cremnomys blanfordi	0.074	0.15	-0.021, 0.02	2	0.12	0.29	-0.26, -0.08	3
Tatera indica	0.004	0.01	-0.13, 0.002	3	-	-	-	-
Millardia meltada	0.008	0.002	-0.007, 0.02	1	0.02	0.004	-0.009, 0.04	1
Suncus montanus	0.008	0.004	-0.01, 0.02	1	0.17	0.01	0.09, 0.22	1
Bird	0.06	0.07	-0.06, 0.05	2	0.02	0.08	-0.116, -0.017	3
Grewia tiliifolia	0.016	0.01	-0.02, 0.005	2	0.003	0.01	-0.03, 0.01	3
Lantana camara	0.001	0.01	-0.034, 0.016	3	0.001	0.005	-0.02, 0.009	3
Zizyphus oenophilia	0.008	0.02	-0.012, 0.02	3	0.006	0.008	-0.02, 0.018	3

Zizyphus mauritiana	-	-	-	-	0.01	0.015	-0.03, 0.025	2
Cordia obliqua	0.016	0.015	-0.03, 0.03	2	-	-	-	-

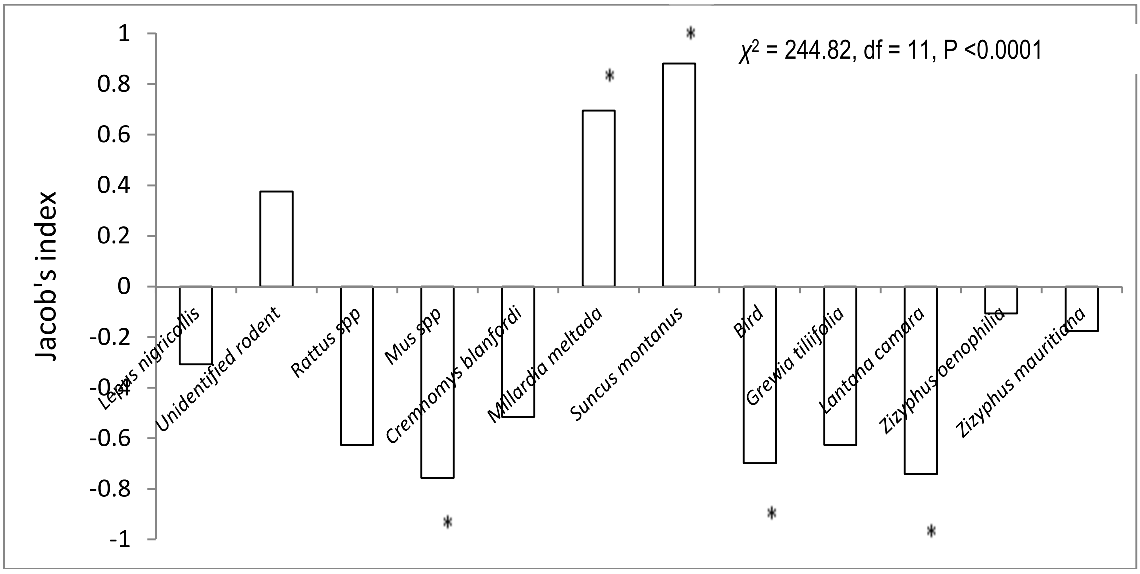
$^{\$}$ = Dry season, $^{\#}$ = Wet season; Selection; 1 = used greater than expected; 2 = proportional use; 3 = used less than expected

Fig 35a. Consumption of major prey items by mongoose in the dry season derived from Jacob's index (2009-2011).



Chi-square test demonstrated that there was a highly significant difference between the expected utilization of the food items and the observed frequency of use; * = significant positive or negative selection according to Bonferroni simultaneous confidence intervals at P ≤ 0.05

b. Consumption of major prey items by mongoose in the wet season derived from Jacob's index (2009-2011).



Chi-square test demonstrated that there was a highly significant difference between the expected utilization of the food items and the observed frequency of use; * = significant positive or negative selection according to Bonferroni simultaneous confidence intervals at P ≤ 0.05

4.4. Discussion

The most important food for small cats was small mammals, while mongooses consumed invertebrates frequently and the diet of civet mainly consisted of fruits. Besides the consumption of principal food items by predators, they also supplemented their diet with other food items. The general feeding habits appear to confirm the results from available studies. Small mammals contribute to the major biomass of numerous small carnivores which was also evident from this study. Thus, the difference in diet observed among habitats probably reflects the behaviour of predators which help them adjust to changes of prey availability with time. If predator choices are contingent upon prey attributes, such as prey availability and energy constraints, these prey attributes should affect predator choice.

The high percent biomass of small mammal and bird in small cat faeces explained the low trophic diversity in April (hottest month). Small cats consumed small mammals more frequently than civets and mongooses in the dry season. They selected *Tatera indica* and *Millardia meltada* in both seasons while *Suncus montanus* was preferred only in the dry season although there were no seasonal differences in their abundance. *Rattus spp* was significantly underrepresented in the dry season although it occurred in high abundance. *Tatera indica* and *Millardia meltada* occurred in low abundance in the study area. Moreover *Tatera indica* lives in colonies specifically in the thorn forest while the distribution of *Millardia meltada* is patchy across the study area, mainly in open grasslands or barren land. Throughout India, *Millardia meltada* is known to occupy crop fields while *Tatera indica*, *Millardia meltada* and *Mus spp* dominate even agricultural landscapes. The relative abundance and behavior of prey species is in turn influenced by factors such as reproductive schedules, vegetation cover, climate and weather, as well as agricultural practices. The minimal occurrence of fruits and seeds in small cat diet was not surprising as it has been reported in the diet of other cat species. Mukherjee et al., (2004) estimated that up to 70% of the daily metabolizable energy in the jungle cat and caracal is obtained from small rodents. The diet of Geoffrey cat as evaluated by faecal analysis, comprised of 42% waterbirds and 40% small mammals (Canepuccia et al., 2007). Diet composition of Geoffroy's cat in the Monte desert of central Argentina (Bisceglia et al., 2008) showed that small mammals constituted up to 94% of its diet. Based on direct observations on the Southern African wildcat (*Felis silvestris)* in the southern Kalahari, Herbst and Mills (2010) studied their seasonal food habits where its preferred prey was murid rodents. In eastern Scotland, wildcats fed almost exclusively on rabbits (Corbett 1979), and in the south and central Spain rabbits were consumed in greater frequency than rodents (Malo et al., 2004). Nevertheless, most of the studies conducted in Europe have reported that rodents, especially murines and microtines, are the main dietary items (Lozano et al., 2006). Bird species that spend time mainly on the ground are also potential sources of food for cats and mongoose. Although Pallas's cats used a broad range of food items, small mammals formed the majority of the diet, and pikas, insects, grasshoppers and beetles were also consumed (Ross et al., 2010). Small mammals are generally abundant in diverse ecosystems which increase the encounter rate with predators. They typically contain a greater percentage of digestible matter with respect to similar-sized birds or reptiles (Hume 2005). In consequence, small mammals and particularly small rodents comprise the bulk of the diet of several small-sized felids (< 7 kg of body weight) (Lozano et al., 2006, Sliwa 2006).

In the diet of civet, invertebrate remains occurred higher in deciduous forest than thorn forest. The choice of fruit species was comparatively less in deciduous forest than thorn and semi-evergreen forest because it may not have fulfilled optimum nutritional requirements for civets that could have prompted foraging even on invertebrates. Extreme environmental conditions could be responsible for food resource scarcity which could explain the low trophic diversity during the hottest month in the thorn forest, and during heavy monsoon in deciduous and semi-evergreen forests. Fruit biomass consumption was higher than invertebrates. Fruit occurrence was higher in the dry season than wet season since the period synchronized with the fruiting season of most food tree species. The wet season, on the other hand did not offer wide choices of fruit. Civet consumed fruits more than small cat and mongooses.

The year round fruit production of principal food plants of civet in Mudumalai suggests that they were tracking fruit production. The main fruit-bearing plants, potentially utilized by civets across most studies are members of the families, Euphorbiaceae, Moraceae, Rutaceae, Rubiaceace, Sapotaceae, Bombacaceae, Sapindaceae, Ebenaceae, Elaeocarpaceae, Myrsinaceae, Musaceae, Loganiaceae, Gnetaceae, Caricaceae, Anacardiaceae, Myristicaceae, Poaceae, Pandanaceae, Solanaceae, Loganiaceae, Myrtaceae, Caprifoliaceae, Rhamnaceae, Vitaceae, Celastraceae, Arecaceae, Musaceae, Passifloraceae, Flacourtiacea, Bromeliaceae and Cucurbitaceae. There was variation in fruit species occurrence in civet faeces across months. In tropical forests fruit availability prevails throughout the year (Terborgh 1986). Even among the fruit species that were monitored there were some that produced fruits seasonally and some were unpredictable. *Grewia tiliifolia*, *Lantana camara*, *Cassia fistula*, *Zizyphus oenophilia* and *Cordia obliqua* were keystone plants for the civet in Mudumalai, however of these *Grewia tiliifolia* and *Lantana camara* produced fruits throughout the year. These were identified as important food plants that could be available during periods of low fruit diversity a fruit tracking strategy which has also been reported in other studies (Mudappa 2001). Similar strategies have been expressed where locomotion in the common palm civet increased during the period of low food supply and when plants had a clumped distribution in Chitwan National Park (Joshi et al., 1995). The selection of fruit species varied in the three habitats probably because civets fed on fruits that were commonly available in the particular habitat. Dietary switching has been reported in brown palm civet (Mudappa 2001), masked palm civet (Zhou et al., 2008) and common palm civet (Joshi et al., 1995). Further study is needed to identify which traits of frugivores more or less tie their occurrence to fruit abundance. Civets showed strong preferences for certain species of fruits. High intra and inter-specific variation in fleshy fruit nutrient composition has been established (Wendeln et al., 2000), hence it is probable that civets meet their protein requirements by consuming high diversity of fruits and flowers (Mudappa 2001). Similarly brown palm civet meet their nutritional demands through low metabolic rates and low energy needs by switching their diet to the opportunistic consumption of invertebrates and rodents to supplement their diet (Mudappa et al., 2010). It is evident that although they are mostly frugivory in nature by physiology and anatomy, their diet could vary temporally according to the diversity and availability of food. I did not record flowers in feacal samples as opposed to being recorded in other studies (Mudappa 2001). Most plant species recorded in the diet from this study have been reported in earlier studies (Joshi et al., 1995, Mudappa 2001) although their percent contribution varied. Nocturnal feeding observations of palm civets from literature have recorded substances that are not quantifiable in the scats, including nectar and stem sap. Therefore, they may use a greater diversity of soft-tissue food that could not be documented by faecal analysis. Palm civets are opportunistic feeders and consume a diversity of plant and animal species in their diet. Fruits and berries were a major portion of the diet in this study. Civets did not simply track the abundance of food resources passively in their environment, although fruit was present throughout all sampling periods in this tropical environment. Fruits generally have relatively low protein content but the ease with which they can be procured, relative to time and effort invested, allow civets to meet altleast minimal protein requirements (Rode and Robbins 2000). The preference for fleshy fruit may result from fruit characteristics that influence the plant–vertebrate seed disperser relationship. Herrera (1989) argues that carnivore-dispersed plants tend to have large, pulp rich (high content of fibre and low proteins and minerals) and scented fruits that generally fall to the ground after ripening. These features, together with a low number of seeds per pulp unit increase the conspicuousness of fruits and reduce the probability of seeds being destroyed by carnivore teeth, thus ensuring that effective dispersal-vectors do not destroy seeds.

The presence of grass/herbs leaves in the faeces is consistent with other studies, mostly concluding a possible role of leaves in scouring the intestine during digestion. Common Palm Civet selected at least 18 fruit species in Kerala, mostly from non-native plants (Nayar et al., 2006). Krishnakumar and Balakrishnan (2003) identified 10 fruit species from Common Palm Civet faeces in two semi-urban habitats in Trivandrum, Grassman (1998) recorded 13 fruit species in faeces pooled from Common and Masked Palm Civet (*Paguma larvata*) in Kaeng Krachan National Park (Thailand) and Su Su and Sale (2007) recorded about 31 types of fruits in Common Palm Civet faeces analysed from Hlawga, Myanmar. Mudappa (2001) recorded 57 fruit species in the diet of brown palm civet in rainforests of south India. Su Su and Sale (2007), in Myanmar, found only few vertebrate and invertebrate remains in the diet of *P. hermaphroditus*. The overall diet diversity of civets was low in Mudumalai compared to Kalakad-Mundanthurai Tiger Reserve probably because the latter is a rainforest with relatively high plant species richness (Mudappa 2001). It was evident that civets were tracking fruiting episodes as shown in other studies (Mudappa 2001). Fruits in tropical rainforests are fleshy, offering wider choices throughout the year and superabundant (Terborgh 1986). The physiological adaptations of viverrids have not been fully explored although they seem to make up for low quality diet by being opportunistic and by their rapid gut passage time or low metabolic rate (Mudappa 2001).

Small mammals were the main dietary component of mongoose followed by invertebrates. Mongooses consumed bird remains more in thorn forest than semi-evergreen forest while reptiles were consumed higher in deciduous than thorn and semi-evergreen forest. Trophic diversity and niche breadth was higher in deciduous probably since the type of food preferred (mainly invertebrates) by mongooses could be most abundant in these forests. In the dry season, mongoose consumed invertebrates more than small cats while it did not differ from civet suggesting substantial diet overlap between mongoose and civet in Mudumalai. Mongooses consumed pilli millipedes and dung beetles in greater frequency than other carnivores. They selected *Millardia meltada* positively and significantly in both seasons while it also selected *Suncus montanus* in the wet season.

Mongooses are ground-foraging burrowing species that often live in underground burrows or in dense shrub cover. Some populations are largely insectivorous; others may largely consume fruits for parts of the year. The mean percent frequency of the food items from the faecal pellets has shown that the major food component (more than 65%) of the diet of the small Indian mongoose were rodents (Mahmood et al., 2011). They place their noses to the earth sniffing until they smell an insect and then they either snatch it up as the bug meanders along above ground or digs it out from below the dirt (Osborn 1998). Parts of the insects and a scorpion were found in the stomachs of two small Indian mongooses collected in the Rajasthan Desert in India (Prakash 1959). So findings in literature confirm results of the present study that mongooses were opportunistic with varied diet. The high frequency occurrence of pilli millipedes and insects in the diet of mongoose was probably due their dentition which was designed to crush food items that are calcareous.

Competition often occurs when two species share limited resources and results in each species exerting a negative effect on the other. There was significant dietary partitioning in selected food categories in civets during dry and wet season. Fruit occurrence was higher in the dry season than wet season due to high availability in the dry season. Small cats consumed small mammals more frequently than civets resulting in low niche overlap with civet however there was substantial overlap with mongoose. Mongoose consumed invertebrates more than small cats however it did not differ from civet explaining the high niche overlap between mongoose and civet. In the wet season, mongoose and civet consumed more invertebrates resulting in high niche overlap. Invertebrates are probably accessible to carnivores during wet season due to their increased activity. While invertebrates represented the most-frequently consumed food item in mongoose, they often consumed rodents. Large mammal remains found in faeces probably represented carrion from dead animals in most cases. Overall diversity was higher in civets compared to small cats and mongoose due to the variety of fruit species selected. Each carnivore community exhibited moderate overall niche breadth in Mudumalai.

The abundance of their primary prey item could possibly allow them to coexist without competing for food, although periods of food shortage might increase the food-niche overlap but this may not necessarily indicate competition among the three communities. The food items selected by lesser carnivores are generally abundant in the environment yet not all food items can be quantified easily due to high diversity. Future research must include quantification of invertebrates for clarification in the true prey selection by small carnivores, although this would still be quite challenging. Future studies addressing prey activity and their influence on the feeding habits and use of space and time could shed light upon the mechanisms involved in both prey and predators. Although this study could not address feeding habits at the species level, it was useful in providing information on dietary trends at the community level. In the future feeding ecology should be addressed at the species level to reveal interesting feeding strategies exhibited by each species.

Chapter 5

HABITAT USE OF SMALL CARNIVORES

5.1 Introduction

A fundamental problem in ecology is the issue of scale, and in particular, the appropriate scale at which research needs to be focused to ascertain the dynamics that function to drive and organize a community (Levin 1992). Questions related to patterns of species distribution and occupancy need to focus on processes that may operate at multiple spatial scales. A fundamental concept in animal ecology is that each species occurs within a limited range of environmental conditions, defining its habitat niche (Hutchinson 1957). Some species have peculiar habitat preferences. The presence of a species and its association with habitat characteristics at different spatial scales is a function of vegetation structure and food availability (Lantschner et al., 2012) that may change over space and time. For small sized carnivores, presence may not be directly related to habitat type, but to the complexity or heterogeneity of vegetation structure and landscape attributes. The "continuum model" (Fischer and Lindenmayer 2006) assumes each species response to their environment and suggests gradual change in habitat quality through space. The model conceptualizes landscape as varying across gradients of food, shelter, space, and climate, which may be defined with respect to habitat variables important for individual species. From a conservation standpoint, this model focuses on habitat heterogeneity to enhance the number of niches available for different species. Moreover, interspecific competition among Indian carnivores, killing and/or predation of smaller by larger carnivores can have marked effects on distribution, habitat use and/or abundance of non-dominant species (Ramesh 2010). Trophic-niche diversity may favour carnivore species coexistence. Greater plant species richness also promotes higher richness of frugivores probably mediated by the trophic chain, small mammals (Andrews and O'Brien 2000). Habitat use by lesser carnivores has been well investigated using a variety of methods (Sarmento et al., 2010, Gupta 2011). For instance, in rain forests, fluctuations in limited and unreliable food resources are probably influential on species occurrences, and each species can be assumed to respond uniquely to any combination of environmental changes that may alter the relative competitive status of each species in the community (Morris et al., 2000). The apparent suitability of the environment for different carnivores can be explained with reference to the characteristics of microclimate and level of resources usually present. These characteristics satisfy the need for shelter (shrub cover, hollow trees), food (diverse communities of small mammals, birds, fleshy fruits), and anti-predator cover, which are important aspects for the survival. Behavioural responses of small carnivores offer several possible reasons for seasonal differences in occupancy. Few have addressed seasonal habitat use of small carnivores in an occupancy framework (Kirk and Zielinski 2009). Spatial structure of frugivore carnivore populations appears to be influenced by the occurrence of fruit-rich patches (Rosalino et al., 2004). It is suggested that the high habitat heterogeneity in sites may contribute to a higher degree of close co-existence between sympatric species. Microclimatic factors have been put forward as a key to habitat selection of species of marten (Buskirk and Powell 1994) and badger (Neal 1986). Even if the same habitat space is used by similar species, there might be some segregation due to preferences of particular sites.

One of the main objectives in community ecology is to understand the conditions allowing species to co-exist, which requires identifying how co-occurring species use and share space and resources. In the space composed of 'n' dimensions accounting for the many environmental variables (ecological space), the niche is conveniently (Hutchinson 1957) defines the conditions and resources necessary for the species to persist. Both the spatial structures, geo-referenced data and biotic interactions should be studied together, as multiple species can select similar resources but be distributed differentially across space. Although the advantages of this approach have been extensively reviewed, very little attention has been given to scaling issues when selecting variables for inclusion in candidate models. Most past studies on wildlife habitat relationships focused on identifying important habitat variables. Animals often hierarchically select habitats; therefore, it is important to evaluate habitat selection at a range of spatial scales in order to reveal the true grain at which the animal responds within the site (Kotliar and Wiens 1990).

There are modern techniques of enumerating distribution factors at multiple scales and as far as the elusive, nocturnal, lesser carnivores are concerned, the convenient method of monitoring is with the application of site occupancy models. Among current monitoring methods, camera-trapping has proven to be successful in determining carnivore occurrence at multiple spatial locations (Sarmento et al., 2010, Lantschner et al., 2012). How does the forest matrix fulfill important requirements of lesser carnivore communities should be investigated much in detail following the occupancy framework. Site occupancy provides a reasonable estimate of population status and trends, and it also provides an unbiased, cost-effective alternative method for large-scale, multispecies monitoring programs (MacKenzie et al., 2002). To correct for biases in comparisons between photographic capture frequencies of species, information is needed about the relation between capture probability and macro/micro-habitat variables surrounding a camera location. There is abundant evidence that small mammals respond to forest loss or are sensitive to the configuration of forest patches within a non-forest matrix (Mudappa et al., 2007). Spatial ecology of small carnivores has been studied in detail (Mukherjee et al., 2010, Gupta 2011). Differences between study areas in habitat quality, food availability, interspecific/intra-guild competition might be contributory factors as well as differences in habitat utilization and diet (Jennings et al., 2011). Habitat use of small carnivores is poorly known in India (Kumar and Umapathy 1999, Mudappa 2001, Joshi et al., 1995). Linking models to field data like vegetation complexity, food diversity, resting and denning characteristics would be useful at the site level.

In order to address how differences in habitat scale influence small carnivore occupancy, I applied a rigorous statistical approach to model site occupancy and detection probability based on mark–recapture theory (MacKenzie et al., 2002). I estimated the probability that a site is usable (i.e. that a species may use the site), which I defined here as occupancy (ψ). I aimed to use camera trap data to determine carnivore occupancy as a function of various site variables hypothesized to influence the probability of occupancy in Mudumalai during dry and wet seasons. Occupancy modeling is a robust analysis that uses presence/absence data to estimate the probability of occurrence (ψ—also probability of use) by incorporating an additional parameter of detection probability ($\hat{p}$-MacKenzie et al., 2006). This approach also applies regression analysis via the logit link to model species occurrence and detection probabilities as a function of habitat and site-specific covariates. Multiple models may be explored and by ranking these models using an information-theoretic approach, I examined the relative support and strength of evidence for each model based on the data. The information criterion used is Akaike's Information Criterion (AIC) which uses the maximized log-likelihood to estimate the information distance between the best approximating model and the true generating mechanism (Anderson 2008). The criterion also allows us to use multiple models in our working hypothesis through model-averaging parameter estimates (Burnham and Anderson 1998). Occupancy findings can be used to draw inferences about habitat use, selection and predict occurrence at multiple scales.

The objectives of this study are to: (1) identify the variables that most strongly influence habitat selection by each species and the scale at which each of these variables is most important (2) assess differences in habitat-use by carnivore species in dry and wet seasons (3) compare the resultant species occupancy at both spatial scales in the dry season. Finally I concluded my results by compiling remotely sensed and micro-scale variables thought to drive carnivore distribution. This chapter focused on habitat spectrums that shape the small carnivore guild in Mudumalai. It allowed me to test hypotheses about plausible determinants of species distribution patterns wherein I expect effects of different combinations of covariates to vary as per the biological needs of small carnivore species.

5.2 Materials and methods

5.2.1 *Camera trapping*

I selected 114 km^2 of study area which was divided into three intensive sampling zones; deciduous (35 km^2), semi-evergreen (40 km^2) and the dry thorn forest (39 km^2) to conduct camera trap surveys for two years (2009-2010 and 2010-2011, Fig 4 in chapter 3). Camera trap survey was carried out in the dry and wet season in deciduous and dry thorn forest while the semi-evergreen forest was sampled only in the dry season due to inaccessibility in the wet season. I conducted a thorough preliminary survey to identify suitable sites where indirect evidence, such as tracks or scats was detected. I deployed passive-infrared camera traps in a systematic grid 1 km x 1 km using Deercam DC300 (DeerCam, Park Falls, USA) and Stealthcam (Bedford, Texas, USA). The mean inter-camera trap distance was 1.31 km. Each year I deployed 25 pairs of camera traps in the deciduous forest, 21 in the semi-evergreen forest and 26 in the dry thorn forest. Stations consisted of two independently operating passive-infrared cameras mounted on opposite sides of a trail or dirt road to get photocaptures of small carnivores. Cameras were placed approximately 25 cm above the ground, 5-8 m away from the centre of the trail and set to be active for 24 h/day. Cameras were placed on roads, trails, stream-beds, near water holes, termite mounds and fruiting trees to maximize lesser carnivore photo captures. No bait or lure was used at any location to attract animals. The photocapture delay was set to 1 min and sensitivity was set to high. I checked sampling stations on an average, every 3 days to ensure continued operation and replaced batteries and film when necessary. Photographs provided information on the date and time of the picture taken. Although some cameras were damaged by elephants (*Elephas maximus*), I replaced them with new ones to avoid loss of data. Each camera trap survey was sampled for 30 days in each habitat. The 30 days trapping duration was divided into 10 sampling occasions of 3 days each for further analysis.

5.2.2 *Non-spatial/microscale measurements*

In most field situations, occupancy and detection probabilities are not constant across all sample units (Royle et al., 2005) but instead vary by site characteristics. I selected covariates most likely to influence the distribution of small carnivores based on their ecological background from available literature (Kumar and Umapathy 1999, Mudappa et al., 2010, Mukherjee et al., 2010).

Continuous variables: I measured microhabitat features in 20 m diameter plots centered around each camera station. Variables were measured separately in dry and wet seasons for two years hence the mean of the values was used for further analyses. The number of *Grewia tiliifolia, Zizyphus spp* trees and active termite mounds were recorded. Small mammal richness (from live trapping data as explained in chapter 2) and fruiting tree diversity were also recorded. I noted only those fruiting trees that comprised ≥1% of the diet in lesser carnivores (chapter 4). Cover types; canopy cover, shrub cover (≤1 cm DBH and , ≤ 1 m height), rock cover, ground cover, coffee shade-understory cover was estimated as ≤25%, 26–50%, 51–75%, or ≥75%. I measured the distance from each camera station to the nearest water source and village/settlement (considered as disturbance index) using a handheld global positioning system (GPS) device.

Categorical variable:- included presence/absence of major habitat types (dry thorn, deciduous, semi-evergreen and riparian), terrain type (plain, gentle slope, undulating and steep), prey species (*Tatera indica*, *Millardia meltada*), liana and vine (irrespective of species) at each camera station.

5.2.3 *Spatial/macroscale measurements*

Field-based measurements provide detailed information over a set of sites that form a small proportion of the study landscape while remotely sensed data provide information at broad spatial scales for the entire landscape of interest. To measure remotely sensed variables, I used the Geographic Information System (GIS) in ArcGIS 9.3 (Environmental Systems Research Institute [ESRI], Inc., Redlands, CA, USA) for data extraction. I created comma-separated value (.csv) file from longitude and latitude coordinates of each camera trap location from an Excel spreadsheet. Among the many indices available, the normalized difference vegetation index (NDVI) shows the best correlation for a variety of vegetation-related parameters. Hence, I used remotely sensed mean NDVI values to explore species–habitat relationships. Mean NDVI corresponds to tree density and canopy cover irrespective of seasonal changes.

Digital elevation data SRTM 30 m resolution was obtained from DIVAGIS website (version 7.1.7.2, http://www.diva-gis.org). Slope and aspect was calculated from elevation using the Surface analysis option in Spatial Analyst toolbox.

I obtained 19 'bioclimatic' variables based on the global climate data sets developed by Hijmans et al., (2005). These GIS data sets characterize global climates using average monthly weather station data and are available at 30 arc-seconds resolution (approximately 1 km^2). However since most climatic variables were highly correlated (R^2 = >0.5), I used only independent climatic variables; isothermality (mean diurnal temperature range/[maximum temperature of warmest month/minimum temperature of coldest month]), max temperature of the warmest month, precipitation of warmest quarter and precipitation of coldest quarter. The WorldClim parameters express spatial variation in annual means, seasonality and extreme or limiting climatic factors and represent biologically meaningful variables for characterizing species distributions.

Surface water bodies (rivers and streams) were extracted for country wide data from DIVAGIS (version 7.1.7.2, http://www.diva-gis.org) along with field data locations of potential water sources. I used the Euclidean distance tool to create a raster "distance to" (km) layer for the closest water source and village settlement. This tool calculated distance from each camera location within the raster layer a straight line distance to the nearest source variable of interest. Locations of villages/settlements were also plotted and Euclidean distance tool was applied to measure distance of each pixel to the nearest village.

Elevation, slope, aspect and distance covariates were extracted at a 30 m x 30 m resolution having a single value per site by averaging all the pixel values within each site.

The continuous remote variables were all mean values for each camera location extracted from the relevant layers using the Zonal statistics tool in ArcGIS 9.3 (ESRI, Inc., Redlands, California, USA). These abiotic variables were then analyzed together with the species detection history only for the dry season for comparisons with the microscale–dry season occupancy estimates. These values were used as covariates to create candidate models for occupancy and habitat use.

Table 22. Micro and macro-scale covariates and *a priori* predictions about their influence on habitat use of small carnivores in Mudumalai.

Covariate category	**Variables and abbreviations**	**Predictions for occupancy**
Non-spatial variables		
1. Forest type-Categorical	Presence/absence of dry thorn (dtf), deciduous (dec), semi-evergreen (sem) and riparian habitat (riparian)	'dtf' positively influences grey mongoose, jungle cat and rusty-spotted cat. 'dec' positively influences ruddy, stripe-necked mongoose. 'sem' influences brown palm civet. No forest type would influence small Indian civet. 'riparian' positively influences common palm civet.
2. Terrain type-Categorical	Presence/absence of plain (p), gentle slope (gs), undulating (u) and steep (s)	'p' and 'gs' positively influences grey, ruddy mongoose and jungle cat, 'u' positively influences small Indian civet and stripe-necked mongoose, 's' positively influences rusty-spotted cat and brown palm civet.
3. Food-Categorical	Presence/absence of *Tatera indica* (*tatera*), *Millardia meltada* (*millardia*), liana (L) and vine (V).	'*tatera*' and '*millardia*' positively influences jungle cat, rusty-spotted cat and mongooses except others. 'L' and 'V' positively influences civets except others.
4. Food-Continuous	Number of *Grewia tillifoila*	'*grewia*' influences civets and '*zizyphus*' influences

	(*grewia*) and *Zizyphus spp* (*zizyphus*) trees	mongooses except other species
5. Food-Continuous	Fruiting tree diversity (ftd)	'ftd' positively influences civets
6. Food-Continuous	Rodent species richness (rr)	'rr' positively influences cats and mongooses
6. Food-Continuous	Number of termite mounds (termite)	'termite' positively influences mongooses
5. Distance (km) to nearest village/settlement and/or water source-Continuous	Measured distance using global positioning system (GPS) from camera location to nearest village/settlement (d2v) and water source (d2w)	Both negatively influence jungle cat, rusty-spotted cat, grey mongoose, ruddy mongoose, common palm civet and small Indian civet.
7. Cover-Continuous	Rock cover (rc), canopy cover (cc), ground cover (gc), shrub cover (sc), coffee-shade understorey (*coffee*). All cover types were measured as percentage ocularly.	'rc' positively influences jungle cat and rusty-spotted cat. 'cc', 'gc', 'sc' positively influences mongooses and civets. '*coffee*' positively influences brown palm civet.
Spatial variables **(continuous)**		
1. Isothermality (˚C)	bio3	'bio3' positively influences small felids, grey and ruddy mongoose
2. Max temperature of warmest month (˚C)	bio5	'bio5' positively influences small felids, grey and ruddy mongoose
3. Precipitation of Warmest Quarter (mm)	bio18	'bio18' positively influences civets
4. Precipitation of coldest quarter (mm)	bio19	'bio19' positively influences stripe-necked mongoose
5. Elevation (masl)	alt	'alt' negatively influences small felids and positively influences brown palm civet.
6. Slope (degrees)	slope	'slope' positively influences rusty-spotted cat, common palm civet, brown palm civet and stripe-necked mongoose
7. Aspect (degrees)	aspect	'aspect' positively influences common palm civet and brown palm civet
8. Euclidean distance (km) to nearest village/settlement and water source	d2v and d2w	Both negatively influence jungle cat, rusty-spotted cat, grey mongoose, ruddy mongoose, common palm civet and small Indian civet.

5.3 *Data analysis*

5.3.1 *Carnivore occupancy*

For each species, I fitted multiple models, representing different hypotheses about the processes that generated the data.

At each study site, carnivore occupancy was estimated using likelihood based method (MacKenzie et al., 2002). I created carnivore detection history (e.g., 1100100) for each camera station, consisting of binary values with '1' indicating species detection during the sampling occasion and '0' indicating non-detection (Otis et al., 1978). Detection histories for each species except for leopard cat were created separately for dry and wet season. Occupancy surveys described in MacKenzie et al., (2002) and Royle and Nichols (2003) use sample units as sites. Implicitly, it is assumed that each site is independent and no animal will move between sites during the survey period unless the movement of animals is very small compared to the selected grid size. Hence, setting up a grid system of 1 km x 1 km for small carnivores and using these models may minimize the violation of occupancy closure at each site due to their small home ranges. The survey duration of 30 days was divided into 10 sampling periods of 3 days each to increase detection probability for each sampling period. Because I considered each camera station an independent site, each sampling period was a temporal repeat of the survey. Detection histories were pooled for two years from the 3 study sites and entered together as single-season models in PRESENCE v.4 (Proteus Wildlife Research Consultants, New Zealand; http://www.proteus.co.nz). Although the data were collected over two years, the grouping of data did not violate any of the assumptions of the modeling process.

5.3.2 *Model set development*

I defined a number of candidate models incorporating micro and macro variables (covariates) based on *a priori* hypotheses and available literature regarding factors that could influence site occupancy and detection probability (Table 22). Models were run separately for dry and wet season sampling using covariates and camera trapping detection histories for each season since different variables could influence species occupancy and habitat use in different seasons allowing me to explore the power of covariates for different seasons. Since correlation between variables can lead to model over-fitting, I computed Pearson's correlation coefficient (r) between each pair of variables, using SPSS 16.0 statistical software following which I selected only independent variables keeping $r = 0.5$ as the cut off value.

I standardized all continuous variables (micro and macro-scale) to *z*-scores prior to analysis (Cooch and White 2005). As originally conceived (MacKenzie et al., 2002), ψ is interpreted as the overall proportion of a study area that is used by a given species (MacKenzie et al., 2006). I produced a global model that contained all potential covariates for occupancy and allowed detection probability (p) to vary by all covariates. I followed a 2-step procedure where I first modeled p and then modeled ψ. The potential covariates for occupancy were then allowed to vary, individually or in combination, while detection was either maintained in the global model or remained constant, i.e. ψ(covariate)p(covariate), or ψ(covariate)p(.). I also considered the simplest model, where both occupancy and detection probability remained constant, $\psi(.)p(.)$. Hence, these covariates were used to model site occupancy. Model selection, computation of model weights, and averaging of parameters followed the framework of Burnham and Anderson (1998). I examined covariates that best explained overall species occupancy (ψ). I ran 10,000 bootstraps and Pearson goodness-of-fit tests to assess the fit of the models to the data (Mackenzie and Bailey 2004). I eliminated from the candidate set models that did not result in convergence or for which convergence was suspected because of inestimable parameters. Because the ratio of sample sizes (n) to the maximum number of estimated parameters (k) was < 40, I ranked models according to AIC_c (AIC adjusted for small sample size, Burnham and Anderson 1998). I addressed cases of poor model fit (i.e., the model chi-square value was >95% of the bootstrap values) by estimating an overdispersion factor ($\hat{c}$), inflating standard errors by a factor of $\sqrt{\hat{c}}$, and using a quasi-corrected AIC_c ($QAIC_c$) for model selection (Burnham and Anderson 1998). I calculated model-averaged parameters using Akaike weights for proportion of sites used and detection probabilities. To infer the relative influence of each covariate on occurrence, model weights were summed over all models containing the particular covariate. Akaike weights are equivalent to Bayesian posterior model probabilities and indicate the relative support of a model (Wintle et al., 2003). Finally I compared species-specific site occupancies between micro and macro-scale variables from estimates achieved from dry season modeling to maintain uniformity in comparisons. Site occupancy of species could not be compared between seasons due to variation in the sampled sites, however I emphasized on how different variables influenced species-occupancy in different seasons.

5.4 Results

I found significant differences in the apparent nature and strength of relationships between species occurrence and a number of habitat variables across the two scales.

5.4.1. *Microscale variables*

Dry season*:* Naïve occupancy in the dry season ranged from 0.09 for rusty spotted cat to 0.38 for small Indian civet (Table 23). Individual species predicted site occupancy (model averaged) ranged from as low as 0.25(± 0.02) for common palm civet to 0.63(± 0.08) for stripe necked mongoose. Predicted detection probability ranged from 0.03(± 0.03) for rusty spotted cat to 0.15(± 0.03) for small Indian civet.

Wet season*:* Individual species predicted site occupancy (model averaged) ranged from as low as 0.06 for stripe necked mongoose to 0.36 for small Indian civet (Table 24). Individual species predicted site occupancy (model averaged) ranged from 0.20(± 0.08) for rusty-spotted cat to 0.69(± 0.10) for small Indian civet. Predicted detection probability ranged from 0.05(± 0.05) for rusty spotted cat to 0.26(± 0.05) for ruddy mongoose.

5.4.2. *Macroscale variables*

Individual species predicted site occupancy (model averaged) ranged from 0.20(± 0.08) for rusty spotted cat to 0.69(± 0.10) for small Indian civet (Table 25). Predicted detection probability ranged from 0.06(± 0.03) for rusty spotted cat and common palm civet to 0.23(± 0.06) for ruddy mongoose.

5.4.3. *Individual species estimates*

***Jungle cat*:** Model-averaged occurrence (*Ψ*) as estimated from micro and macroscale was higher than the naïve estimate 0.18 (dry season) and 0.24 (wet season). After accounting for detection probability, the estimated occupancy was 0.58(± 0.10) in the dry season and 0.37(± 0.12) in the wet season. Macroscale models generated occupancy estimate of 0.31(± 0.10) which was lower than microscale model estimate. The top-ranked model for jungle cat in the dry season (microscale) $\Delta AIC_C = 0.00$, included rock cover and canopy cover (Table 23) and had an Akaike weight (w_i) = 0.20, implying a 20% chance of being the best of the models evaluated. Of the microscale variables measured, three variables (rock cover, canopy cover and shrub cover) were substantially associated with jungle cat occurrence. Rock cover was positively associated with jungle cat occurrence while canopy cover and shrub cover were negatively associated (Table 26). The top ranked model for jungle cat in the wet season (microscale) $\Delta AIC_C = 0.00$ included the thorn forest (Table 24) and had an Akaike weight (w_i) = 0.17 implying a 17% chance of being the best of the models evaluated. Of these sets of variables, three variables were substantially associated with jungle cat occurrence. Thorn forest was positively associated, while canopy cover and ground cover were negatively associated with jungle cat occurrence (Table 27). The top ranked model for jungle cat in the dry season (macroscale) $\Delta AIC_C = 0.00$, included distance to village (Table 25) with an Akaike weight (w_i) = 0.09, implying only 9% chance of being the best of the models evaluated. Of these set of variables, distance to village, precipitation of the warmest quarter and NDVI were substantially associated with jungle cat occurrence. Distance to village and precipitation of the warmest quarter were negatively associated with jungle cat occurrence (Table 28).

***Rusty spotted cat**:* Model-averaged occurrence (*Ψ*) as estimated from micro and macroscale was higher than the naïve estimate 0.09 (dry season) and 0.08 (wet season). After accounting for detection probability, the estimated occupancy was 0.35(± 0.17) in the dry season and 0.20(± 0.18) in the wet season. Macroscale models generated occupancy estimate of 0.20(± 0.08) which was lower than microscale (dry season) estimate. The top-ranked model for rusty spotted cat in the dry season (microscale) $\Delta AIC_C = 0.00$, included thorn forest (Table 23) with an Akaike weight

(w_i) = 0.11 implying 11% chance of being the best of the models evaluated. Of these set of variables, thorn forest was the only variable with substantial positive association with rusty spotted cat occurrence (Table 26). The top ranked model for rusty spotted cat in the wet season (microscale) $\Delta AIC_C = 0.00$, included *Tatera indica* (Table 24) with an Akaike weight (w_i) = 0.11 implying 11% chance of being the best of the models evaluated. Of these set of variables, *Tatera indica* and shrub cover were substantially associated with rusty spotted cat occurrence. *Tatera indica* was positively associated while shrub cover was negatively associated with rusty spotted cat occurrence (Table 27). The top ranked model for rusty spotted cat in the dry season (macroscale) $\Delta AIC_C = 0.00$, included precipitation of warmest quarter and slope with an Akaike weight (w_i) = 0.21 (Table 25) implying 21% chance of being the best of the models evaluated. Of these set of variables, both, precipitation of warmest quarter and slope were negatively associated with rusty spotted cat occurrence (Table 28).

***Small Indian civet**:* Model-averaged occurrence (Ψ) as estimated from micro and macroscale was higher than the naïve estimate 0.38 (dry season) and 0.36 (wet season). After accounting for detection probability, the estimated occupancy was 0.49(± 0.08) in the dry season and 0.55(± 0.10) in the wet season. Macroscale models generated occupancy estimate of 0.69(± 0.10) which was higher than microscale model estimates (dry season). The top-ranked model for small Indian civet in the dry season (microscale) $\Delta AIC_C = 0.00$, included ground cover, vine, canopy cover, thorn forest, distance to water and rodent richness (Table 23), with an Akaike weight (w_i) = 0.12, implying 12% chance of being the best of the models evaluated. Of the set of variables, ground cover, vine, thorn forest, distance to water and rodent richness were positively associated while canopy cover was negatively associated with small Indian civet occurrence (Table 26). The top ranked model for small Indian civet in the wet season (microscale) $\Delta AIC_C = 0.00$, included thorn forest, distance to water, shrub cover and *Grewia tiliifolia* (Table 24) with an Akaike weight (w_i) = 0.55 implying 55% chance of being the best of the models evaluated. In this model, thorn forest, distance to water, shrub cover and *Grewia tiliifolia* were positively associated with small Indian civet occurrence (Table 27). The top ranked model for small Indian civet in the dry season (macroscale) $\Delta AIC_C = 0.00$, included precipitation of coldest quarter (Table 25) with an Akaike weight (w_i) = 0.36, implying 36% chance of being the best of the models evaluated. In this model, precipitation of coldest quarter was positively associated with small Indian civet occurrence (Table 28).

***Common palm civet**:* Model-averaged occurrence (*Ψ*) as estimated from micro and macroscale was higher than the naïve estimate 0.17(dry season) and 0.14(wet season). After accounting for detection probability, the estimated occupancy was 0.25(± 0.02) in the dry season and 0.21(± 0.09) in the wet season. Macroscale models generated occupancy estimate of 0.34(± 0.14) which was higher than microscale-dry season estimate. The top-ranked model for common palm civet in the dry season (microscale) $\Delta AIC_C = 0.00$, included riparian and distance to village/settlement (Table 23), with an Akaike weight (w_i) = 0.13, implying 13% chance of being the best of the models evaluated. Of the set of variables, riparian was positively associated while canopy cover and distance to village were negatively associated with common palm civet occurrence (Table 26). The top ranked model for common palm civet in the wet season (microscale) $\Delta AIC_C = 0.00$, included distance to village and plain, with an Akaike weight (w_i) = 0.11, implying 11% chance of being the best of the models evaluated (Table 24). In the top ranked models, undulating terrain was positively associated while plain, shrub cover and distance to village were negatively associated with common palm civet occurrence (Table 27). The top ranked model for common palm civet in the dry season (macroscale) $\Delta AIC_C = 0.00$, included precipitation of warmest quarter and aspect, with an Akaike weight (w_i) = 0.18, implying 18% chance of being the best of the models evaluated (Table 25). Of the set of variables, aspect was positively associated while precipitation of warmest quarter was negatively associated with common palm civet occurrence (Table 28).

Brown palm civet: Model-averaged occurrence (*Ψ*) as estimated from micro and macroscale was higher than the naïve estimate, 0.17 (dry season). After accounting for detection probability, the estimated occupancy was 0.36(± 0.18) in the dry season. Macroscale models generated occupancy estimate of 0.32(± 0.06) which was almost equal to microscale (dry season) estimate. The top-ranked model for brown palm civet in the dry season (microscale) $\Delta AIC_C = 0.00$, included steep, distance to village and distance to water, with an Akaike weight (w_i) = 0.27, implying 27% chance of being the best of the models evaluated (Table 23). Of the set of variables, distance to village and steep were positively associated while distance to water was negatively associated (Table 26). The top-ranked model for brown palm civet in the dry season (macroscale) $\Delta AIC_C = 0.00$, included aspect with an Akaike weight (w_i) = 0.27, implying 27% chance of being the best of the models evaluated (Table 25). In this model, aspect was positively associated with brown palm civet occurrence (Table 28).

***Stripe-necked mongoose**:* Model-averaged occurrence (Ψ) as estimated from micro and macroscale was higher than the naïve estimate, 0.32 (dry season) and 0.06 (wet season). After accounting for detection probability, the estimated occupancy was 0.63(± 0.08) in the dry season and 0.24(± 0.14) in the wet season. Macroscale models generated occupancy estimate of 0.43(± 0.10) which was lower than the microscale (dry season) estimate. The top-ranked model for stripe-necked mongoose in the dry season (microscale) $\Delta AIC_C = 0.00$, included rodent richness, canopy cover, and distance to water, with an Akaike weight (w_i) = 0.20, implying 20% chance of being the best of the models evaluated (Table 23). In this model, canopy cover, and distance to water were positively associated while rodent richness was negatively associated (Table 26). The top-ranked model for stripe necked mongoose in the wet season $\Delta AIC_C = 0.00$, included shrub cover and distance to water, with an Akaike weight (w_i) = 0.34, implying 34% chance of being the best of the models evaluated (Table 24). In the top ranking models, ground cover was positively associated while shrub cover and distance to water were negatively associated with stripe necked mongoose occurrence (Table 27). The top-ranked model for stripe necked mongoose in the dry season (macroscale) $\Delta AIC_C = 0.00$, included precipitation of coldest quarter and NDVI, with an Akaike weight (w_i) = 0.17, implying 17% chance of being the best of the models evaluated (Table 25). In the top ranking models, NDVI was positively associated while, precipitation of coldest quarter was negatively associated with stripe necked mongoose occurrence (Table 28).

***Ruddy mongoose**:* Model-averaged occurrence (Ψ) as estimated from micro and macroscale was higher than the naïve estimate, 0.31 (dry season) and 0.28 (wet season). After accounting for detection probability, the estimated occupancy was

0.61(± 0.12) in the dry season and 0.24(± 0.14) in the wet season. Macroscale models generated occupancy estimate of 0.33(± 0.07) which was lower than microscale model estimates. The top-ranked model for ruddy mongoose in the dry season (microscale) $\Delta AIC_C = 0.00$, included deciduous forest and ground cover, with an Akaike weight (w_i) = 0.25, implying 25% chance of being the best of the models evaluated (Table 23). In the top ranking models, deciduous and fruiting tree diversity were positively associated while ground cover was negatively associated with ruddy mongoose occurrence (Table 26). The top-ranked model for ruddy mongoose in the wet season $\Delta AIC_C = 0.00$, included distance to water and shrub cover, with an Akaike weight (w_i) = 0.09, implying 9% chance of being the best of the models evaluated (Table 24). In the top ranking models, plain and termite mound were positively associated while shrub cover, ground cover, distance to village and distance to water were negatively associated with ruddy mongoose occurrence (Table 27). The top-ranked model for ruddy mongoose in the dry season (macroscale) $\Delta AIC_C = 0.00$, included precipitation of warmest quarter with an Akaike weight (w_i) = 0.19, implying 19% chance of being the best of the models evaluated (Table 25). In this model, precipitation of warmest quarter was negatively associated with ruddy mongoose occurrence (Table 28).

***Grey mongoose**:* Model-averaged occurrence (Ψ) as estimated from micro and macroscale was higher than the naïve estimate, 0.21 (dry season) and 0.20 (wet season). After accounting for detection probability, the estimated occupancy was 0.51(± 0.11) in the dry season and 0.32(± 0.10) in the wet season. Macroscale models generated occupancy estimate of 0.30(± 0.13) which was lower than microscale model estimates. The top-ranked model for grey mongoose in the dry season (microscale) $\Delta AIC_C = 0.00$, included deciduous and canopy cover, with an Akaike weight (w_i) = 0.15, implying 15% chance of being the best of the models evaluated (Table 23). In the top ranking models, deciduous and termite mound were positively associated while dry thorn forest, canopy cover and rodent richness were negatively associated with grey mongoose occurrence (Table 26). The top-ranked model for grey mongoose in the wet season $\Delta AIC_C = 0.00$, included fruiting tree diversity and termite, with Akaike weight (w_i) = 0.15, implying 15% chance of being the best of the models evaluated (Table 24). In the top ranked models, fruiting tree diversity, termite, distance to water and distance to village were positively associated with grey mongoose occurrence (Table 27). The top-ranked model for grey mongoose in the dry season (macroscale) $\Delta AIC_C = 0.00$, included max temperature of warmest month and precipitation of warmest quarter, with Akaike weight (w_i) = 0.18, implying 18% chance of being the best of the models evaluated (Table 25). In the top ranked models, temperature of warmest month and precipitation of warmest quarter were negatively associated with grey mongoose occurrence (Table 28).

Table 23. Highest ranking models for microscale factors influencing site occupancy of small carnivores during the dry season in Mudumalai Tiger Reserve (2009-2011).

Species	**Model**	**AIC$_C$**	**ΔAIC$_C$**	**w_i**	**Model Likelihood**	**K**	**-2log-likelihood**	**Naïve ψ**	**ψ(± SE)**	**p(± SE)**
Jungle cat (55 models tested)										
JC1.	ψ(cc+rc),p(dtf+rc)	196.62	0.00	0.20	1.00	6	174.73	0.18	0.56(± 0.13)	0.09(± 0.02)
JC2.	ψ(sc+rc), p(dtf+rc)	196.93	0.31	0.17	0.86	6	175.04		0.61(± 0.08)	0.09(± 0.02)
	Model averaged								0.58(± 0.10)	0.09(± 0.02)
Rusty spotted cat (29 models tested)										
RSC1.	ψ(dtf),p(rr)	99.33	0.00	0.11	1.00	4	81.44	0.09	0.35(± 0.17)	0.03(± 0.03)
Small Indian civet (53 models tested)										
SI1.	ψ(gc+vine+cc+dtf+d2w+rr),p(.)	361.57	0.00	0.12	1.00	8	330.96	0.38	0.49(± 0.08)	0.15(± 0.02)
SI2.	ψ(gc+vine+cc+dtf),p(.)	361.71	0.14	0.11	0.93	6	335.10		0.48(± 0.08)	0.15(± 0.03)
SI3.	ψ(gc+vine+cc+dtf+d2w),p(.)	362.02	0.45	0.095	0.80	7	333.41		0.50(± 0.09)	0.15(± 0.02)
SI4.	ψ(gc+vine+cc+dtf+gs),p(.)	362.09	0.52	0.092	0.77	7	333.48		0.48(± 0.09)	0.15(± 0.03)
SI5.	ψ(gc+vine+cc),p(.)	362.48	0.91	0.076	0.63	5	337.87		0.48(± 0.08)	0.15(± 0.03)
	Model averaged								0.49(± 0.08)	0.15(± 0.03)
Common palm civet (54 models tested)										
CP1.	ψ(riparian+d2v),p(.)	168.37	0.00	0.13	1.00	4	149.04	0.17	0.22(± 0.07)	0.12(± 0.03)
CP2.	ψ(riparian+d2v),p(dtf)	168.73	0.36	0.11	0.83	5	147.40		0.25(± 0.13)	0.09(± 0.04)
CP3.	ψ(riparian+cc+d2v),p(.)	168.83	0.46	0.10	0.79	5	147.50		0.23(± 0.07)	0.12(± 0.03)

CP4.	ψ(riparian),p(dtf)	168.96	0.59	0.097	0.74	4	149.63		0.31(± 0.11)	0.13(± 0.04)
	Model averaged								0.25(± 0.02)	0.11(± 0.03)
Brown palm civet (44 models tested)										
BPC1.	ψ(steep+d2v+d2w),p(*coffee*+vine)	159.95	0.00	0.27	1.00	7	131.34	0.17	0.32(± 0.16)	0.06(± 0.02)
BPC2.	ψ(steep+d2v),p(*coffee*+vine)	161.15	1.20	0.15	0.55	6	134.54		0.37(± 0.22)	0.06(± 0.02)
BPC3.	ψ(steep+d2w),p(*coffee*+vine)	161.20	1.25	0.14	0.53	6	134.59		0.39(± 0.15)	0.05(± 0.02)
	Model averaged								0.36(± 0.18)	0.06(± 0.02)
Stripe-necked mongoose (90 models tested)										
SNM1.	ψ(rr+cc+d2w),p(gc+u)	300.54	0.00	0.20	1.00	7	275.21	0.32	0.63(± 0.08)	0.10(± 0.02)
Ruddy mongoose (87 models tested)										
RM1.	ψ(dec+gc),p(d2v+termite+steep)	296.93	0.00	0.25	1.00	7	273.04	0.31	0.62(± 0.09)	0.11(± 0.03)
RM2.	ψ(dec+ftd),p(d2v+termite+steep)	298.09	1.16	0.14	0.56	7	274.20		0.60(± 0.15)	0.12(± 0.03)
	Model averaged								0.61(± 0.12)	0.11(± 0.03)
Grey mongoose (102 models tested)										
GM1.	ψ(dec+cc),p(dtf+termite)	204.90	0.00	0.15	1.00	6	183.01	0.21	0.51(± 0.10)	0.08(± 0.02)
GM2.	ψ(dec+cc+termite),p(dtf+termite)	205.76	0.86	0.10	0.65	7	181.87		0.47(± 0.10)	0.08(± 0.03)
GM3.	ψ(dtf+cc+rr),p(dtf+termite)	205.89	0.99	0.09	0.61	7	182		0.55(± 0.14)	0.08(± 0.02)
	Model averaged								0.51(± 0.11)	0.08(± 0.02)

Information presented for each model includes AIC$_C$ (small-sample size correction to AIC) or QAIC$_C$(quasi-corrected AIC$_C$ with overdispersed data), ΔAIC$_C$(AIC$_c$ – min AIC$_c$), w_i (AIC weight for model i), k (the number of parameters in the model), ψ (probability of carnivore use) and p (detection probability). The (.) convention indicates constant across all sites.

Table 24. Highest ranking models for microscale factors influencing site occupancy of small carnivores during the wet season in Mudumalai Tiger Reserve (2009-2011).

Species	**Model**	**AIC_C**	**ΔAIC_C**	**w_i**	**Model Likelihood**	***K***	**-2log-likelihood**	**Naïve ψ**	**ψ(± SE)**	***p*(± SE)**
Jungle cat										
(82 models tested)										
JC3.	ψ(dtf),*p*(d2v+rc)	175.29	0.00	0.17	1.00	5	155.4	0.24	0.38(± 0.11)	0.13(± 0.04)
JC4.	ψ(dtf+cc),*p*(d2v+rc)	175.60	0.31	0.15	0.86	6	153.71		0.31(± 0.12)	0.13(± 0.05)
JC5.	ψ(gc),*p*(d2v+rc)	176.80	1.51	0.08	0.47	5	156.91		0.41(± 0.13)	0.13(± 0.04)
	Model averaged								0.37(± 0.12)	0.13(± 0.04)
Rusty spotted cat										
(21 models tested)										
RSC2.	ψ(*tatera*),*p*(.)	67.65	0.00	0.11	1.00	3	51.76	0.08	0.20(± 0.19)	0.05(± 0.05)
RSC3.	ψ(sc),*p*(.)	67.71	0.06	0.10	0.97	3	51.82		0.19(± 0.19)	0.05(± 0.05)
RSC4.	ψ(.),*p*(.)	68.01	0.36	0.09	0.83	2	54.12		0.20(± 0.17)	0.05(± 0.05)
RSC5.	ψ(*tatera*+sc),p(.)	68.43	0.78	0.07	0.68	4	50.54		0.15(± 0.14)	0.06(± 0.05)
RSC6.	ψ(.),*p*(p)	68.57	0.92	0.06	0.63	3	52.68		0.24(± 0.20)	0.04(± 0.04)
	Model averaged								0.20(± 0.18)	0.05(± 0.05)
Small Indian civet										
(55 models tested)										
SI6.	ψ(dtf+d2w+sc+*grewia*),*p*(.)	221.9	0.00	0.55	1.00	6	195.29	0.36	0.55(± 0.10)	0.10(± 0.02)
Common palm civet										
(82 models tested)										
CP5.	ψ(d2v+p),*p*(.)	110.49	0.00	0.11	1.00	4	91.16	0.14	0.20(± 0.10)	0.12(± 0.05)

CP6.	*ψ*(d2v+u),*p*(.)	110.6	0.11	0.10	0.95	4	91.27		0.18(± 0.07)	0.13(± 0.05)
CP7.	*ψ*(d2v+p+u),*p*(.)	111.34	0.85	0.07	0.65	5	90.01		0.18(± 0.08)	0.13(± 0.05)
CP8.	*ψ*(d2v+p+sc),*p*(.)	111.61	1.12	0.06	0.57	5	90.28		0.27(± 0.11)	0.09(± 0.04)
	Model averaged								0.21(± 0.09)	0.12(± 0.05)
Stripe-necked mongoose (59 models tested)										
SNM2.	*ψ*(sc+d2w),*p*(gc)	63.15	0.00	0.34	1.00	5	41.82	0.06	0.27(± 0.13)	0.22(± 0.04)
SNM3.	*ψ*(sc+d2w+gc),*p*(gc)	63.59	0.44	0.27	0.80	6	40.26		0.22(± 0.16)	0.22(± 0.04)
	Model averaged								0.24(± 0.14)	0.22(± 0.04)
Ruddy mongoose (51 models tested)										
RM3.	*ψ*(d2w+sc),*p*(rc)	221.16	0.00	0.09	1.00	5	201.27	0.28	0.32(± 0.09)	0.26(± 0.05)
RM4.	*ψ*(d2w+d2v),*p*(rc)	221.31	0.15	0.08	0.93	5	201.42		0.32(± 0.09)	0.26(± 0.05)
RM5.	*ψ*(d2w+termite),*p*(rc)	221.72	0.56	0.06	0.75	5	201.83		0.31(± 0.10)	0.26(± 0.05)
RM6.	*ψ*(d2w+sc+termite),*p*(rc)	221.74	0.58	0.065	0.74	6	199.85		0.32(± 0.10)	0.26(± 0.05)
RM7.	*ψ*(d2w),*p*(rc)	221.76	0.60	0.064	0.73	4	203.87		0.31(± 0.12)	0.26(± 0.05)
RM8.	*ψ*(d2w+gc),*p*(rc)	222.19	1.03	0.052	0.60	5	202.3		0.32(± 0.09)	0.26(± 0.05)
RM9.	*ψ*(d2w+p),*p*(rc)	222.22	1.06	0.051	0.59	5	202.33		0.31(± 0.09)	0.26(± 0.05)
RM10.	*ψ*(.),*p*(rc)	222.37	1.21	0.048	0.55	3	206.48		0.31(± 0.08)	0.26(± 0.05)
RM11.	*ψ*(sc),*p*(rc)	222.42	1.26	0.046	0.53	4	204.53		0.32(± 0.09)	0.26(± 0.05)
RM12.	*ψ*(d2v),*p*(rc)	222.61	1.45	0.042	0.48	4	204.72		0.31(± 0.08)	0.26(± 0.05)
	Model averaged								0.31(± 0.09)	0.26(± 0.05)
Grey mongoose (56 models tested)										
GM4.	*ψ*(ftd+termite),*p*(p)	144.5	0.00	0.15	1.00	5	124.61	0.20	0.31(± 0.09)	0.11(± 0.04)
GM5.	*ψ*(ftd+termite+d2v),*p*(p)	145.14	0.64	0.11	0.73	6	123.25		0.32(± 0.09)	0.11(± 0.04)

GM6.	ψ(ftd+d2w),p(p)	145.53	1.03	0.09	0.60	5	125.64		0.28(± 0.09)	0.11(± 0.04)
GM7.	ψ(ftd+d2v),p(p)	146.37	1.87	0.06	0.40	5	126.48		0.37(± 0.14)	0.10(± 0.04)
	Model averaged								0.32(± 0.10)	0.11(± 0.04)

Information presented for each model includes AIC_C (small-sample size correction to AIC) or $QAIC_C$(quasi-corrected AIC_C with overdispersed data), ΔAIC_C(AIC_C – min AIC_C), w_i (AIC weight for model i), k (the number of parameters in the model), ψ (probability of carnivore use) and p (detection probability). The (.) convention indicates constant across all sites.

Table 25. Highest ranking models for macroscale/remotely measured variables influencing site occupancy of small carnivores in Mudumalai Tiger Reserve (2009-2011).

Species	**Model**	**AIC_C**	**ΔAIC_C**	**w_i**	**Model Likelihood**	**K**	**-2log-likelihood**	**Naïve ψ**	**ψ(± SE)**	**p(± SE)**
Jungle cat (59 models tested)										
JC6.	ψ(d2v),p(bio19+alt)	197.89	0.00	0.09	1.00	5	182.61	0.18	0.30(± 0.11)	0.13(± 0.04)
JC7.	ψ(d2v),p(alt)	198.25	0.35	0.073	0.84	6	180.96		0.33(± 0.11)	0.13(± 0.03)
JC8.	ψ(d2v),p(bio19)	198.32	0.42	0.070	0.81	4	185.03		0.30(± 0.08)	0.13(± 0.03)
JC9.	ψ(d2v),p(bio19+bio18)	198.92	1.02	0.052	0.60	5	183.63		0.31(± 0.10)	0.12(± 0.03)
JC10.	ψ(bio18),p(bio19+alt)	199.54	1.64	0.038	0.44	5	184.25		0.33(± 0.10)	0.13(± 0.03)
	Model averaged								0.31(± 0.10)	0.13(± 0.03)
Rusty-spotted cat (45 models tested)										
RS7.	ψ(bio18+slope),p(.)	82.63	0.00	0.21	1.00	4	69.34	0.09	0.20(± 0.08)	0.06(± 0.03)
Small Indian civet (60 models tested)										
SI7.	ψ(bio19),p(bio19)	349.92	0.00	0.36	1.00	4	336.63	0.38	0.69(± 0.09)	0.14(± 0.03)

SI8.	ψ(bio19),p(bio19+bio3)	350.99	1.07	0.21	0.58	5	335.7		0.70(± 0.12)	0.14(± 0.03)
	Model averaged								0.69(± 0.10)	0.14(± 0.03)
Common palm civet (65 models tested)										
CP9.	ψ(bio18+aspect),p(bio3+bio5)	156.08	0.00	0.18	1.00	6	138.79	0.17	0.36(± 0.19)	0.07(± 0.04)
CP10.	ψ(bio18+aspect),p(bio3)	156.99	0.92	0.12	0.63	5	139.71		0.35(± 0.13)	0.06(± 0.03)
CP11.	ψ(bio18+aspect),p(bio3+d2v)	157.71	1.64	0.081	0.44	6	140.43		0.33(± 0.13)	0.06(±0.03)
CP12.	ψ(bio18+aspect),p(bio3+bio19)	158.02	1.93	0.070	0.38	6	140.72		0.31(± 0.11)	0.07(± 0.03)
	Model averaged								0.34(± 0.14)	0.06(± 0.03)
Brown palm civet (77 models tested)										
BPC4.	ψ(aspect),p(bio18+slope+aspect)	146.73	0.00	0.27	1.00	6	129.44	0.17	0.32(± 0.06)	0.07(± 0.04)
Stripe-necked mongoose (58 models tested)										
SNM4.	ψ(bio19+ndvi),p(d2w+bio5+slope)	305.63	0.00	0.17	1.00	7	286.34	0.32	0.48(± 0.10)	0.19(± 0.06)
SNM5.	ψ(bio19+ndvi),p(d2w+alt+bio5)	306.24	0.61	0.12	0.74	7	286.95		0.45(± 0.10)	0.19(± 0.06)
SNM6.	ψ(bio19+ndvi),p(d2w)	306.32	0.69	0.11	0.71	5	291.03		0.39(± 0.12)	0.16(± 0.03)
SNM7.	ψ(bio19+ndvi),p(d2w+bio5)	306.55	0.92	0.10	0.63	6	289.26		0.41(± 0.10)	0.20(± 0.07)
	Model averaged								0.43(± 0.10)	0.18(± 0.05)
Ruddy mongoose (51 models tested)										
RM13.	ψ(bio18),p(ndvi+alt+bio5)	305.04	0.00	0.19	1.00	6	287.75	0.31	0.33(± 0.08)	0.24(± 0.06)
RM14.	ψ(bio18),p(ndvi+bio5)	305.79	0.75	0.14	0.69	5	290.50		0.33(± 0.06)	0.23(± 0.05)
RM15.	ψ(bio18),p(ndvi+d2v+bio5)	305.91	0.87	0.13	0.65	6	288.62		0.33(± 0.07)	0.22(± 0.07)
RM16.	ψ(bio18),p(ndvi+bio5)	305.99	0.96	0.12	0.62	6	288.71		0.34(± 0.07)	0.22(± 0.06)

	Model averaged								0.33(± 0.07)	0.23(± 0.06)
Grey mongoose (81 models tested)										
GM8.	ψ(bio5+bio18),p(bio3+d2w)	211.15	0.00	0.18	1.00	6	193.86	0.21	0.27(± 0.11)	0.20(± 0.06)
GM9.	ψ(bio5+bio18),p(bio19+bio5)	212.63	1.48	0.08	0.47	6	195.34		0.33(± 0.15)	0.14(± 0.05)
	Model averaged								0.30(± 0.13)	0.17(± 0.05)

Information presented for each model includes AIC$_C$ (small-sample size correction to AIC) or QAIC$_C$(quasi-corrected AIC$_C$ with overdispersed data), ΔAIC$_c$(AIC$_c$ – min AIC$_c$), w_i (AIC weight for model i), k (the number of parameters in the model), ψ (probability of carnivore use) and p (detection probability). The (.) convention indicates constant across all sites.

Table 26. Untransformed estimates of beta coefficients and standard error (± SE) for the covariates contained in the top ranked models of small carnivore occupancy during the dry season in Mudumalai Tiger Reserve (2009-2011).

Species/Covariates	Jungle cat	Rusty-spotted cat	Small Indian civet	Common palm civet	Brown palm civet	Stripe-necked mongoose	Ruddy mongoose	Grey mongoose
Intercept	17.08(± 5.35)	-1.59(± 1.27)	-1.56(± 0.78)	-1.74(± 0.61)	-6.42(± 4.42)	4.06(± 2.01)	-1.77(± 1.27)	-2.89(± 1.88)
Dry thorn forest	-	2.37(± 2.23)	2.31(± 1.58)	-	-	-	-	-3.73(± 2.7)
Deciduous	-	-	-	-	-	-	4.53(± 2.92)	5.28(± 3.84)
Riparian	-	-	-	3.03(± 1.93)	-	-	-	-
Steep	-	-	-	-	2.72(± 1.53)	-	-	-
Gentle slope	-	-	1.24(± 1.03)	-	-	-	-	-
Canopy cover	-1.26(± 1.04)	-	-0.97(± 0.69)	-0.60(± 0.53)	-	3.33(± 1.46)	-	-2.51(± 1.23)
Rock cover	26.4(± 12.42)	-	-	-	-	-	-	-
Shrub cover	-0.89(± 0.72)	-	2.31(± 1.06)	-	-	-	-	-
Ground cover	-	-	-	-	-	-	-3.69(± 2.18)	-
Fruiting tree diversity	-	-	-	-	-	-	0.7(± 0.37)	-

Distance to water	-	-	0.54(± 0.39)	-	-18.91(± 14.79)	1.24(± 0.99)	-	-
Distance to village	-	-	-	-0.98(± 0.66)	1.26(± 0.77)	-	-	-
Rodent richness	-	-	0.46(± 0.31)	-	-	-1.29(± 0.71)	-	-0.94(± 0.55)
Termite mound	-	-	-	-	-	-	-	0.41(± 0.39)
Vine	-	-	4.71(± 2.21)	-	-	-	-	-

Table 27. Untransformed estimates of beta coefficients and standard error (± SE) for the covariates contained in the top ranked models of small carnivore occupancy during the wet season in Mudumalai Tiger Reserve (2009-2011).

Species/Covariates	Jungle cat	Rusty-spotted cat	Small Indian civet	Common palm civet	Stripe-necked mongoose	Ruddy mongoose	Grey mongoose
Intercept	-1.51(± 0.71)	-1.69(± 1.12)	-5.39(± 3.17)	-3.56(± 1.61)	-22.59(± 8.46)	-0.95(± 0.40)	-3.13(± 1.43)
Dry thorn forest	1.93(± 1.05)	-	16.62(± 9.11)	-	-	-	-
Undulating	-	-	-	1.91(± 1.34)	-	-	-
Plain	-	-	-	-3.07(± 2.34)	-	0.89(± 0.72)	-
Canopy cover	-1.08(± 0.88)	-	-	-	-	-	-
Shrub cover	-	-0.88(± 0.75)	4.33(± 2.28)	-1.15(± 1.15)	-12.33(± 5.03)	-0.52(± 0.36)	-
Ground cover	-0.97(± 0.46)	-	-	-	4.54(± 2.11)	-0.42(± 0.34)	-
Fruiting tree diversity	-	-	-	-	-	-	0.82(± 0.43)
Distance to water	-	-	15.70(± 8.02)	-	-40.78(± 16.57)	-0.73(± 0.52)	0.58(± 0.36)
Distance to village	-	-	-	-4.18(± 2.67)	-	-0.68(± 0.51)	1.1(± 1.01)
Tatera indica	-	1.87(± 1.56)	-	-	-	-	-
Grewia tiliifolia	-	-	6.95(± 4.28)	-	-	-	-
Termite mound	-	-	-	-	-	0.32(± 0.25)	0.78(± 0.45)

Table 28. Untransformed estimates of beta coefficients and standard error (± SE) for the macroscale covariates contained in the top ranked models of small carnivore occupancy during the dry season in Mudumalai Tiger Reserve (2009-2011).

Species/Covariates	Jungle cat	Rusty-spotted cat	Small Indian civet	Common palm civet	Brown palm civet	Stripe-necked mongoose	Ruddy mongoose	Grey mongoose
Intercept	-1.60(± 0.99)	-3.25(± 1.02)	1.66(± 1.02)	-1.72(± 1.34)	-2.32(± 1.46)	-0.68(± 0.48)	-1.79(± 0.96)	-2.36(± 1.17)
Bio5	-	-	-	-	-	-	-	-1.87(± 1.20)
Bio18	-0.68(± 0.56)	-5.68(± 3.58)	-	-4.02(± 2.36)	-	-	-2.71(± 0.96)	-3.83(± 1.88)
Bio19	-	-	2.23(± 1.09)	-	-	-1.01(± 0.51)	-	-
Slope	-	-3.56(± 2.86)	-	-	-	-	-	-
Aspect	-	-	-	1.91(± 1.35)	3.88(± 1.75)	-	-	-
Distance to village	-1.26(± 0.80)	-	-	-	-	-	-	-
NDVI	-	-	-	-	-	1.13(± 0.72)	-	-

5.5 Discussion

This multi-season habitat selection emphasizes that critical habitat elements from both fine and broad scales are necessary to create suitable habitat for small mammal species. In the current study, both abiotic and biotic variables describing the topography and a measure of vertical vegetation structure provided the basis for modeling the distribution of species. For most species the best model was not strongly supported, inferences can be based on the entire set by using model averaging.

***Jungle cat**:* The positive relationship with rock cover and thorn forest agrees with the existing ecological setting of jungle cat. This species is known to prefer rocky outcrops presumably since it serves as good den sites. During my field survey I would often find scats deposited over rocks or barren ground. Jungle cats seem to utilize open areas even in the wet season which explains its negative relationship with ground cover. Jungle cats are specifically well adapted to occur in scrub forests, dry deciduous as well as agro-ecosystems in India which explains its high probability of use in the thorn forests. On the contrary its negative relationship with canopy cover and shrub cover, explains the species preference towards open habitats which is consistent with existing knowledge on the species (Gupta 2011). The probability of use by jungle cat was high ($\psi = \geq 0.5$) closer to villages (≤ 1 km) indicating its high tolerance level towards disturbed localities or human-modified landscapes. Even the presence of man-made structures in human settlements may provide suitable sites for denning or refuge. High probability of use ($\psi = \geq 0.5$) at sites with low precipitation of the warmest quarter (mean = 216.2 mm) also supports the fact that jungle cat prefers warm and dry habitats.

***Rusty spotted cat**:* The positive association with thorn forest was the only variable with substantial weight for rusty spotted cat occurrence. Its probability of use was high ($\psi = \geq 0.5$) at sites with low precipitation of warmest quarter (mean = 214.5 mm) and plain terrain (mean = 0.64°). The negative association with shrub cover in the wet season suggests its preference towards open areas during moist conditions. *Tatera indica* seems to occupy habitats similar to rusty-spotted cat which could explain selective predation on this species by small cats (as mentioned in chapter 4) and hence a positive relationship. This rodent species has shown to form a reasonable component in the diet of small cats as well (chapter 4, Mukherjee et al., 2004).

Small Indian civet: The positive relationship with ground cover, shrub cover, vine, thorn forest, distance to water and rodent richness agrees with the species biology. Rabinowitz (1991) reported that the core area of small Indian civet in Thailand comprised 57% of dry deciduous dipterocarp forest characterized by more open canopy. In Malaysia, day bed sites of Malay civet *Viverra tangalunga* were situated at ground level on well drained terrain and were associated with some form of cover such as a fallen log, a vine tangle or thick herbaceous vegetation (Colon 2002). *Viverricula spp.* resting sites are normally at or only a little above ground level within the cover of extremely dense shrubbery (Su Su and Sale 2007). *Viverricula spp* occasionally rest in holes at the ground, such as a natural cavity. This species dwells in open and edge forests, and is probably benefited from the rampant forest conversion over the last century in southeast Asia (Duckworth et al., 2008b) which also explains its negative association with canopy cover and positive association with thorn forest. This species is probably selecting sites with heterogeneous mosaic of patches that could in turn support a diverse assemblage of rodents at the particular site. It is known that *Grewia tiliifolia* produces fruits throughout the year indicating that this species probably had to select sites where this tree species was present due to low fruit availability or non-fruiting season of other tree species during the wet season (chapter 4). Civets occur in dry, open country where dense cover grows along water courses, around rocky outcrops and in broken land with gullies which could explain its proximity to water sources and high probability of use ($\psi = \geq 0.5$) at mean precipitation of coldest quarter, 108.82 mm. Carnivore capture rates should increase with prey species richness, as this represents more trophic niches for carnivores (Moreno-Rueda and Pizarro 2010).

Common palm civet: Features like riparian vegetation composition and dimension of streams seem to have a great influence on common palm civet distribution (Nakashima et al., 2010) which supports the positive relationship with riparian vegetation in this study. In fact, future consideration of habitat modeling for common palm civets should incorporate the length of water courses as an important variable. Civets defecate non-randomly at sites such as the banks of small rivers, rain-flow paths, abandoned trails, and treefall gaps (Nakashima et al., 2010). The characteristics of common palm civet sites in this study closely resemble anecdotal descriptions in earlier studies (Nakashima et al., 2010) that defecation sites are often conspicuous ‘‘open canopy sites’’ and at dried river beds. Joshi et al., (1995) found that, in Nepal, the common palm civet defecated on narrow bridges and fallen trees, especially those spanning streams which was also observed during my field survey although I wasn’t sure of the faecal identification upto species level. Yet, the characteristics of defecation sites may vary according to site-specific environmental characteristics. Joshi et al., (1995) noted that radio-tracked common palm civets left their resting trees in the evening until dawn when they retreated to the canopy. Civets travel randomly and do not necessarily follow trails (Veron and Jennings pers comm.). This species ranges in a wide array of habitats including evergreen and deciduous forest (primary and secondary), plantations and near human habitations (Heydon and Bulloh 1996, Duckworth 1997, Su Su 2005) hence it was not surprising to record a negative association with canopy cover and shrub cover indicating that the species uses even open sites and is quite a generalist. This also agrees with its absence in rainforests or evergreen forests of India (Mudappa 2001). During the day, common palm civets most often rest in trees covered with heavy vines; tree cavities and occasionally in tall trees even without vines (Joshi et al., 1995), vine coverage was the primary criterion for choosing resting sites in Chitawan National Park and yet vine did not influence the distribution of this species in Mudumalai. Aspect (mean =143.55°) was positively associated with common palm civet use ($\psi = \geq 0.5$) implying its affinity towards south-eastern region of the reserve while the case was inverse with precipitation of the warmest quarter (mean =215.48 mm) suggesting its affinity towards warm sites.

Brown palm civet*:* Greater success rate in contiguous rainforests of south India reported by Mudappa et al., (2007) based on track plot visitation rates by brown palm civet. This species is known to suffer from fragmentation probably because of its inability to survive in a matrix of tea/coffee plantations devoid of continuous tree cover and sufficient diversity of fruit resources (Mudappa 2001 Mudappa et al., 2007) which explains its decrease in probability of use at sites close to settlements or tea/coffee estates in Mudumalai. The species preference towards steep areas suggests its affinity towards the semi-evergreen forests which is characteristic of this feature coupled with high leaf litter, tall trees with large girth size and tangled branches.

Fruits eaten by this species were mostly small (<1 cm diameter), multi-seeded, pulpy berries, and drupes with moderate to high water content (Mudappa et al., 2010) could describe its low use probability of sites close to water sources (Table 26). High probability of use at a mean aspect of 311.77° suggests the species affinity towards the north-western part of the reserve. There are a whole range of types of distributions, continuous, random, transitional or patchy. However not all the habitat types used by the species could be of high quality especially in the case of carnivores. Animals could move from low quality to patches of high quality habitat to fulfill their biological needs.

***Stripe-necked mongoose**:* This species has been recorded in deciduous and evergreen forest, swampy clearings, plantations, open scrub and along watercourses (Webb-Peploe 1947, Van Rompaey and Jayakumar 2003). In deciduous forests it is usually found in swampy clearings, along watercourses, and in open scrub (Krishnan 1972) suggesting its occupancy across a wide range of habitats and hence low probability of use near water sources. Shrubs are important for animals because Yellow mongooses (*Cynictis penicillata*) in the southern Kalahari favored burrow locations under shrub cover (Blaum et al., 2007) hence overall shrub and ground cover seems to be an essential habitat component for mongooses as it provides shelter and burrow sites. Its affinity towards high canopy cover and NDVI suggests this species needs shaded areas for resting or foraging. High probability of use ($\psi = \geq 0.5$) at a mean precipitation of coldest quarter, 101.06 mm explains its affinity towards moderate moisture.

Ruddy mongoose: Fruiting tree diversity was directly proportional to probability of site use by ruddy mongoose. Fruit rich sites could act as surrogates for high arthropod diversity in tropical biomes that are primary prey for mongooses. Host fruit characteristics are responsible in part, for variable direct effects on host plant fitness by insect frugivores. Variable direct effects between insect frugivores and host plant fitness combined with rapid decay of infested fruits is likely to have implications for seed dispersal and see predation by the third trophic level where in this case could be vertebrate frugivores (lesser carnivores and rodents). Invertebrate frugivores mostly comprise Orders; Lepidoptera, Hemiptera, Coleoptera and Diptera that are predated by mongooses. Fruit rich patches could also attract seed predation rates. Deciduous forest in Mudumalai is well dominated with teak trees and well wooded vegetation with high leaf litter. Increased activity of termites and other soil and litter-dwelling invertebrates in this forest would serve as important forage for ruddy mongoose which explains its high probability of use with sites having termite mounds. Teak forests are dominated by fungus-growing species (Macrotermitinae). Invertebrates show a strong seasonal activity pattern, with a distinct low during the long dry season which supports the positive relationship between ruddy mongoose and termite mounds in the wet season. Termites provide essential nutrients as forage and termite mounds provide suitable burrowing sites for mongooses. Although there is not much literature from India on behavioural aspects of this species, a similar size species from Africa, common dwarf mongoose (*Helogale parvula*) uses termite mounds as burrows (Creel and Hoffmann 2008). The banded mongoose (*Mungos mungo*) from central and eastern parts of Africa lives in savannas, open forests and grassland, especially near water, but also in dry, thorny bushland and often use termite mounds as various types of dens for shelter (Hiscocks and Perrin 1991). Ruddy mongoose showed maximum affinity towards low ground cover in both seasons and low shrub cover in the wet season suggesting its preference towards open areas. This species is a generalist, since it has been recorded in forested areas including dry forests, dry thorn areas, and disturbed forests. In India, this species is found exclusively in dry forests. Gupta (2011) recorded a positive correlation between ruddy mongoose and distribution of villages in Sariska Tiger Reserve which supports the high probability of use by the species near villages in the present study. It also showed affinity towards closest water sources in the wet season as also reported by Gupta (2011). In the wet season, canopy cover was positively associated with ruddy mongoose probably because high canopy cover provides sufficient shade. In this model, precipitation of warmest quarter was negatively associated with ruddy mongoose suggesting that the species preferred sites with low precipitation (mean = 214.25 mm).

***Grey mongoose**:* Grey mongoose was negatively associated with thorn forest while it was positively associated with deciduous forest. Although the species can thrive in both forests (Kumara and Singh 2007) it might have to spatially compete with jungle cat and hence the negative relation with thorn forest. Its positive relation with termite mound also suggests a similar explanation as stated for ruddy mongoose. Negativity towards canopy cover indicates the species preference towards open areas. It preferred areas with low rodent richness in the dry season. It was positively associated with fruiting tree richness in the wet season probably due to less fruit availability wherein the animal had to move towards fruit rich patches. Its positive affinity close to water sources and villages in this study has also been reported in past studies on the species (Kumar and Umapathy 1999, Gupta 2011). Its negative affinity towards maximum temperature of the warmest month and precipitation of the lowest quarter indicates its high probability of use ($\psi = \geq 0.5$) towards low temperature (mean = 30.66°C) and precipitation (mean = 214.03 mm). Bioclimatic preferences exhibited by the species reflect the forest type (dry deciduous) it thrives onto.

The species seasonal habitat use at both spatial scales highlighted the importance of the forest matrix in the region and supports the park as a priority conservation unit. The study most importantly proves that microscale habitat features predicted occupancy estimates higher than macroscale variables for small cats and mongoose species. There was no major difference in estimates for brown palm civet while macroscale estimates were higher for small Indian civet and common palm civet. This finding indicates that certain species could be sensitive to local-scale measures while for others remote scale measures are important.

Brown palm civet is an important community member of secondary growth forests because of their role in seed dispersal and reforestation process. The eastern part of the reserve (Masinagudi range) is approximately 60 km^2 of dry thorn forest, where majority of the land is heavily grazed by livestock with many villages, private lodges and small scale agriculture and farming and cattle operations. Hunting and illegal poaching does not occur much in the reserve, however from as far as I am aware from speaking with local people, brown palm civet and common palm civet used to be hunted by tribal communities for meat. Rare species inherently occur at low densities with patchy distributions. Derived parameters from occupancy analysis are robust (MacKenzie et al., 2006), but given the relatively small sample size in my study, only limited inferences should be drawn on their ecology. However, given more sampling effort at more sites, these parameters may be very valuable for estimating their true habitat use. The occupancy modeling approach applied in this study demonstrates the usefulness of this tool to determine the habitat use when all species exhibit different detection probabilities. More cameras over a wider area provide greater detections and increase robustness of the models. This shows that occupancy modeling is a superior approach to traditional habitat use techniques for small carnivores. One of the benefits of the occupancy modeling approach is that this research can continue as a multi-year mammal community monitoring program and help understand habitat changes associated with extinction and colonization of forest patches. It is important to understand these factors now because there are several additional compounding factors, such as climate change, deforestation, and an increasing human population that will affect biologically sensitive communities in the future. Small mammal species are suited for the exploitation of heterogeneous and patchy environments and are able to deal with changing habitat conditions. Thus, it seems that a considerable number of mainly generalist species favour similar microhabitats, which can be regarded as particularly suitable for locomotion, foraging or avoiding predators, thus indicating an increased probability of using similar or even the same food resources. Future studies should emphasize on multi-trophic interactions and indirect effects on small carnivore communities would portray a very complex ecological system. Diet of *Rattus rattus* and *Mus musculus* include seed and pulp. In this context I may expect *a priori* that fruit rich patches could attract fruit eating rodents thereby enhancing predation opportunity by small carnivores. The diversity of a given resource allows a larger number of species to co-exist in a given area and consequently the structure of the habitat is an important factor influencing community structure. A world of research possibilities exist ahead and a complete understanding of the remarkable details of small carnivore community and trophic interactions await.

Chapter 6

HABITAT SUITABILITY MODELING OF SMALL FELIDS, VIVERRIDS AND HERPESTIDS

6.1 Introduction

An understanding of the relationship between spatial distribution of animals and their habitat parameters plays an important role in conservation and management of lesser known species. Remote sensing and geographical information system (GIS) can be used as a tool for deriving information on the habitat preference of wildlife species. Research has become increasingly focused on the extrapolation of species distribution from incomplete data or a really small data set to obtain reliable distribution maps efficiently. By processing environmental information and presence/absence data, several statistical methods can provide estimates on the probability of occurrence of a given species (Guisan and Zimmermann 2000). Poorly known species are usually represented by low numbers. The ecological niche can be defined as the set of environmental conditions (abiotic factors) under which a species is able to maintain viable populations without immigration (Grinnell 1924). The quality of model predictions can be affected by the quality of the source data. Models can be weakened either by propagating errors inherent in the data-gathering processes or by not including all relevant ecological data. Niche-based modelling of potential distributions has been used recently to examine various ecological and evolutionary aspects, such as competition between phylogenetically related species (Mukherjee et al., 2010) or variation in species niche requirements through evolutionary time (Peterson and Holt 2003). At geographic scales habitat models provide a way to develop hypotheses about features that affect distribution (Manel et al., 2000). Distribution models should consider appropriate scale (spatial and temporal), predictive ability, and include an assessment of uncertainty. The challenge of developing habitat models at large scales, and using them to monitor habitat suitability, has been undertaken using various approaches. Identifying proper variables, but at an incorrect scale, may lead to weak or incorrect apparent relationships.

Predictive habitat modeling and mapping, based on species habitat relationships forms an analytical foundation for informed conservation planning, mapping patterns of biodiversity, detecting distributional changes from monitoring data and quantifying how variation in species performance is related to one or several controlling factors (Phillips et al., 2006). Predictive species mapping lays its foundation in the ecological niche-theory and predictor analysis from the spatial distribution of variables that correlate with or control the species occurrence. Available and published distribution maps on small carnivores in India in books and reports were created traditionally by compiling locality records in the species distribution range.

Predictive models provide an alternative way to build distribution or habitat suitability maps of species from field based intensive surveys and their correlations with the variables of interest or importance to the species. It is imperative to investigate how small carnivores respond or relate to variables within a Protected Area setting that hold acres of optimal habitat. Predictive species distribution models exist for selected small carnivore species around the world for eg:- jungle cat *Felis chaus* , leopard cat (*Prionailurus bengalensis*) (Mukherjee et al., 2010), fisher *Martes pennanti* (Zielinski et al., 2006), flat headed cat (*Prionailurus planiceps*) (Wilting et al., 2010), Andean cat (*Leopardus jacobita*) (Marino et al., 2011), Colombian Weasel (*Mustela felipei*) (Burneo et al., 2009), and European wild cat (*Felis silvestris*) (Monterroso et al., 2009). Jennings and Veron (2011) predicted the distribution of large-spotted civet (*Viverra megaspila*), Malay civet (*Viverra tangalunga*), large Indian civet (*Viverra zibetha*), small Indian civet (*Viverricula indica*), short-tailed mongoose (*Herpestes brachyurus*), Javan mongoose (*Herpestes javanicus*), collared mongoose (*Herpestes semitorquatus*), and crab-eating mongoose (*Herpestes urva*) within Southeast Asia using ecological niche modeling, determined their habitat/elevation niche preferences, examined interspecific differences in niche parameters, and investigated possible factors that affected their distribution and niche patterns.

Most determinant variables are environmental, bioclimatic, topographic and/or disturbance indices which are essential in documenting distribution across a wider landscape. Animals that are rare and secretive adapt to the environment for a variety of reasons some of which that are very hard to justify or interpret. The small carnivores are known to be either microhabitat specialists or generalists. Ecological requirements of small carnivores provide substantial evidence that their distribution is determined by resources at the home-range scale. It is essential for managers to be able to distinguish between habitat suitability and actual population size. For most species, biological information is available to guide decisions regarding appropriate variables for inclusion in candidate models, which greatly facilitates development of a suite of candidate models. Ecological niche models, the Genetic Algorithm for Rule-set Prediction (GARP) and maximum entropy (MaxEnt) have been used to assess potential ranges and distributional connectivity of Asian and African viverrids (Papes and Gaubert 2007, Jennings and Veron 2011).

A predictive distribution model, or habitat suitability model, usually consists of a probability map depicting the likelihood of occurrence of a species.The categorization of habitat quality displayed in the spatial model can be used to prioritize areas requiring protection based on their value. This statement is made on the premise that the probability of species presence is positively correlated with the quality of the habitat. By providing baseline information about the spatial arrangement of potentially suitable habitat for a species, habitat suitability maps can be used to facilitate protection and restoration of critical habitat, and hence they have broad applicability in conservation biology and wildlife management (Manel et al., 2000). Several developments in ecological niche modelling (ENM) have provided new tools to estimate species ranges and identify suitable habitats (Gaubert et al., 2005a, Papes and Gaubert 2007, Thorn et al., 2009). The maximum entropy framework appears to be robust even if only few occurrence records are available (Pearson et al., 2007). The maximum entropy-based machine-learning method is used for making predictions when incomplete data are available. “MaxEnt estimates the probability distribution for a species occurrence that is most spread out given the constraints derived from the available data” (Phillips et al., 2006). These constraints are deduced from environmental conditions at species presence records and require that the expected value of an environmental variable or its function must be within a confidence interval derived from its empirical mean (Phillips et al., 2006). Maximum entropy modeling is an active area of research in statistics and machine learning, and progress in the field as a whole can be readily applied in this study.

The North-east and Western Ghats are rich biogeographic units and biodiversity "Hotspots" in the world (Myers et al., 2000). Status reports, synoptic work, behavioural observations on the small carnivores in the Western Ghats have provided considerably useful knowledge on their habitat use (Prater 1971, Mudappa 2001). The diverse mammalian community in the north-eastern region is driven by contiguous rainforest habitats with South-east Asian forests along with wide altitudinal and habitat variation (Datta et al., 2008). Although small carnivore diversity is higher in the North-east than in the Western Ghats, the small carnivore assemblages in both hotspots are unique. The North-western and central India comprises semi-arid and dry tropical forests holding promising populations of sympatric species like jungle cat (*Felis chaus*), desert cat (*Felis silvestris ornate*), rusty spotted cat (*Prionailurus rubiginosus*), small Indian civet (*Viverricula indica*), common palm civet (*Paradoxurus hermaphrodites*), grey mongoose (*Herpestes edwardsii*) and ruddy mongoose (*Herpestes smithii*) (Gareja and Dharaiya 2011).

The study was aimed 1) to generate a habitat suitability model for predicting small carnivore distribution in Mudumalai Tiger Reserve using environmental variables and 2) to identify environmental variables associated with small carnivore occupancy.

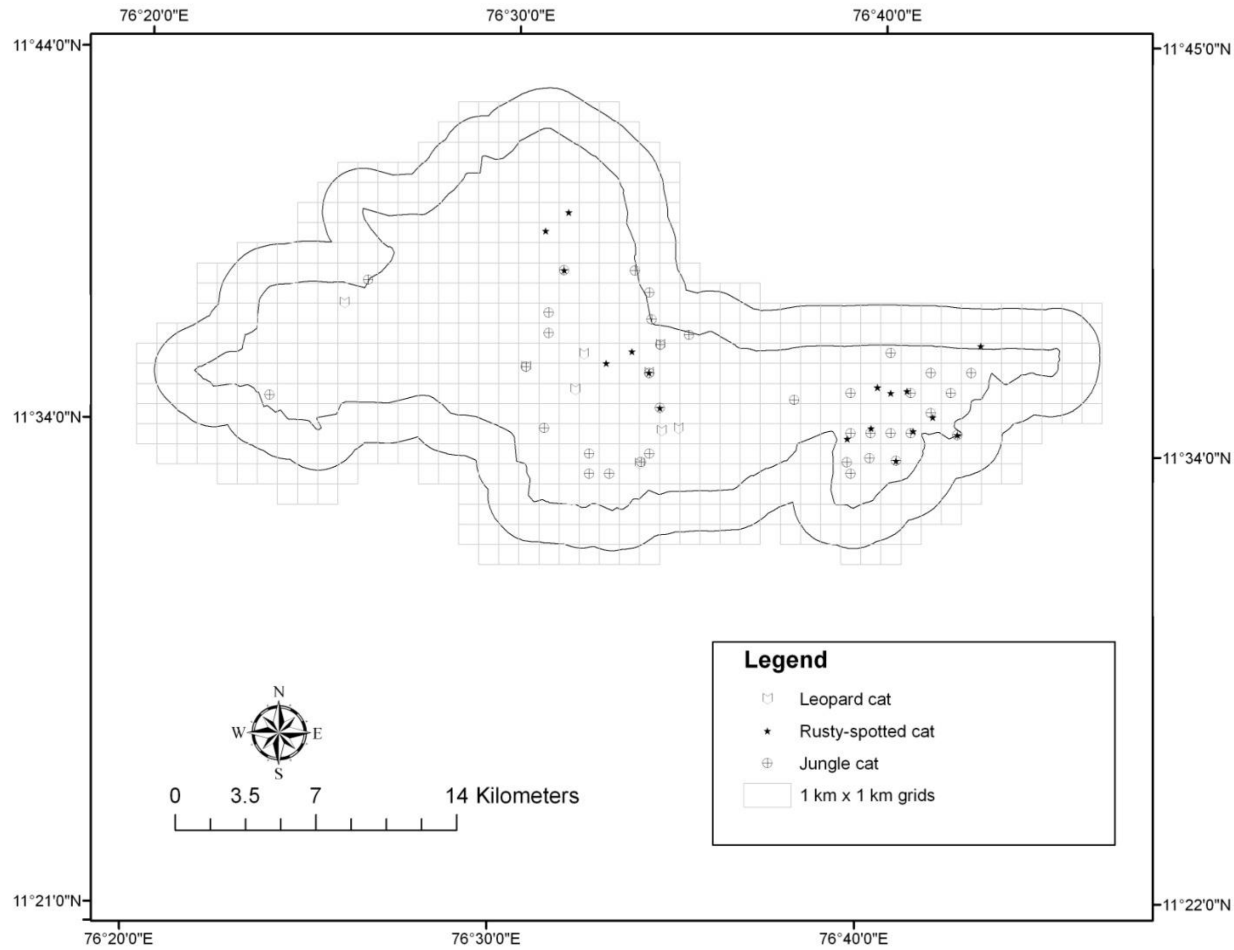
6.2 Materials and methods

The fundamental limitation of presence-only data is that sample selection bias (whereby some areas in the landscape are sampled more intensively than others) has a much stronger effect on presence-only models than on presence-absence models (Phillips et al., 2009). For any modelling method – few samples provide limited information for determining relationships between the species and its environment (Pearson et al., 2007). The MaxEnt approach performs better with presence-only data than other methods such as bioclimatic envelope modeling or genetic algorithms (Elith et al., 2006).

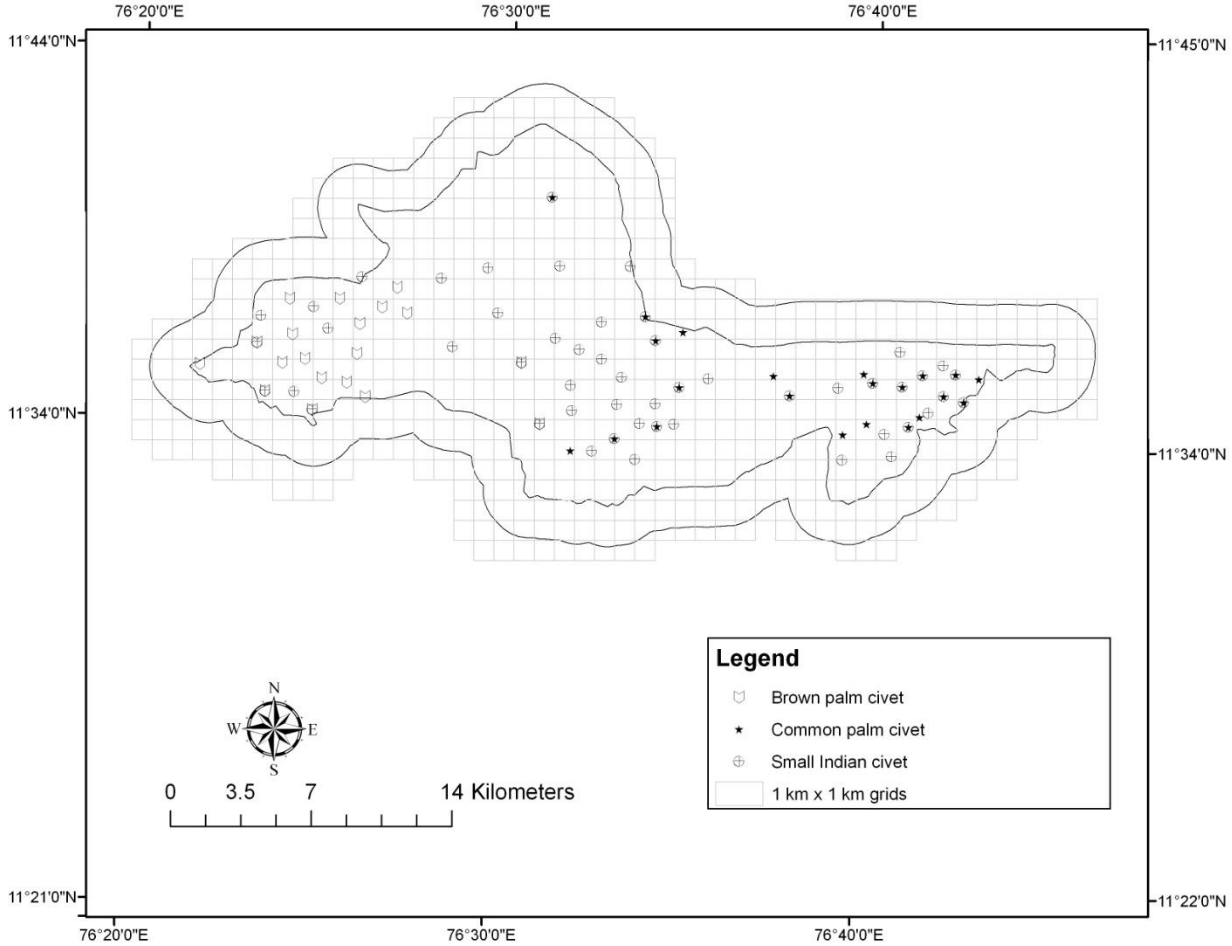
6.2.1. Small carnivore species distribution data

Species distribution data was collected from camera trapping (methodology has been explained in chapter 3), direct sighting records, interviews with forest staff, local people (from villages and settlements, resort owners), and road kill incidents. All the records were pooled from 2009-2011 for every species separately. The small carnivores analysed were jungle cat (36 point localities), leopard cat (9), rusty-spotted cat (18), small Indian civet (51), common palm civet (22), brown palm civet (19), stripe-necked mongoose (55), ruddy mongoose (51) and grey mongoose (35). I created comma-separated value (.csv) file containing longitude and latitude coordinates of each species occurrence from an Excel spreadsheet. These were plotted in ArcGIS 9.3 and overlayed on 1 km^2 grid cells. Based on which the distribution maps were generated (Fig 36a, b, c).

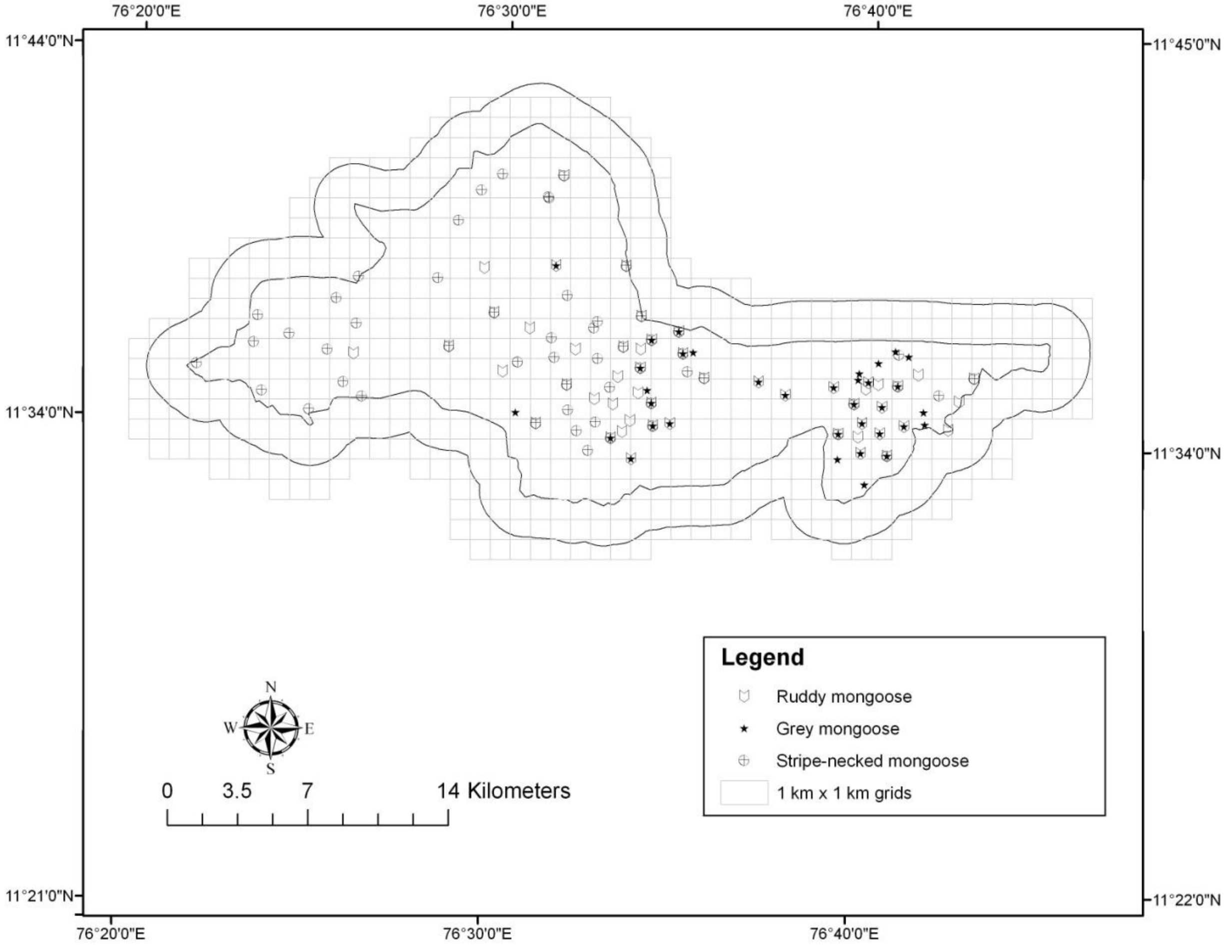
Fig 36a. Spatially unique localities of small cats in Mudumalai Tiger Reserve (2009-2011).



b. Spatially unique localities of civets in Mudumalai Tiger Reserve (2009-2011).



c. Spatially unique localities of mongooses in Mudumalai Tiger Reserve (2009-2011).



6.2.2 Extraction of explanatory environmental predictors

I used the Geographic Information System (GIS) in ArcGIS 9.3 (Environmental Systems Research Institute [ESRI], Inc., Redlands, CA, USA) for data extraction.

For species habitat suitability modeling the resolution, or pixel size of the raster data should relate to the home range of the species under investigation, yet this assumption could be violated for those that exceed more than grid size (eg: small cats and civets), however a fishnet of 1 km^2 was overlayed on the boundary and buffer layer of Mudumalai.

Climatological variables, topographical variables, biological variables and anthropogenic variables were extracted using Zonal Statistic tool in the Spatial Analyst toolbox such that every 1 km^2 pixel is assigned the variable value.

A classified forest cover map from Forest Survey of India of categories; 1 = water bodies, 2 = non-forest, 3 = scrub, 4 = open forest, 5 = dense forest and 6 = very dense forest were obtained. The land use land cover map at 1:250000 scale was derived from DIVAGIS (version 7.1.7.2, http://www.diva-gis.org) where original data was resampled onto a 30 seconds grid (source-GLC2000). These were classified as categorical variables: 1 = tropical evergreen, 2 = subtropical evergreen, 8 = moist deciduous, 9 = dry deciduous, 16 = degraded forest.

I used the Shuttle Radar Topography Mission (SRTM) digital elevation data, aggregated from the native 90-m resolution to 1 km. In addition to mean elevation, the standard deviation based on the 90-m data was included as an indicator of surface ruggedness. Slope and aspect was calculated from elevation layer using Surface analysis tool from spatial analyst toolbox in Arcmap. The Slope layer had values measured in degrees (0 – 90) for each 1 km^2 pixel. The aspect degree was converted such that each pixel has a value assigned from -1 to 360 degrees.

Normalized difference vegetation index (NDVI) is a method of measuring and mapping the density of green vegetation. For its measurement scientists use satellite sensors that observe the distinct wavelengths of visible and near infrared sunlight which is absorbed and reflected by the plants, then the ratio of visible and near-infrared light reflected back up to the sensor is calculated. The ratio gives a number from minus one (−1) to plus one (+1). An NDVI value of zero means no green vegetation and close to +1 (0.8–0.9) indicates the highest possible density of green leaves. The 'normalized difference vegetation index' is calculated by the formula: NDVI = (IR−R)/(IR + R), where IR = infrared light and R = red light.

I obtained the 19 'bioclimatic' variables based on the global climate data sets developed by Hijmans et al., (2005). These GIS data sets characterize global climates using average monthly weather station data and are available at 30 arc-seconds resolution (approximately 1 km^2). I used the mean human influence index (HII) at 1 km^2, a measure of human influence on global surface, combining data of population density, land transformation, human access and presence of infrastructures. This information is a composite measure of multiple human activities that include urban extent, population density, roads, navigable rivers and agricultural land; (Last of the WILD DATA v. 2 2005). The HII values range from 0 (no human influence) to 64 (maximum human influence possible under the method). Actual evapo-transpiration (AET) is the effective quantity of water that is removed from the soil due to evaporation and transpiration processes. AET is dependent on the available atmospheric energy, vegetation characteristics, quantity of water available in the soil and soil hydrological properties. I used global AET layers (the average 1950-2000 period) evapotranspiration (mm) at 30 arc-seconds (920 meters at equator). Surface water bodies (rivers and streams) were extracted for the country wide data from DIVAGIS (version 7.1.7.2, http://www.diva-gis.org) along with field data locations of potential water sources. I used the Euclidean distance tool to create a raster “distance to” (km) layer for the closest water source (d2w). This tool calculated a straight line distance to the nearest source variable of interest within the raster layer such that each pixel is assigned a value of distance to water and village.

Since most climatic and NDVI variables were highly correlated (R^2 = >0.5), to avoid the multicollinearity issue, I did not include other climatic and NDVI variables that were strongly correlated. I used only independent variables; bio3 = isothermality (mean diurnal temperature range/[maximum temperature of warmest month/minimum temperature of coldest month]), bio5 = max temperature of the warmest month, bio18 = precipitation of the warmest quarter, bio19 = precipitation of the coldest quarter, NDVI-March, NDVI-June and NDVI-July. Multicollinearity was also checked for all combinations of environmental variables.

6.2.3 Modelling approach

MaxEnt

MaxEnt is a machine learning algorithm that estimates the most uniform distribution (maximum entropy) across the study area given the constraint that the expected value of each environmental predictor variable under this estimated distribution matches its empirical average (Phillips et al., 2006). I used MaxEnt because it performs better than other presence-only modeling techniques (Elith et al., 2006), especially with low numbers of occurrence locations (Papes and Gaubert 2007). The reliability of the results of MaxEnt has been confirmed by its good capacity to predict novel presence localities for poorly known species (Pearson et al., 2007). This method has been used to develop habitat suitability models for a range of mammals (Monterroso et al., 2009, Wilting et al., 2010, Jennings and Veron 2011). The modelled probability is a ‘Gibbs’ distribution (i.e. exponential in a weighted sum of the features) and the model logistic outputs have a natural probabilistic interpretation representing degrees of habitat suitability (0 = unsuitable to 0.99 = best habitat) (Pearson et al., 2007). Consequently, maximum entropy modeling consistently outperforms other methods of modeling spatial distribution (Elith et al., 2006, Phillips et al., 2006), except possibly those utilizing multiple repeated visitations to the same sampling points that correct for imperfect detection of individuals (i.e., occupancy modeling; MacKenzie et al., 2006). Like most maximum-likelihood estimation approaches, the MaxEnt algorithm *a priori* assumes a uniform distribution and performs a number of iterations in which the weights associated with the environmental variables, or functions thereof, are adjusted to maximize the average probability of the point localities (also known as the average sample likelihood), expressed as the training gain (Phillips 2006). These weights are then used to compute the MaxEnt distribution over the entire geographic space. Consequently, this distribution expresses the suitability of each grid cell as a function of the environmental variables for that grid cell. A high value of the function (in units of cumulative probability) for a particular grid cell indicates that the grid cell is predicted to have suitable conditions for the species in question (Phillips 2006). MaxEnt has several characteristics that make it highly suitable for the task of modelling species ranges (Phillips et al., 2006). These include a deterministic framework; the ability to run with presence-only point occurrences; a high performance with few point localities (Hernandez et al., 2006); better computing efficiency, enabling the use of large-scale high-resolution data layers; continuous output from least to most suitable conditions; and the ability to model complex responses through a number of distinct feature classes (e.g. functions of environmental variables).

Data analyses and validation

A set of ascii environmental layers and a .csv file of known locations of a species were used to produce probability maps that predict the potential distribution of a species.

The measure of fit implemented by MaxEnt is the area under the curve (AUC) of a receiver operating characteristic (ROC) plot (ranging from 0.5 = random to 1 = perfect discrimination). The selection and contribution of each variable depends on the other variables in the model, so highly correlated variables (bioclimatic layers and monthly NDVI) may greatly influence the final model. Only one of a pair of correlated (r > 0.50) variables was used in the model; the variable with the least biological importance was eliminated.

As a result, the final reduced data set used in this study converged to a total of 17 environmental layers which were projected to the UTM zone to match their coordinates, clipped to the extent of the boundary along with 2 km buffer, resampled to the cell size of 30 arc-seconds (~ 1 km^2), and entered with the occurrence data into MaxEnt version 3.3.3 (http://www. cs.princeton.edu/~schapire/MaxEnt). I selected the jackknife option in the program through which the importance of individual environmental data layers can be estimated. It also provided response curves showing how the prediction depends on a particular environmental variable (Phillips 2006). For all model runs in this study, I used the MaxEnt default settings for regularization and in selecting the feature classes (functions of environmental variables). These include linear, quadratic, product, threshold and hinge features, depending on the number of point localities (Phillips 2006). Description for each of the feature type models can be found in Phillips et al. (2006) and Phillips and Dudik (2004). The Auto feature type allows the set of features used to depend on the number of presence records for the species being modeled using general empirically-derived rules. It should be noted that the model algorithm (MaxEnt) used in this study is largely robust to covariance among variables, and that data reduction was performed mainly to improve interpretation. I set the program to run 1000 iterations with a convergence threshold of 0.00001, a regularization multiplier of 1, a maximum of 10000 background points, the output grid format as “ logistic, ” algorithm parameters set to “ auto features, ” and all other parameters at their default settings (Phillips and Dudik 2008). I had the program randomly withhold 25% of the presence locations to test the performance of each model. The Linear Quadratic Hinge feature type was the best fit model for predicted distribution of jungle cat, common palm civet, brown palm civet, stripe-necked mongoose, ruddy mongoose and grey mongoose. Linear Quadratic feature type model was the best fit for rusty-spotted cat while the Linear feature type model fitted well for leopard cat and small Indian civet distribution,

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6.2.4. Variable contribution and response curves

I considered MaxEnt’s heuristic estimates of the relative contribution of environmental variables to the models and the results of jackknife analyses for each environmental layer (Phillips and Dudik 2008). For the variables with highest predictive value, I examined the response curves showing how each of these environmental variables affects the MaxEnt prediction (Phillips and Dudik 2008). The curves illustrate how the logistic prediction changes as each environmental variable is varied, while keeping all other environmental variables at their average sample value. The curves thus represent the marginal effect of changing exactly one variable. Each of the models was then re-run a second time, after selecting only those variables that contributed at least 2% to the initial model result. This methodology reduced the total numbers of variables used in the analysis.

6.3. Results

6.3.1 Jungle cat habitat modeling

Distribution models for jungle cat performed well based on the high (0.91) AUC value (Fig 37). Annual precipitation had the highest predictive power (46%) to the model output (Table 29). The jackknife test of variable importance showed the highest gain when the variable annual precipitation of the warmest quarter was used in isolation which therefore appears to have the most useful information by itself (Fig 37). This variable also decreased the gain when it was omitted showing that it had the most information that was not present in other variables (Fig 38). The variables annual precipitation of the warmest quarter, elevation, NDVI (March), forest type, aspect, NDVI (July) and distance to water together contributed 96% to jungle cat model. NDVI (March) was negatively related to predicted jungle cat presence where high probabilities were predicted at low NDVI areas (Fig 39a). Aspect and distance to water response curves showed bi-modal distribution (Fig 39b, c). The response curve for elevation showed that highest predicted suitability areas in areas of low to medium elevation (200 – 900 m). The land cover categories with the highest predicted probability of jungle cat as calculated in included deciduous and degraded areas in the south-eastern part of the reserve. The annual precipitation of the warmest quarter showed a skewed response curve with suitable conditions towards 210 mm of precipitation (Fig 39g). Predicted jungle cat presence showed a sigmoid curve starting from low values that accelerated and approached high NDVI (July) values (Fig 39h). The response curve showed jungle cat predicted probabilities decreasing abruptly from low to high topography wetness index. The MaxEnt model generated a map (predicted probability of occurrence; Fig 40) of potential jungle cat distribution in Mudumalai showing that high probability areas (considered to be areas with > 0.6 probability of presence) accounted for a small portion (38 km^2) of the reserve with the Linear Quadratic Hinge feature type models indicating the suitable habitat.

Fig 37. ROC curve of Sensitivity versus Specificity for the habitat model of jungle cat.

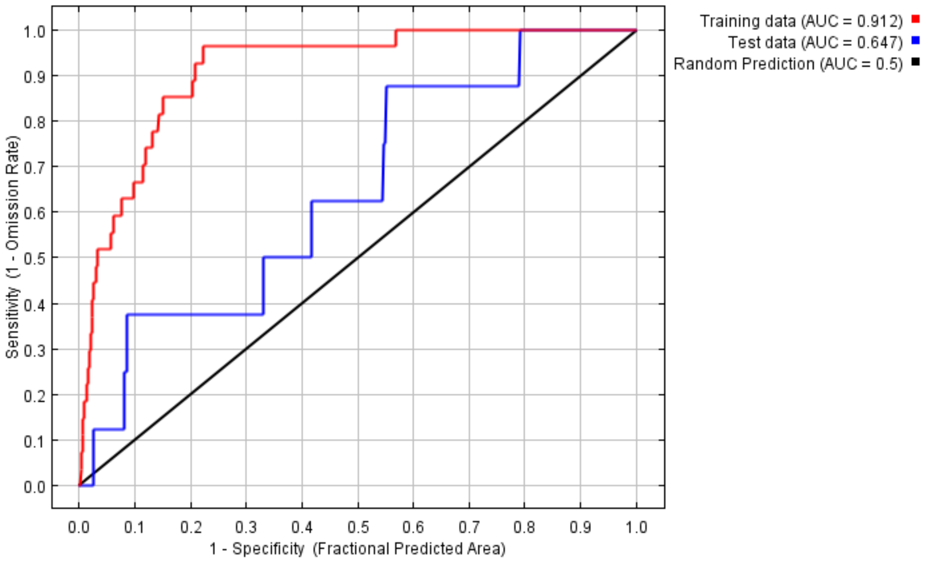


Fig 38. Jackknife analyses of individual predictor variables important in the development of the full model for jungle cat in relation to the overall model quality or the “regularized training gain.” Dark blue bars indicate the gain achieved when including only that variable and excluding the remaining variables; light blue bars show how much the gain is diminished without the given predictor variable.

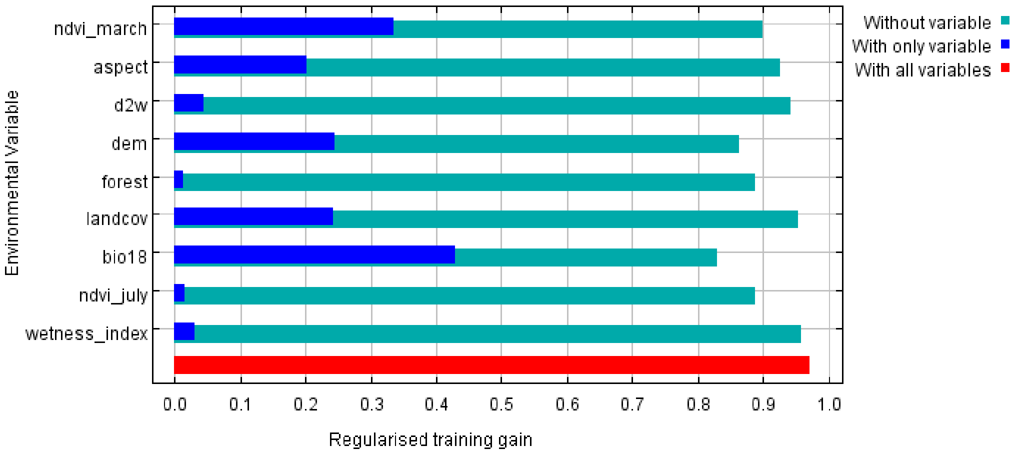


Fig 39. Graphical representation of the relationship between (a) ndvi_march, (b) aspect, (c) distance to water source, (d) elevation, e) forest type, f) landcover type, g) annual precipitation of the warmest quarter, h) ndvi_july i) topography wetness index, and jungle cat probability of presence (2009-2011). Each of the curves represents a different MaxEnt model created using only the corresponding variable.

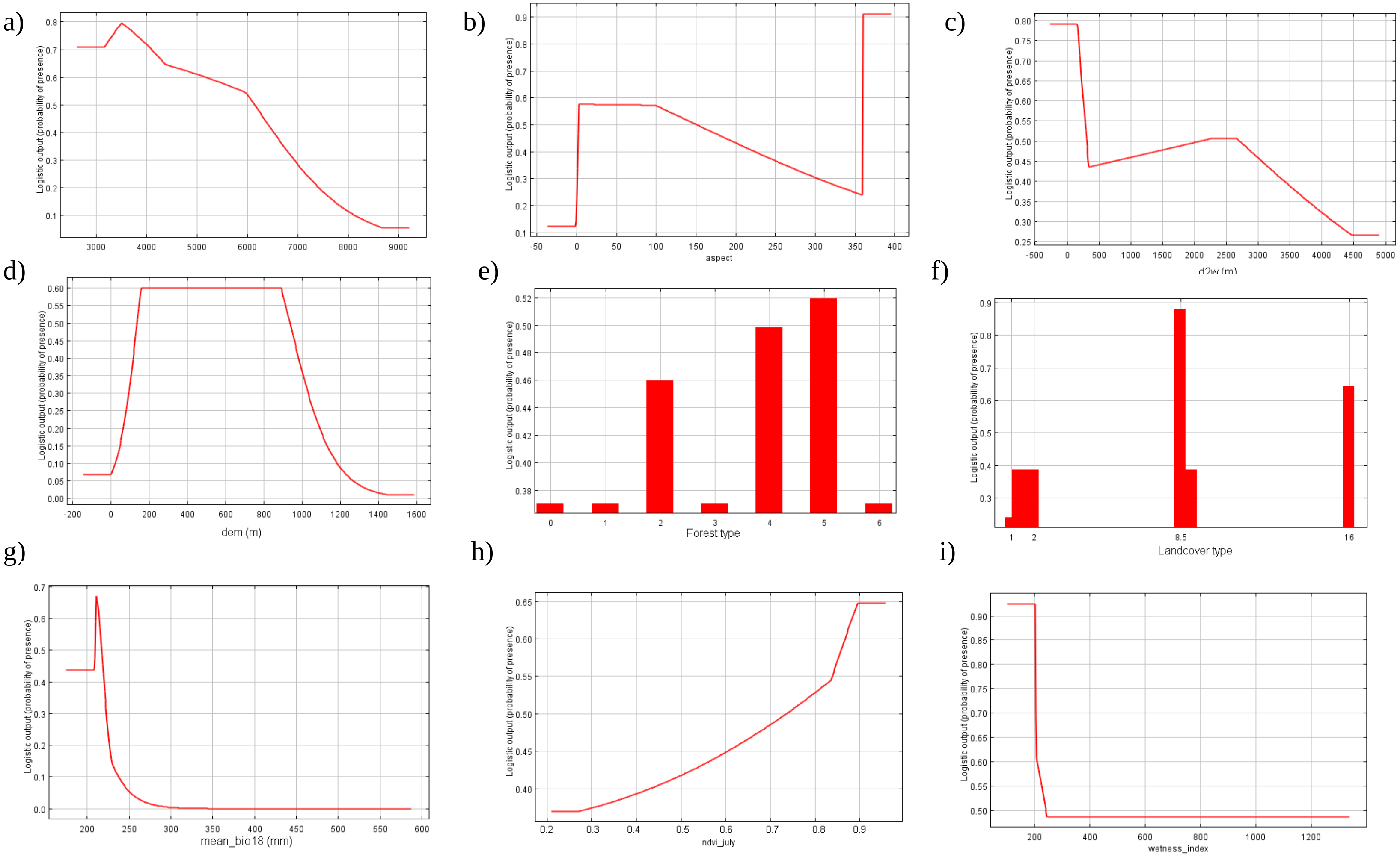
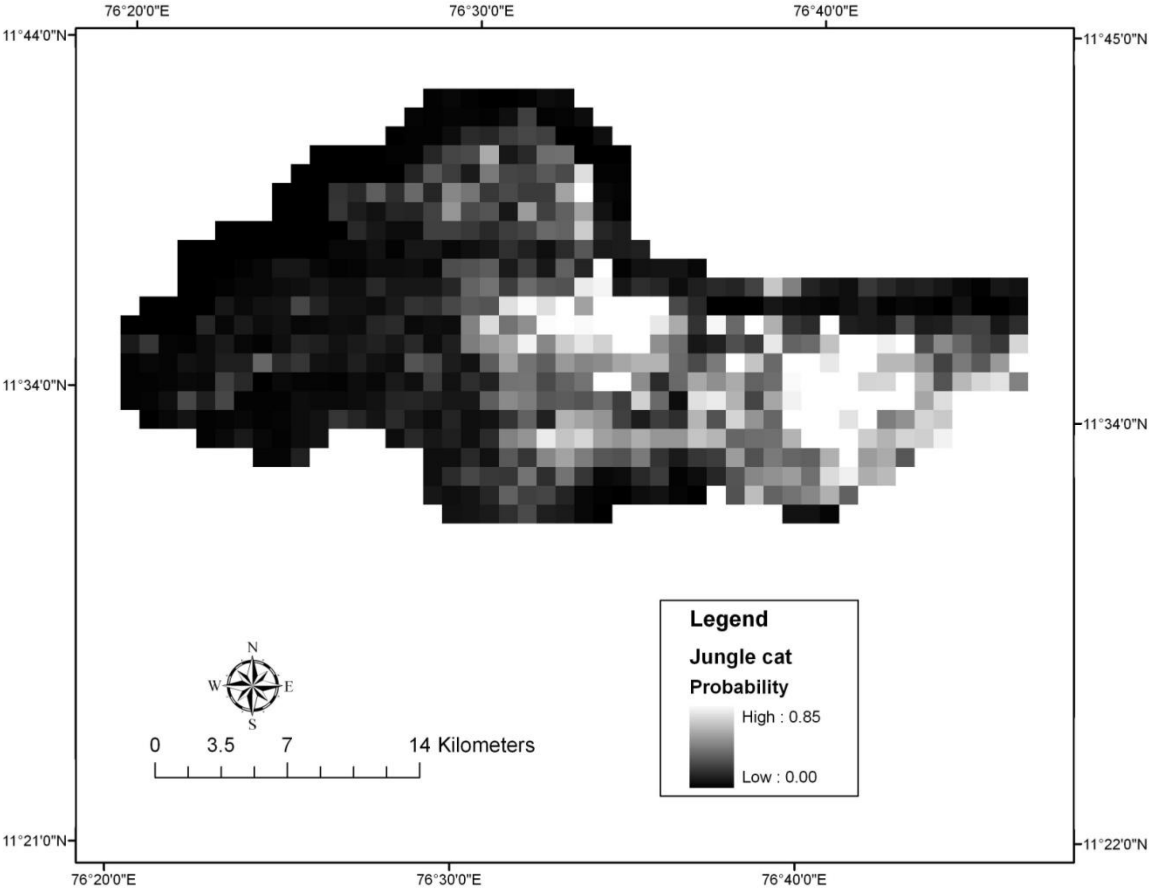


Fig 40. Predicted distribution for jungle cat in Mudumalai Tiger Reserve estimated by MaxEnt modeling (2009-2011). Potential areas are shown in grey shading with the white color indicating higher probabilities of occurrence.



6.3.2 Rusty-spotted cat habitat modeling

Distribution models for rusty-spotted cat performed well based on the high (0.88) AUC value (Fig 41). Land cover type had the highest predictive power (30.4%) to the model output (Table 29). The jackknife test of variable importance showed the highest gain when the variable annual precipitation at the warmest quarter type was used in isolation which therefore appears to have the most useful information by itself (Fig 42). This variable elevation decreased the gain when it was omitted showing that it had the most information that was not present in other variables. The variables landcover type, aspect, elevation, annual precipitation at the warmest quarter and wetness index contributed 95.6% to rusty-spotted cat model. Aspect, wetness index and annual precipitation of the warmest quarter were negatively related to predicted rusty-spotted cat presence (Fig 43a, e, f). The annual precipitation of the warmest quarter decreased abruptly from low to high towards 300 mm or more (Fig 43d). The response curve for elevation showed a positive relationship with predicted rusty-spotted cat suitability where high probabilities occurred at higher elevations (> 1200 m) (Fig 43b). The land cover categories with the highest predicted probability for rusty-spotted cat as calculated includes only degraded areas in the south-eastern part of the reserve. The response curve for annual precipitation of the warmest quarter showed high predicted suitability in areas with low precipitation 150-220 mm (Fig 43d). The MaxEnt model generated a map (predicted probability of occurrence; Fig 44) of potential rusty-spotted cat distribution in Mudumalai showing that high probability areas (considered to be areas with > 0.6 probability of presence) accounted for a small portion (56 km^2) of the reserve with the Linear Quadratic feature type models indicating the suitable habitat. The model output also predicted highly suitable areas in the buffer zone towards the south-eastern region of the reserve

Fig 41. ROC curve of Sensitivity versus Specificity for the habitat model of rusty-spotted cat.

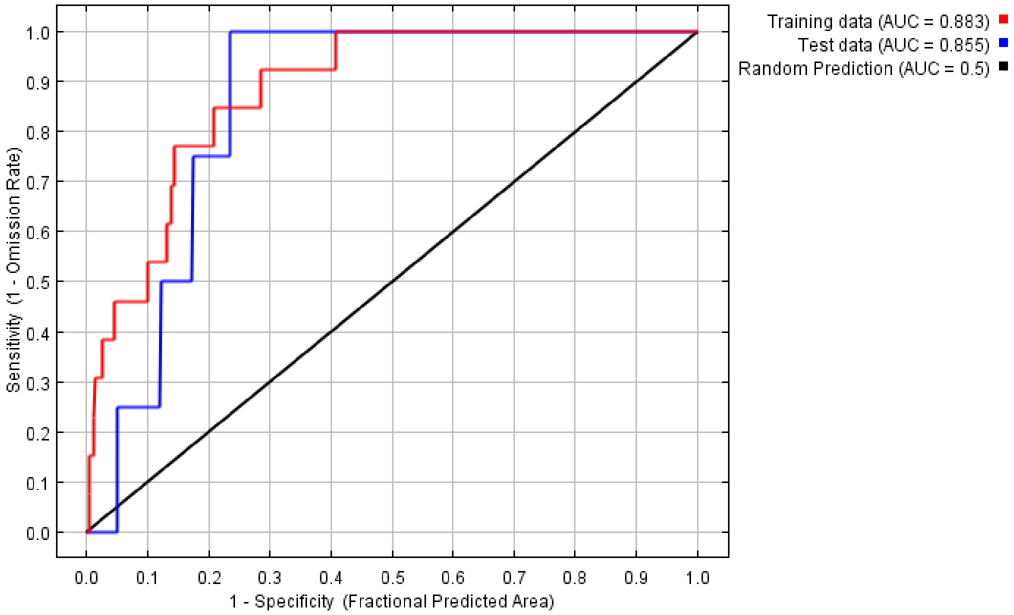


Fig 42. Jackknife analyses of individual predictor variables important in the development of the full model for rusty-spotted cat in relation to the overall model quality or the “regularized training gain.” Dark blue bars indicate the gain achieved when including only that variable and excluding the remaining variables; light blue bars show how much the gain is diminished without the given predictor variable.

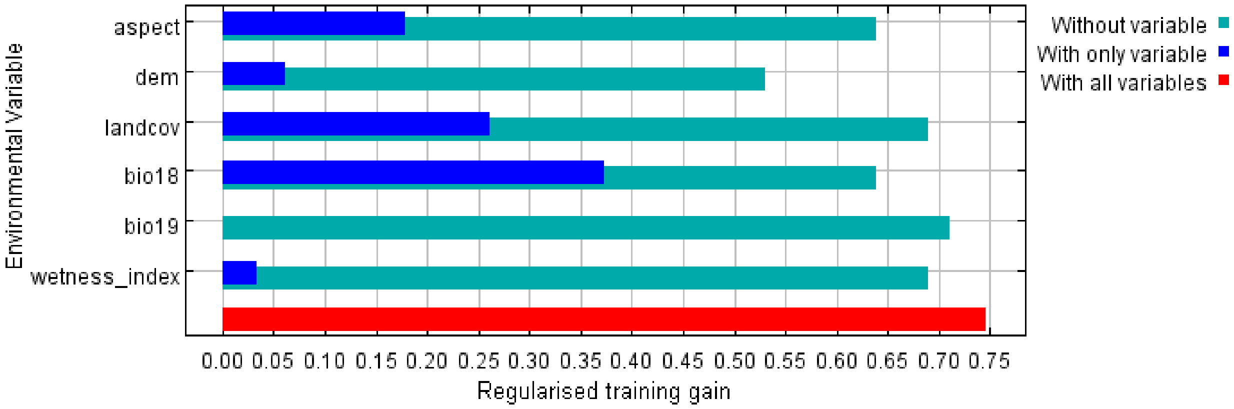


Fig 43. Graphical representation of the relationship between (a) aspect, (b) elevation, c) landcover type, d) annual precipitation of the warmest quarter, e) annual precipitation of the coldest quarter f) topography wetness index, and rusty-spotted cat probability of presence (2009-2011). Each of the curves represents a different MaxEnt model created using only the corresponding variable.

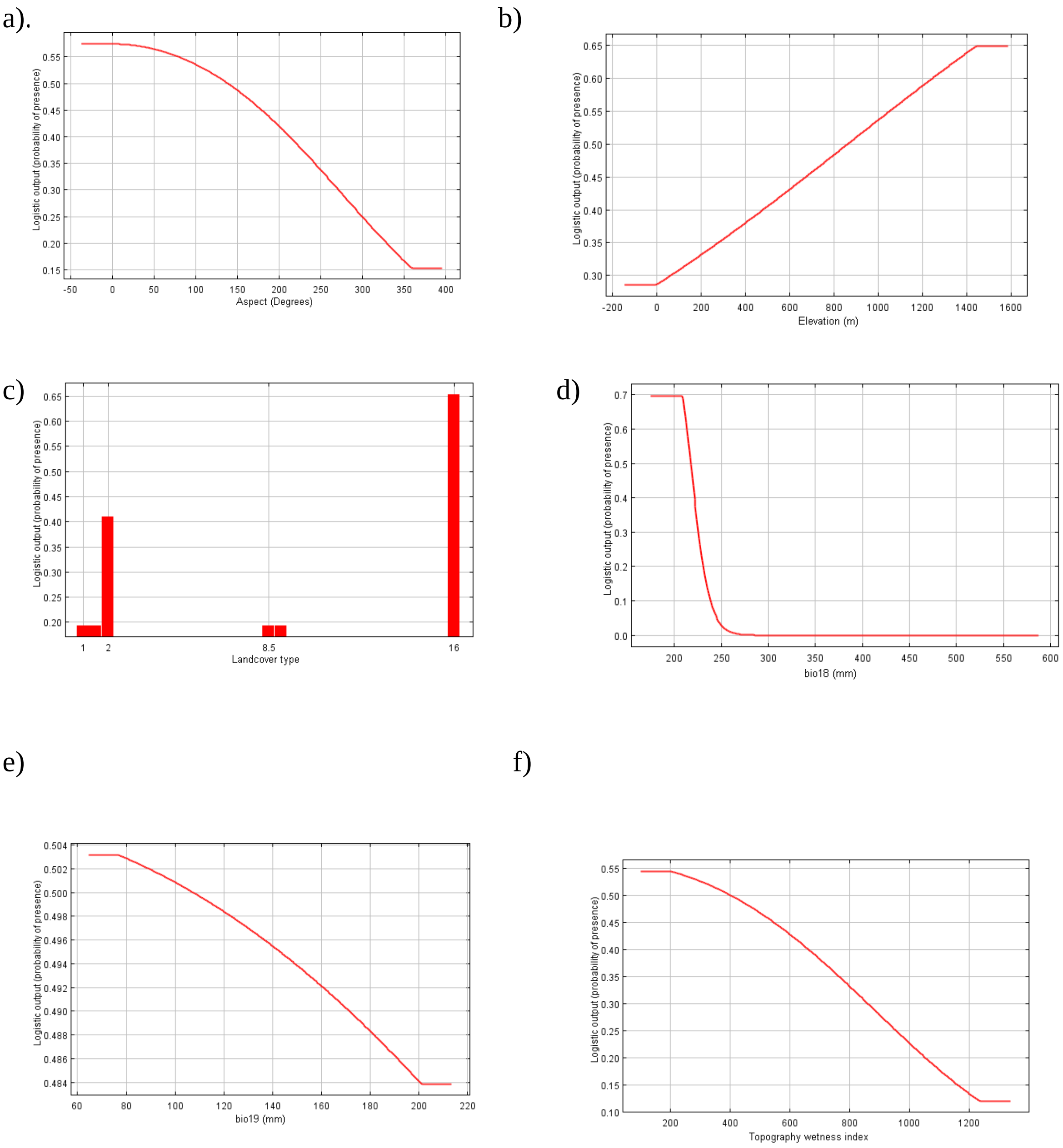
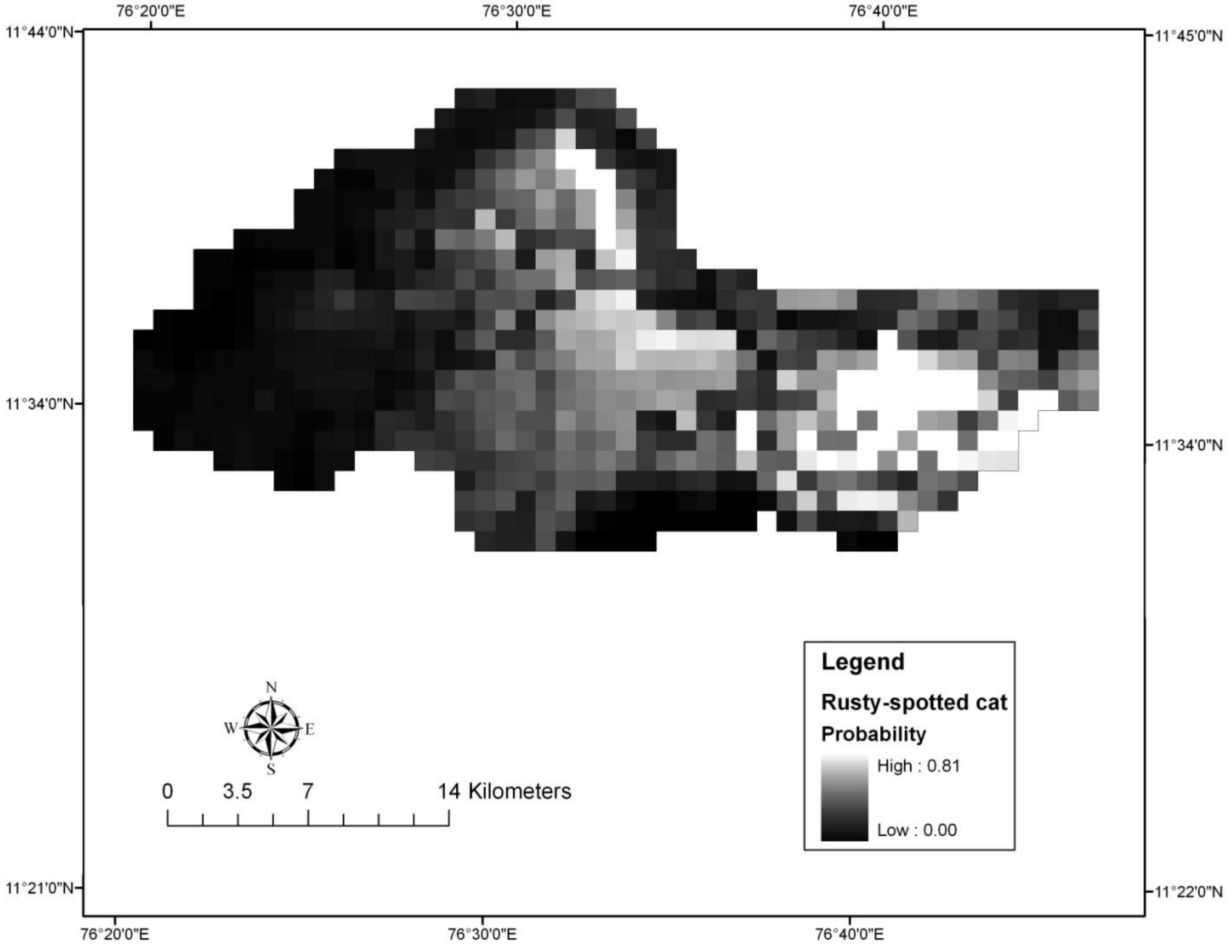


Fig 44. Predicted distribution for rusty-spotted cat in Mudumalai Tiger Reserve estimated by MaxEnt modeling (2009-2011). Potential areas are shown in grey shading with the white color indicating higher probabilities of occurrence.



6.3.3 Leopard cat habitat modeling

Distribution models for leopard cat performed well based on the high (0.89) AUC value (Fig 45). Land cover type had the highest predictive power (67.8%) to the model output (Table 29). The jackknife test of variable importance showed the highest gain when the variable landcover type was used in isolation which therefore appears to have the most useful information by itself (Fig 46). This variable decreased the gain when it was omitted showing that it had the most information that was not present in other variables. The variables landcover type, elevation, isothermality and actual evapotranspiration contributed 96.3% to leopard cat model. Actual evapotranspiration showed an S-shaped curve (Fig 47a) while elevation and isothermality were positively related to predicted leopard cat presence (Fig 47 b, e). Predicted higher probabilities for leopard cat occurred at high elevations (> 1200 m) achieved stabilization beyond 1400 m and areas with high soil water content (evapotranspiration) and isothermaility indicating affinity towards moist and warm localities. The land cover and forest type categories with the highest predicted probability for leopard cat as calculated includes sub-tropical evergreen and dense forests of the reserve (Fig 47c, d). The MaxEnt model generated a map (predicted probability of occurrence; Fig 48) of potential leopard cat distribution in Mudumalai showing that high probability areas (considered to be areas with > 0.6 probability of presence) accounted for a small portion (58 km^2) of the reserve with the Linear feature type model indicating the suitable habitat.

Fig 45. ROC curve of Sensitivity versus Specificity for the habitat model of leopard cat.

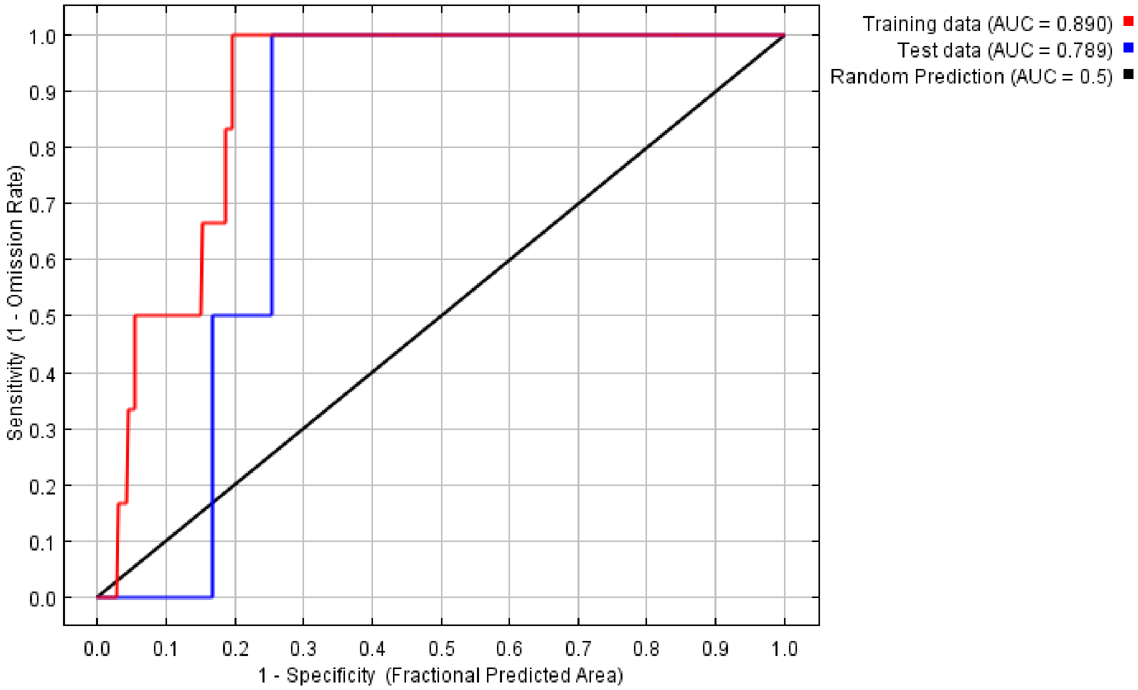


Fig 46. Jackknife analyses of individual predictor variables important in the development of the full model for leopard cat in relation to the overall model quality or the “regularized training gain.” Dark blue bars indicate the gain achieved when including only that variable and excluding the remaining variables; light blue bars show how much the gain is diminished without the given predictor variable.

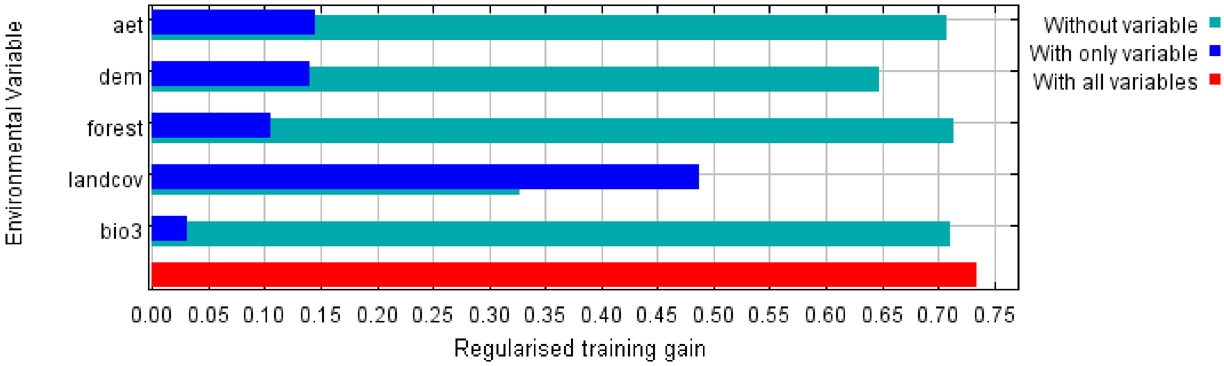
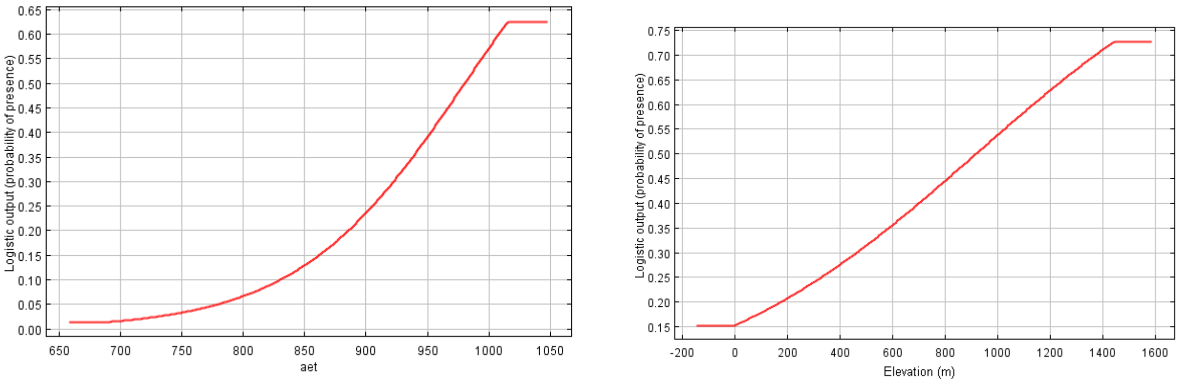
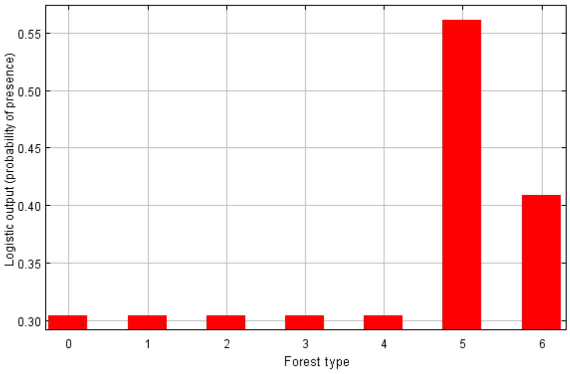


Fig 47. Graphical representation of the relationship between (a) actual evapo-transpiration, (b) elevation, c) forest type, d) landcover type, e) isothermality, and leopard cat probability of presence (2009-2011). Each of the curves represents a different MaxEnt model created using only the corresponding variable.

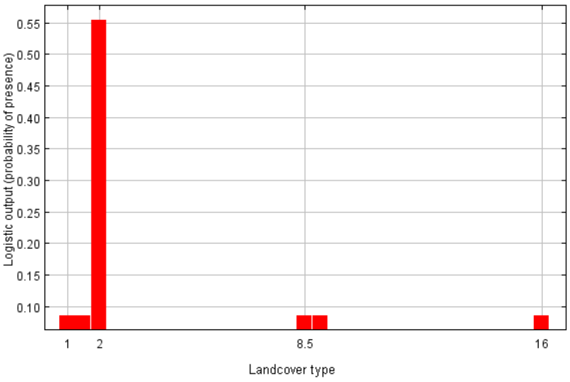
a) b)



c)


Logistic output (probability of presence)
0.55
0.50
0.45
0.40
0.35
0.30
0
1
2
3
4
5
6
Forest type

d)


Logistic output (probability of presence)
0.55
0.50
0.45
0.40
0.35
0.30
0.25
0.20
0.15
0.10
1
2
8.5
16
Landcover type

e)

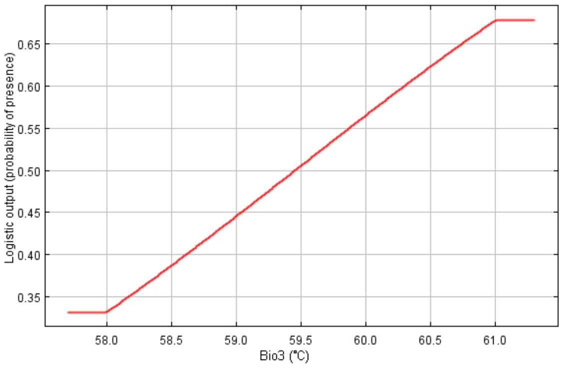
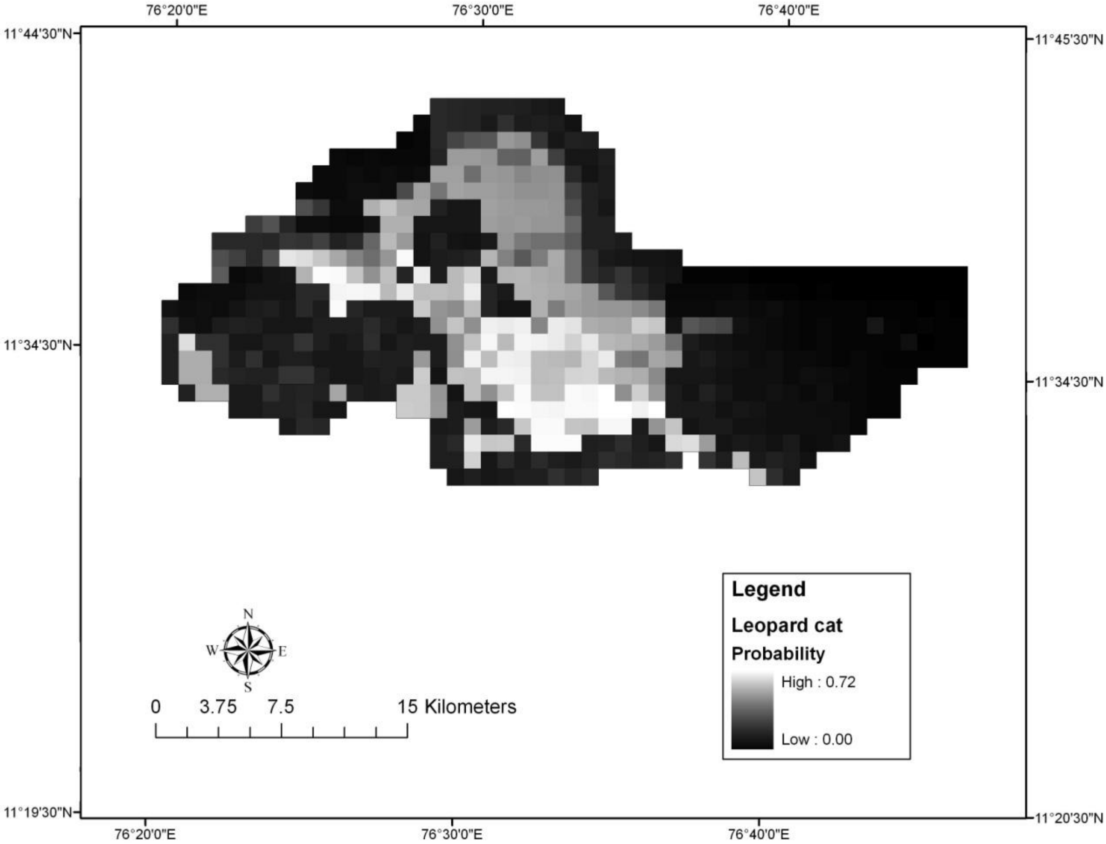

Logistic output (probability of presence)
0.65
0.60
0.55
0.50
0.45
0.40
0.35
58.0
58.5
59.0
59.5
60.0
60.5
61.0
Bio3 (℃)

Fig 48. Predicted distribution for leopard cat in Mudumalai Tiger Reserve estimated by MaxEnt modeling (2009-2011). Potential areas are shown in grey shading with the white color indicating higher probabilities of occurrence.



6.3.4 Small Indian civet habitat modeling

Distribution models for small Indian civet performed well based on the high (0.87) AUC value (Fig 49). Elevation had the highest predictive power (31.7%) to the model output (Table 29). The jackknife test of variable importance showed the highest gain when the variable aspect was used in isolation which therefore appears to have the most useful information by itself (Fig 50). The variable annual precipitation at the coldest quarter decreased the gain when it was omitted showing that it had the most information that was not present in other variables. The variables elevation, aspect, forest type, annual precipitation at the coldest quarter, NDVI (March) and topography wetness index contributed 92.1% to small Indian civet model. NDVI (March) and wetness index reponse curve showed a bimodal peak with probability of small Indian civet distribution (Fig 51a). Medium aspect (100˚) and elevation (400-

900 m) led to high probabilities (Fig 51b, c). Probabilities were skewed towards plain areas (1°, Fig 51g) and then stabilized from 5° onwards. High probabilities peaked at 100 mm of annual precipitation of the coldest quarter (Fig 51f). The MaxEnt model generated a map (predicted probability of occurrence; Fig 52) of potential small Indian civet distribution in Mudumalai showing that high probability areas (considered to be areas with > 0.6 probability of presence) accounted for a small portion (49 km^2) of the reserve with the Linear feature type model indicating the suitable habitat.

Fig 49. ROC curve of Sensitivity versus Specificity for the habitat model of small Indian civet.

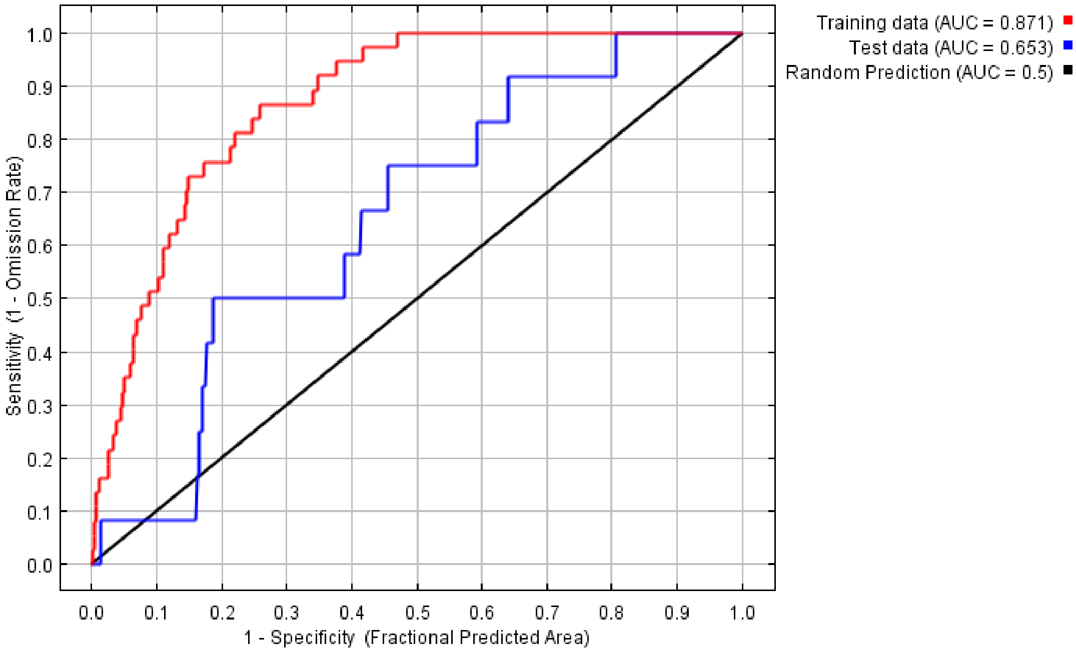


Fig 50. Jackknife analyses of individual predictor variables important in the development of the full model for small Indian civet in relation to the overall model quality or the "regularized training gain." Dark blue bars indicate the gain achieved when including only that variable and excluding the remaining variables; light blue bars show how much the gain is diminished without the given predictor variable.

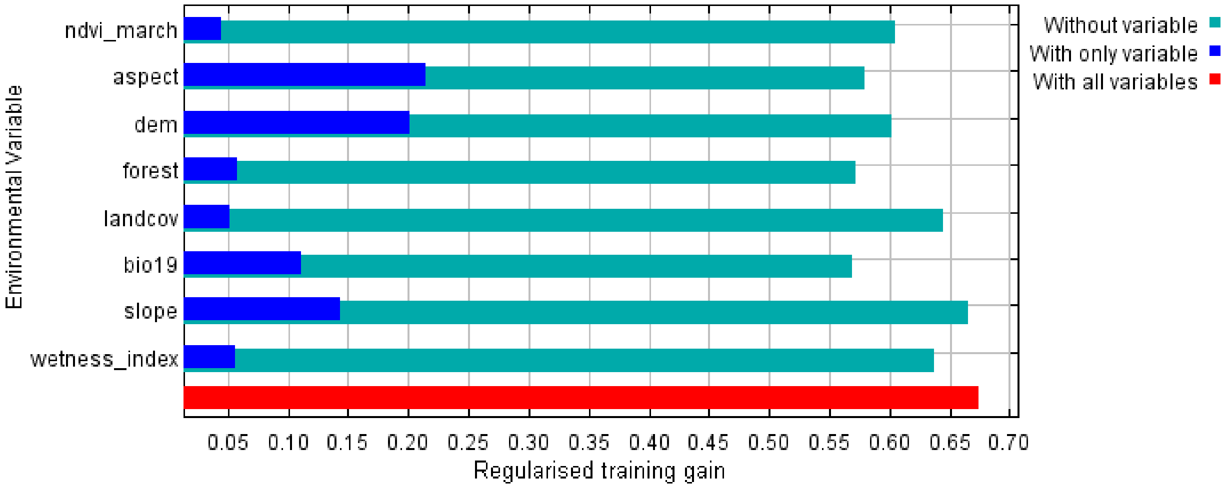


Fig 51. Graphical representation of the relationship between (a) NDVI_March, (b) aspect, c) elevation, d) forest type, e) landcover type f) annual precipitation of the coldest quarter g) slope, h) topography wetness index, and small Indian civet probability of presence (2009-2011). Each of the curves represents a different MaxEnt model created using only the corresponding variable.

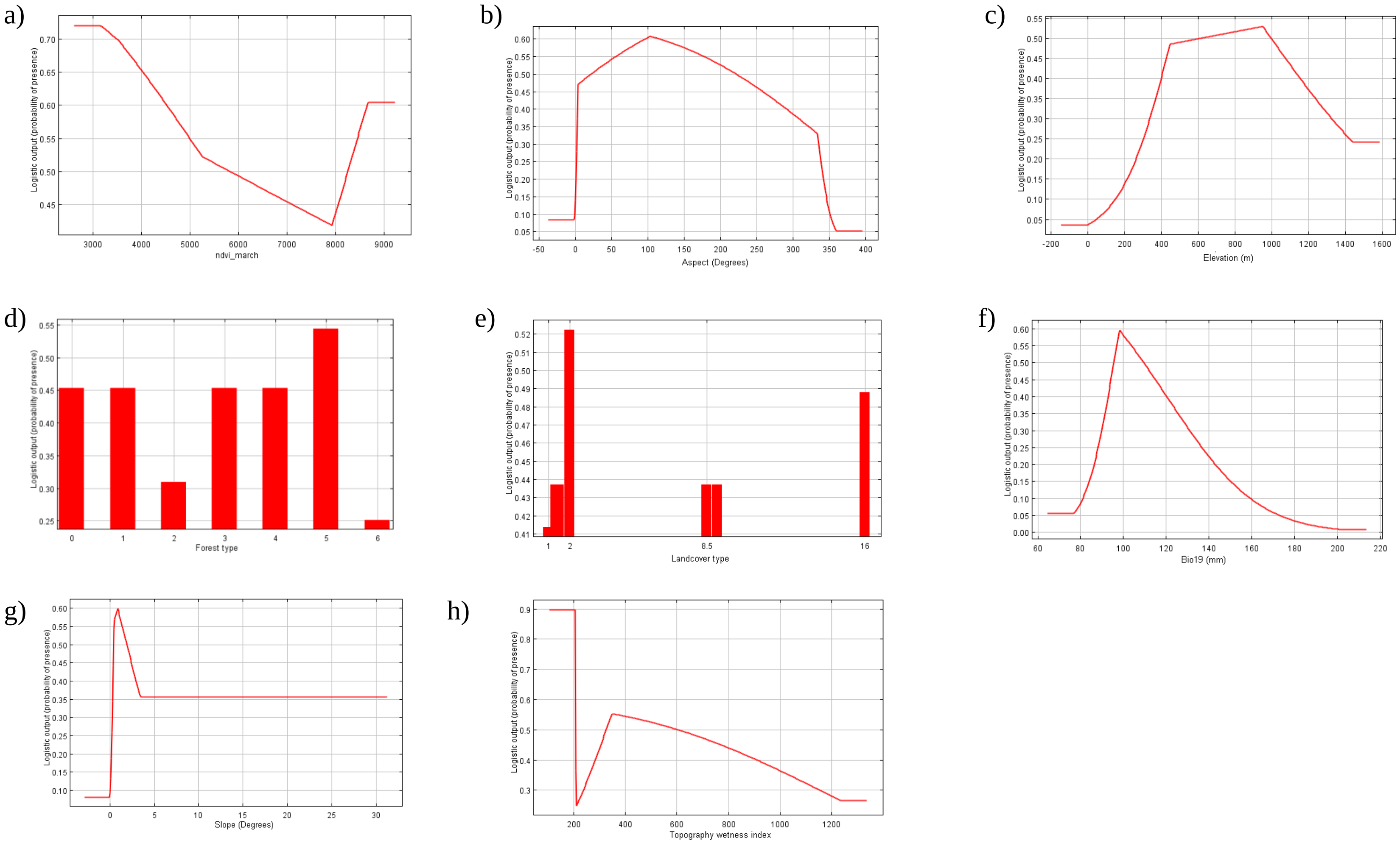
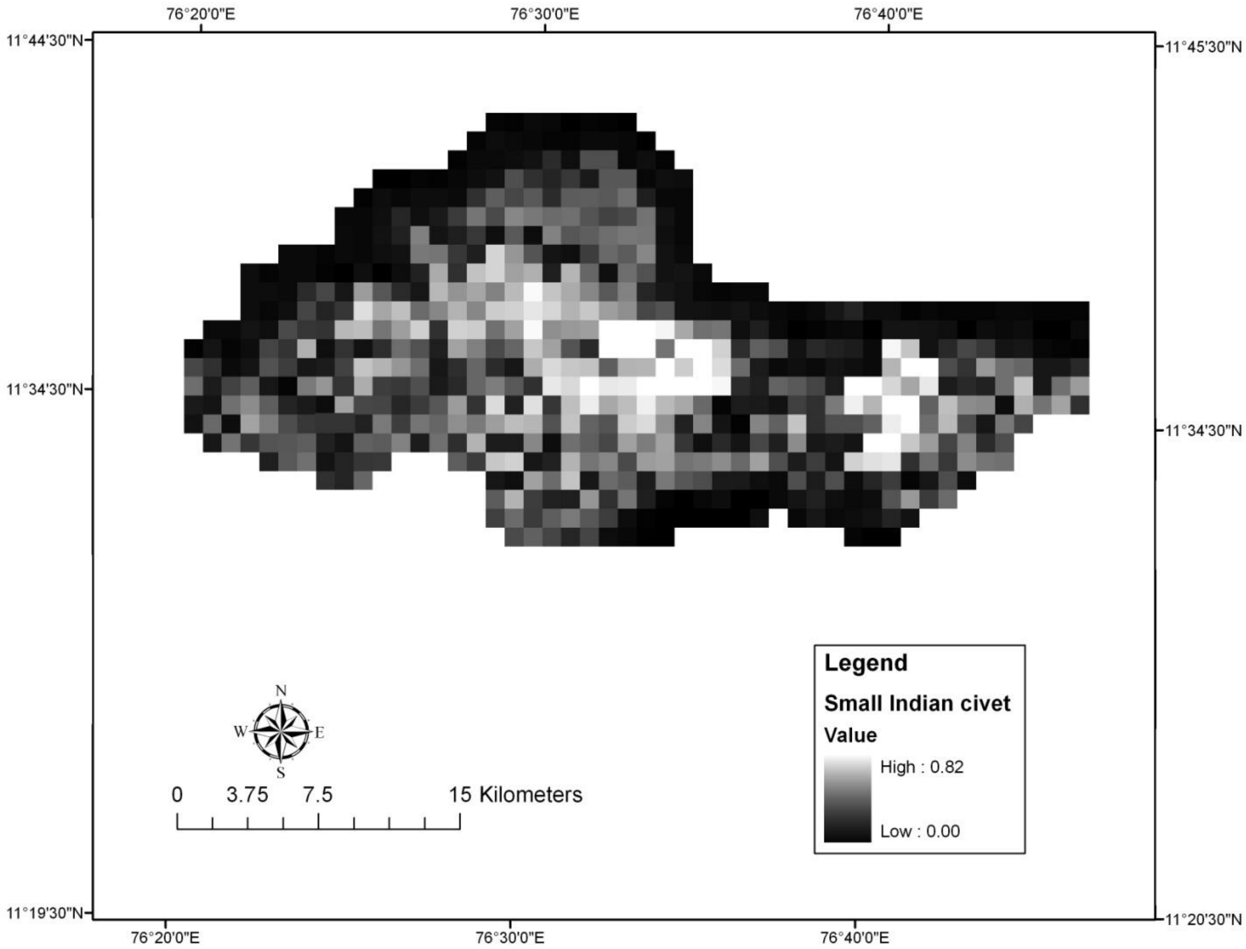


Fig 52. Predicted distribution for small Indian civet in Mudumalai Tiger Reserve estimated by MaxEnt modeling (2009-2011). Potential areas are shown in grey shading with the white color indicating higher probabilities of occurrence.



6.3.5 Common palm civet habitat modeling

Distribution models for common palm civet performed well based on the high (0.91) AUC value (Fig 53). Annual precipitation of the warmest quarter had the highest predictive power (39%) to the model output (Table 29). The jackknife test of variable importance showed the highest gain when the variable annual precipitation of the warmest quarter was used in isolation which therefore appears to have the most useful information by itself (Fig 54). The variable elevation decreased the gain when it was omitted showing that it had the most information that was not present in other variables. The variables annual precipitation of the warmest quarter, elevation, actual evapotranspiration, landcover type and forest contributed 90.4% to common palm civet model. Annual precipitation of the warmest quarter was skewed towards 210 mm (Fig 55a). Actual evapotranspiration peaked at 825 mm and elevation at 850 m (Fig 55b, e). Common palm civet probabilities were predicted highest in non-forest areas (Fig 55c) and dry deciduous habitats (Fig 55d). Predicted high probabilities were achieved from low to high NDVI (July) and then gradually dropped down towards highest NDVI values (Fig 55f) while NDVI (June) did not bring about any change in common palm civet probabilities (Fig 55g). Topography wetness index decreased abruptly from low values and remained constant towards 400 or more (Fig 55h). The MaxEnt model generated a map (predicted probability of occurrence; Fig 56) of potential common palm civet distribution in Mudumalai showing that high probability areas (considered to be areas with > 0.6 probability of presence) accounted for a small portion (23 km^2) of the reserve with the Linear Quadratic Hinge feature type model indicating the suitable habitat.

Fig 53. ROC curve of Sensitivity versus Specificity for the habitat model of common palm civet.

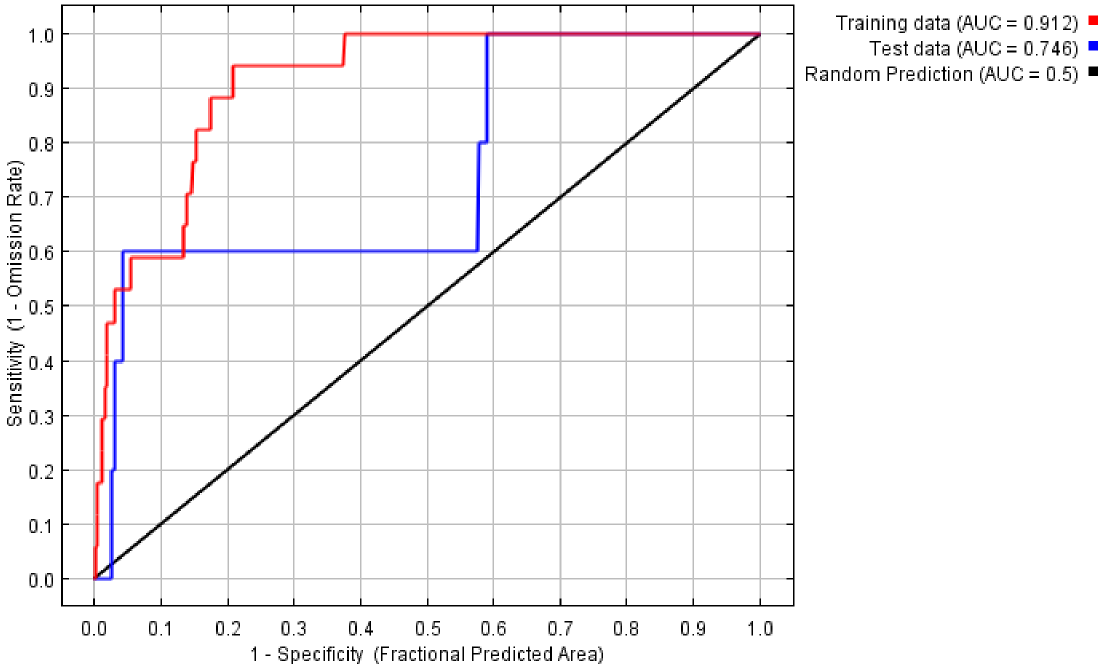


Fig 54. Jackknife analyses of individual predictor variables important in the development of the full model for common palm civet in relation to the overall model quality or the “regularized training gain.” Dark blue bars indicate the gain achieved when including only that variable and excluding the remaining variables; light blue bars show how much the gain is diminished without the given predictor variable.

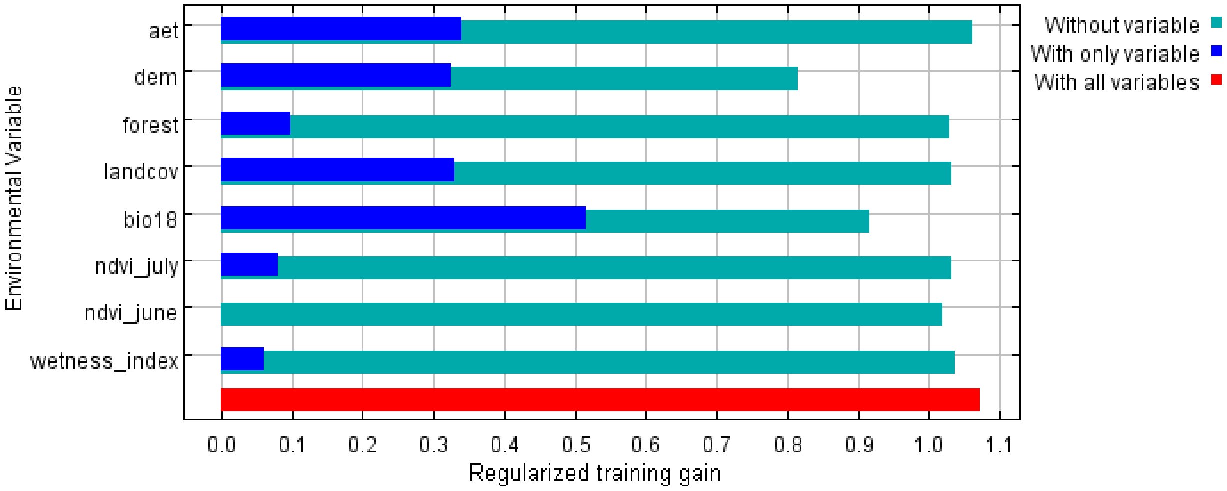


Fig 55. Graphical representation of the relationship between (a) precipitation of the warmest quarter, (b) AET, c) forest type, d) landcover type, e) elevation, f) NDVI_July, g) NDVI_June, h) topography wetness index and common palm civet probability of presence (2009-2011). Each of the curves represents a different MaxEnt model created using only the corresponding variable.

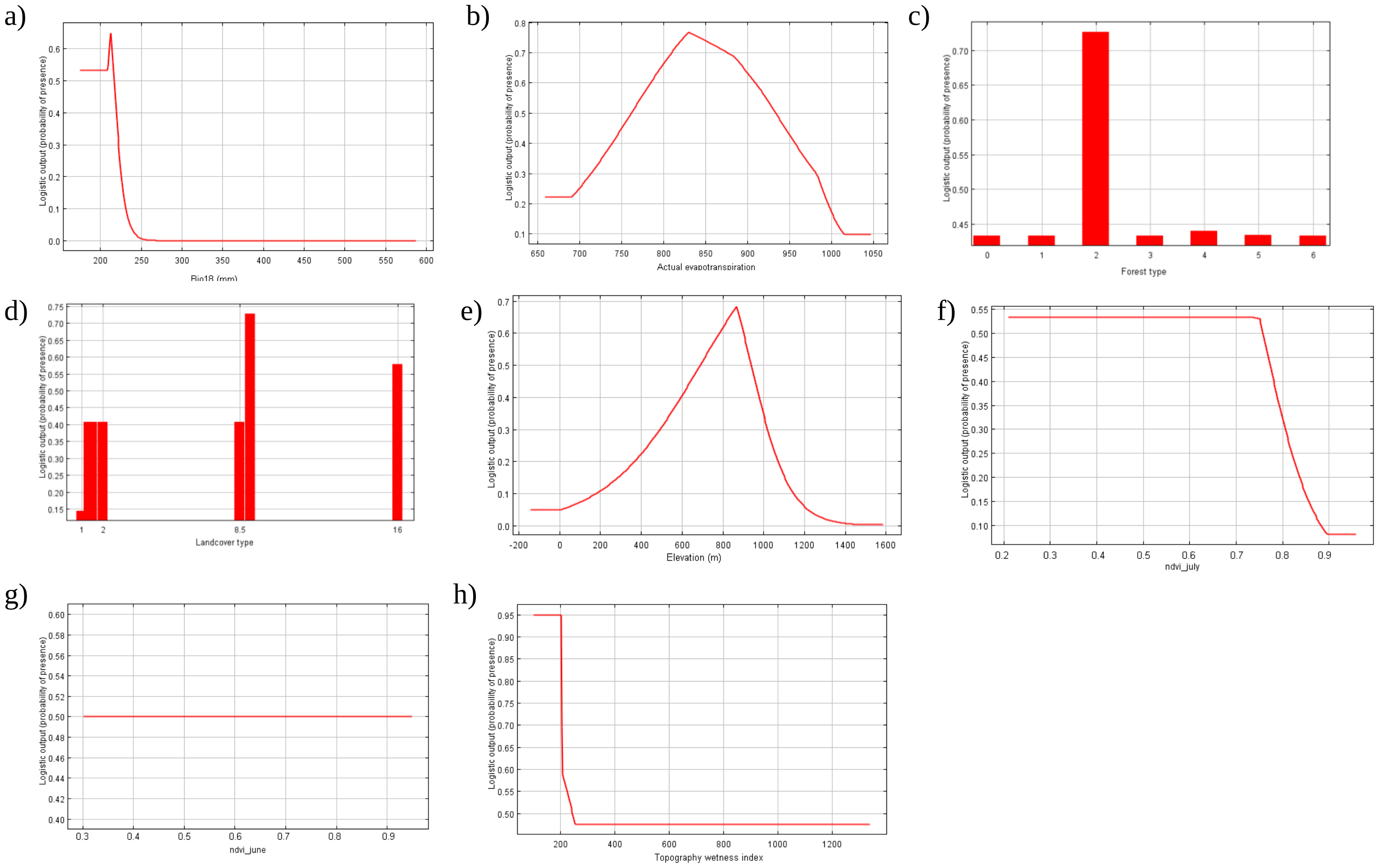
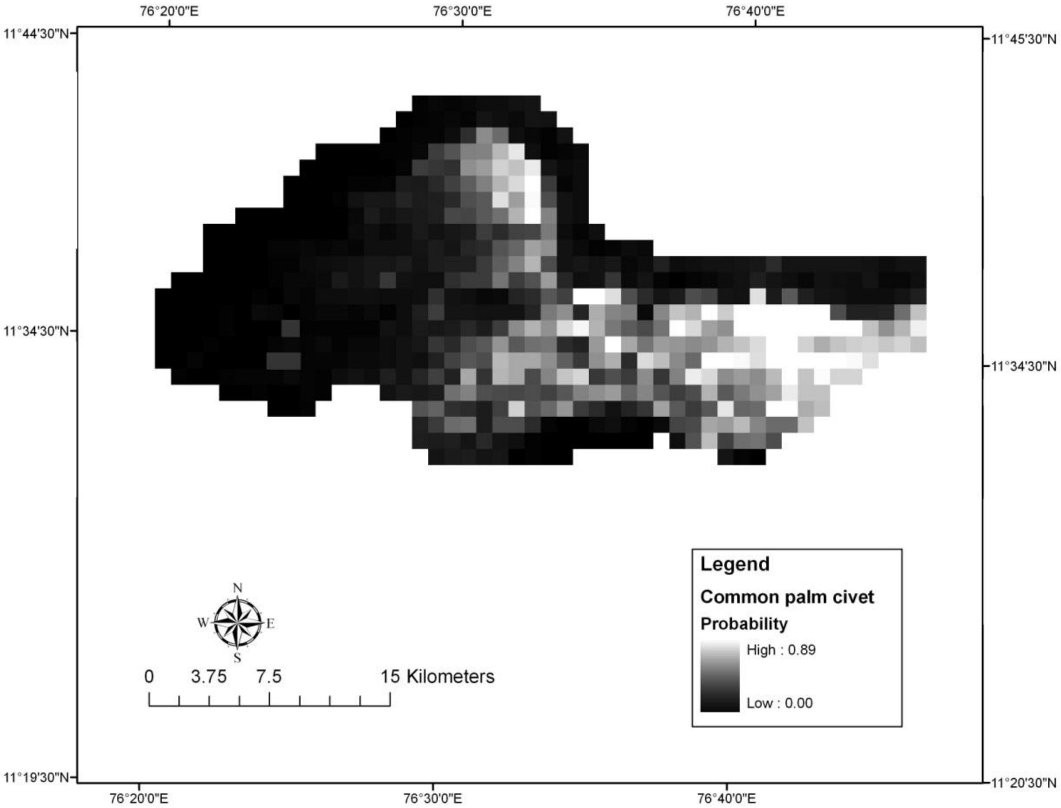


Fig 56. Predicted distribution for common palm civet in Mudumalai Tiger Reserve estimated by MaxEnt modeling (2009-2011). Potential areas are shown in grey shading with the white color indicating higher probabilities of occurrence.



6.3.6 Brown palm civet habitat modeling

Distribution models for brown palm civet performed well based on the high (0.95) AUC value (Fig 57). Annual precipitation of the warmest quarter had the highest predictive power (39.7%) to the model output (Table 29). The jackknife test of variable importance showed the highest gain when the variable NDVI (March) was used in isolation which therefore appears to have the most useful information by itself (Fig 58). The variable annual precipitation of the coldest quarter decreased the gain when it was omitted showing that it had the most information that was not present in other variables. The variables NDVI (March), annual precipitation of the coldest and warmest quarter and elevation contributed 93.8% to brown palm civet model. Predicted probabilities were skewed towards high NDVI (March) and actual evapotranspiration (0.78 and 980 mm respectively, Fig 59a, b) and moderate aspect (260˚, Fig 59c). Probabilities showed an S-shaped curve from low values and gradually achieving stabilization at higher elevation (> 1400 m, Fig 59d) and a bell-shaped curved with NDVI (June) (Fig 59 g). Annual precipitation of the warmest and coldest quarter showed skewed response curves towards low values (240 and 100 mm respectively, Fig 59e, f). The MaxEnt model generated a map (predicted probability of occurrence; Fig 60) of potential brown palm civet distribution in Mudumalai showing that high probability areas (considered to be areas with > 0.6 probability of presence) accounted for a small portion (34 km^2) in the north-western part of the reserve with the Linear Quadratic Hinge feature type model indicating the suitable habitat.

Fig 57. ROC curve of Sensitivity versus Specificity for the habitat model of brown palm civet.

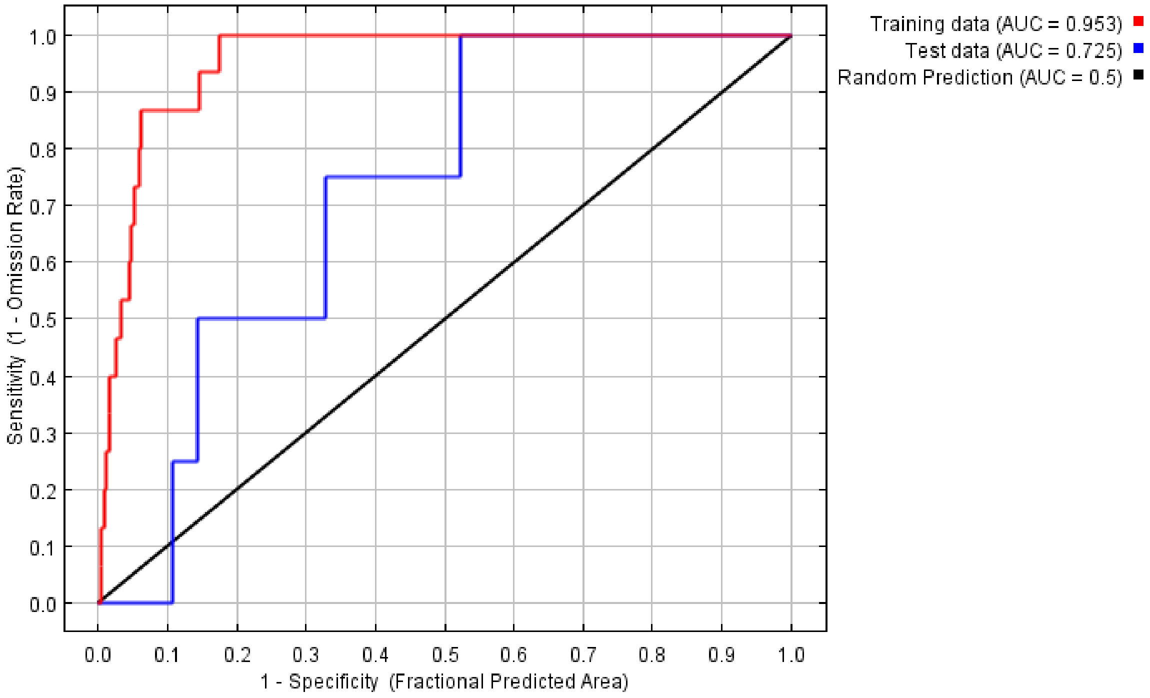


Fig 58. Jackknife analyses of individual predictor variables important in the development of the full model for brown palm civet in relation to the overall model quality or the “regularized training gain.” Dark blue bars indicate the gain achieved when including only that variable and excluding the remaining variables; light blue bars show how much the gain is diminished without the given predictor variable.

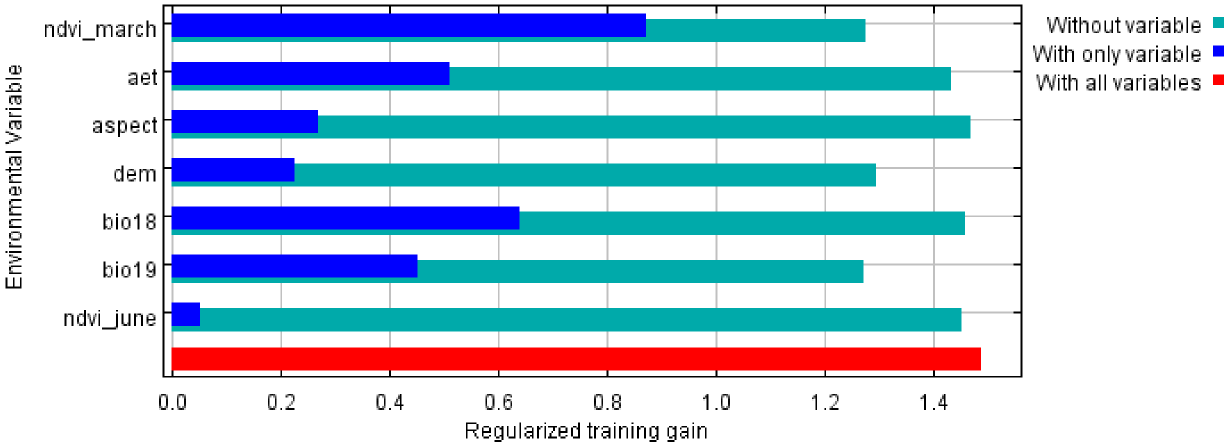
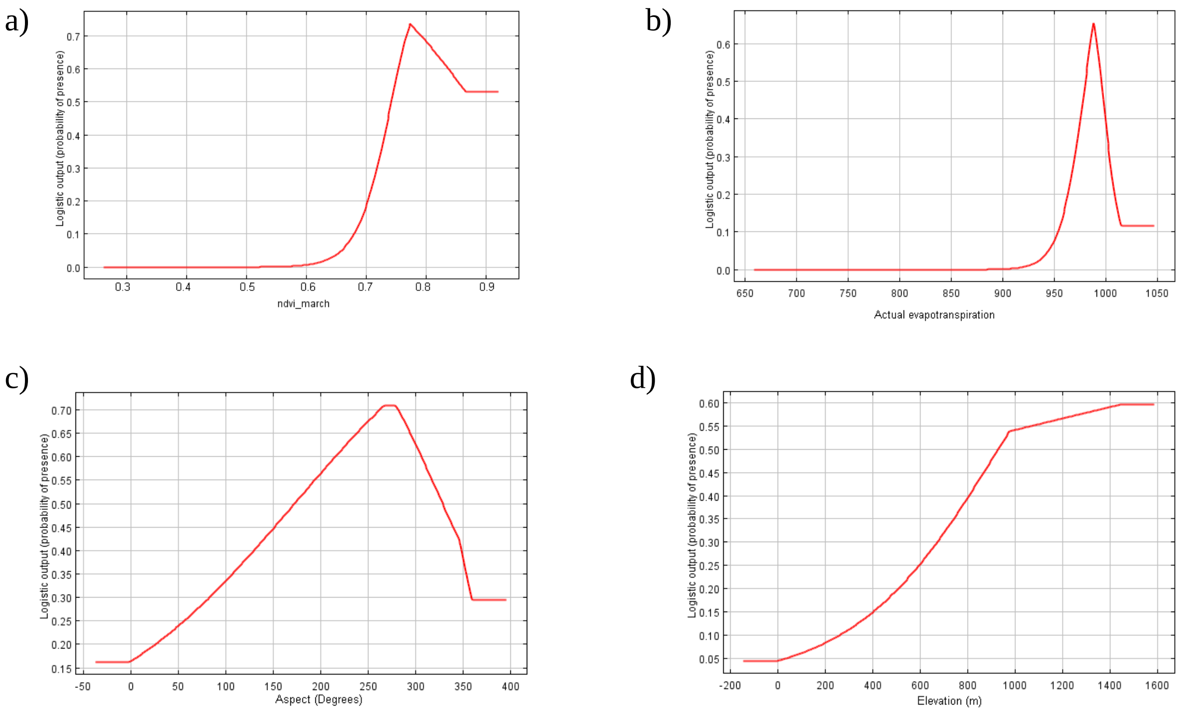
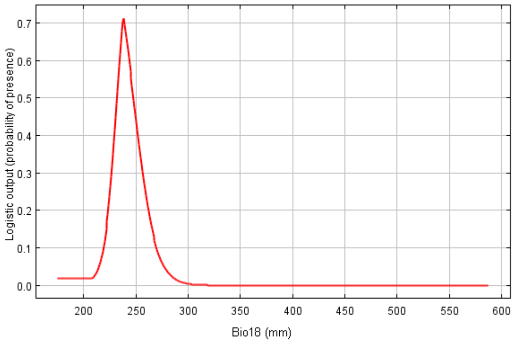


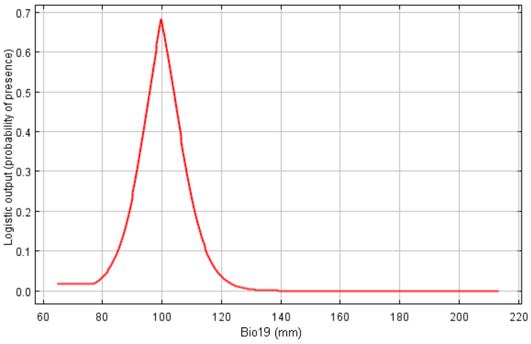
Fig 59. Graphical representation of the relationship between (a) NDVI (March), (b) actual evapotranspiration, c) aspect, d) elevation, e) annual precipitation of the warmest quarter, f) annual precipitation of the coldest quarter g) NDVI (June) and brown palm civet probability of presence (2009-2011). Each of the curves represents a different MaxEnt model created using only the corresponding variable.



e)



f)



g)

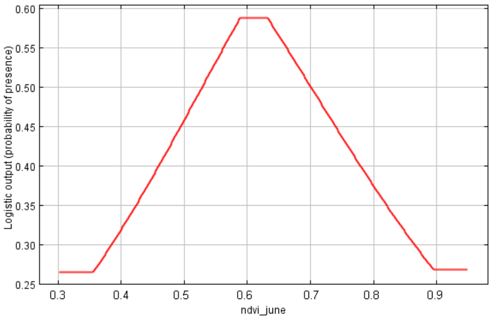
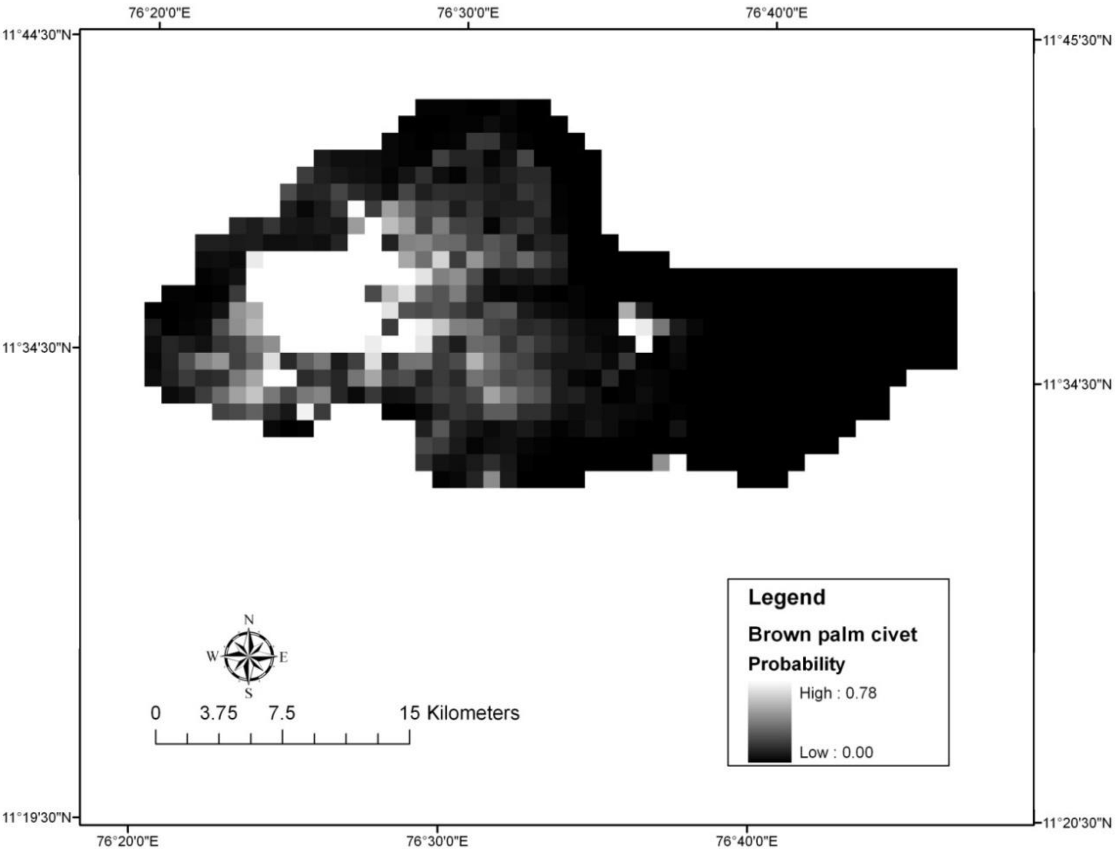


Fig 60. Predicted distribution for brown palm civet in Mudumalai Tiger Reserve estimated by MaxEnt modeling (2009-2011). Potential areas are shown in grey shading with the white color indicating higher probabilities of occurrence.



6.3.7 Stripe-necked mongoose habitat modeling

Distribution models for stripe-necked mongoose performed well based on the high (0.83) AUC value (Fig 61). Elevation had the highest predictive power (41.7%) to the model output (Table 29). The jackknife test of variable importance showed the highest gain when elevation was used in isolation which therefore appears to have the most useful information by itself (Fig 62). This variable decreased the gain when it was omitted showing that it had the most information that was not present in other variables. The variables elevation, landcover type, annual precipitation of the coldest quarter, aspect, slope, actual evapotranspiration and forest type contributed to 94.6% for stripe-necked mongoose model. Predicted probabilities were skewed towards high actual evapotranspiration (Fig 63a). High probabilities occurred at medium aspect (Fig 63b), elevation (200-900 m, Fig 63c) and landcover types; subtropical evergreen and dry deciduous forests (Fig 63e). Probabilities were skewed towards low annual precipitation of the warmest and coldest quarter (Fig 63f, g). The response curve for slope showed a bi-modal curve (Fig 63h) and probabilities against topography wetness index dropped abruptly from 200 and gradually decreased towards high values (Fig 63i). The MaxEnt model generated a map (predicted probability of occurrence; Fig 64) of potential stripe-necked mongoose distribution in Mudumalai showing that high probability areas (considered to be areas with > 0.6 probability of presence) accounted for a substantial portion (57 km^2) depicting a random probabilistic distribution in the reserve with the Linear Quadratic Hinge feature type model indicating suitable habitat.

Fig 61. ROC curve of Sensitivity versus Specificity for the habitat model of stripe-necked mongoose

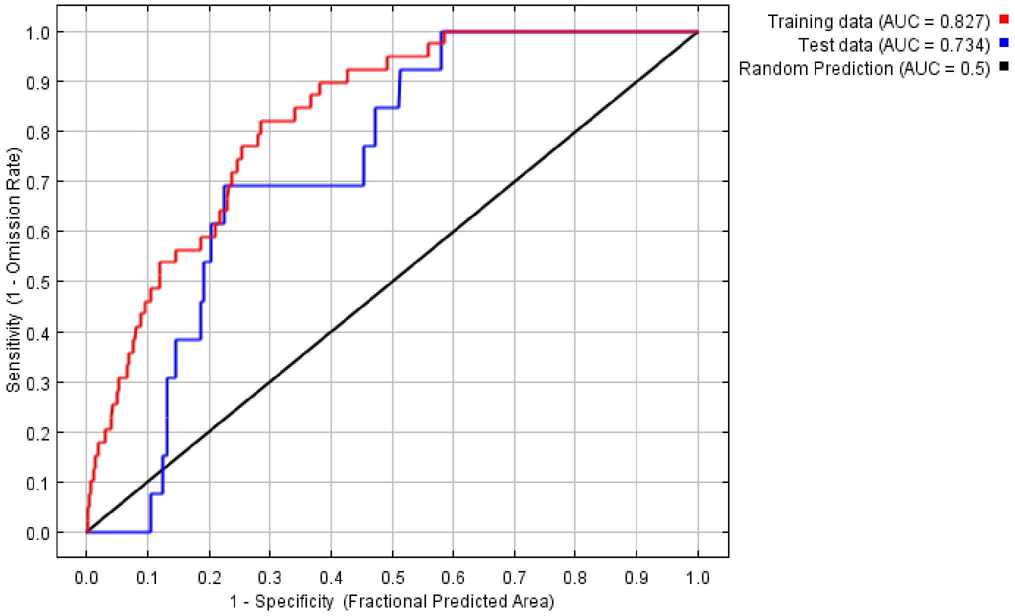


Fig 62. Jackknife analyses of individual predictor variables important in the development of the full model for stripe-necked mongoose in relation to the overall model quality or the "regularized training gain." Dark blue bars indicate the gain achieved when including only that variable and excluding the remaining variables; light blue bars show how much the gain is diminished without the given predictor variable.

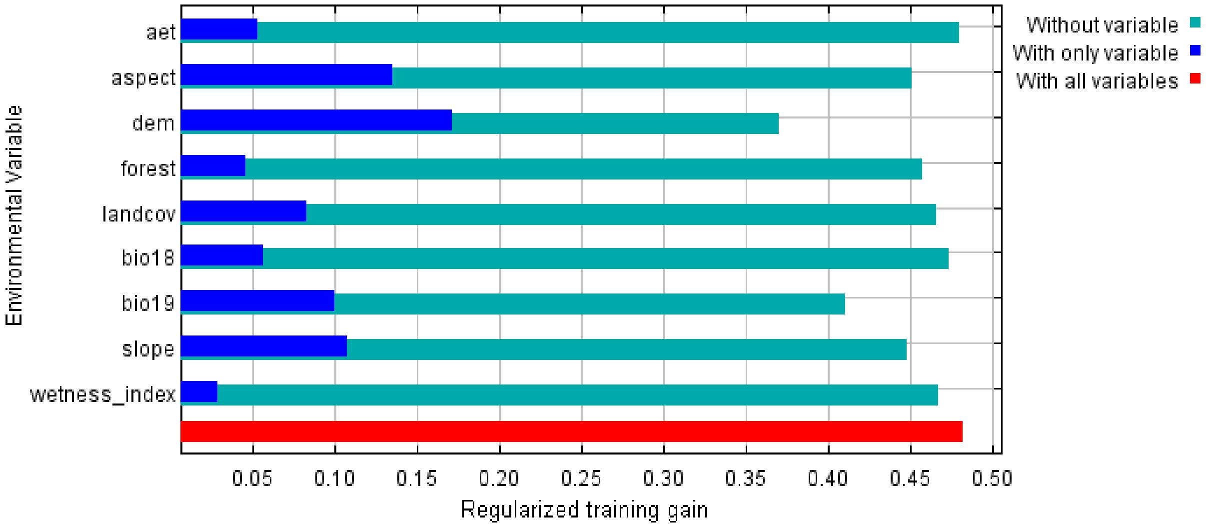


Fig 63. Graphical representation of the relationship between (a) actual evapotranspiration, (b) aspect, c) elevation, d) forest type e) landcover type f) annual precipitation of the warmest quarter, g) annual precipitation of the coldest quarter h) slope i) topography wetness index and stripe-necked mongoose probability of presence (2009-2011). Each of the curves represents a different MaxEnt model created using only the corresponding variable.

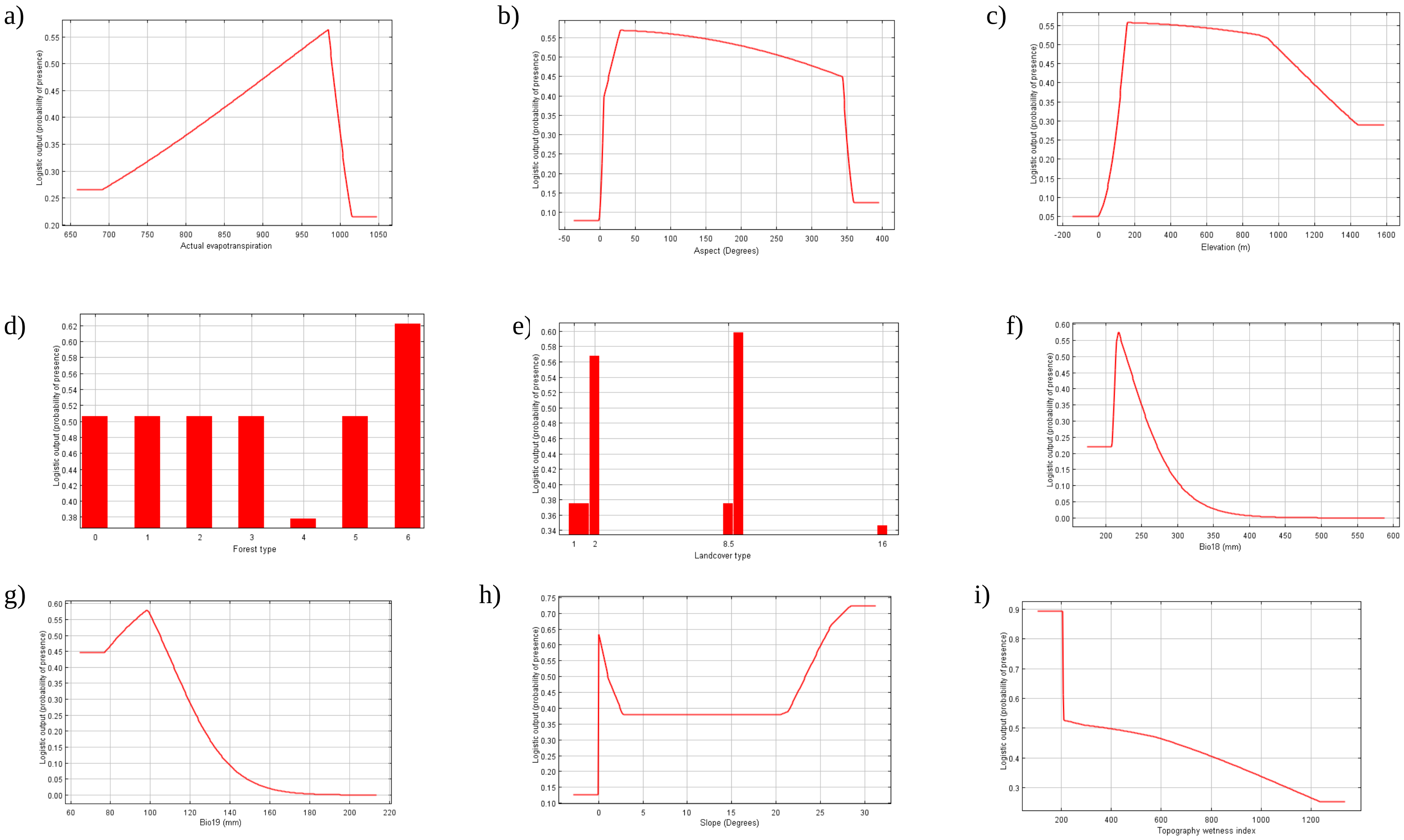
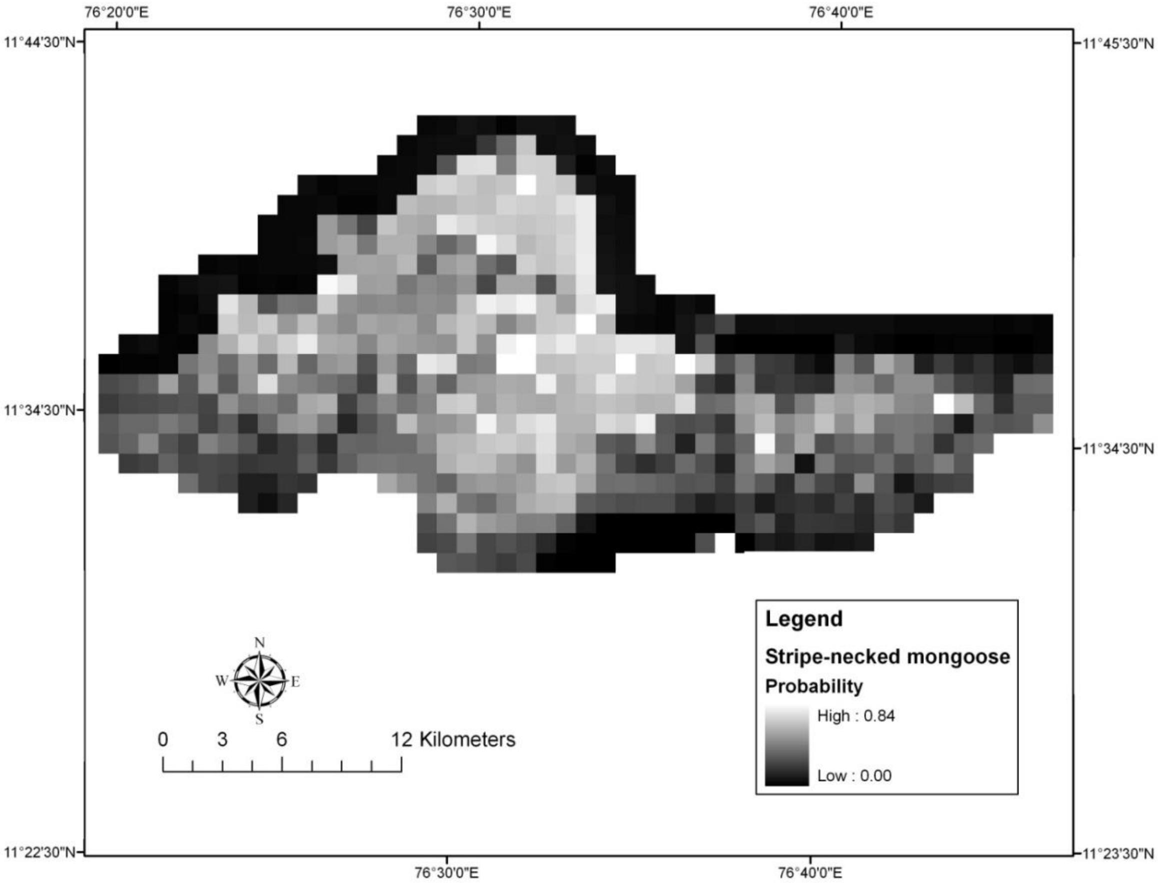


Fig 64. Predicted distribution for stripe-necked mongoose in Mudumalai Tiger Reserve estimated by MaxEnt modeling (2009-2011). Potential areas are shown in grey shading with the white color indicating higher probabilities of occurrence.



6.3.8 Ruddy mongoose habitat modeling

Distribution models for ruddy mongoose performed well based on the high (0.91) AUC value (Fig 65). Elevation had the highest predictive power (21.6%) to the model output (Table 29). The jackknife test of variable importance showed the highest gain when annual precipitation of the warmest quarter was used in isolation which therefore appears to have the most useful information by itself (Fig 66). This variable aspect decreased the gain when it was omitted showing that it had the most information that was not present in other variables. The variables elevation, aspect, annual precipitation of the warmest quarter, NDVI (March), landcover type, forest type, annual precipitation of the coldest quarter and actual evapotranspiration contributed to 96% of ruddy mongoose model. The response curve for NDVI (March) and distance to water showed a bi-modal curve for predicted ruddy mongoose probabilities (Fig 67a, d). Actual evapotranspiration showed almost a bell-shaped curve with probabilities occurring at moderate values (Fig 67b). Probabilities were skewed towards 40-60° of aspect (Fig 67c), 220 mm of annual precipitation of the warmest quarter (Fig 67h) and 100 mm of the coldest quarter (Fig 67i). Probabilities dropped abruptly at lowest topography wetness index (200, Fig 67j) The MaxEnt model generated a map (predicted probability of occurrence; Fig 68) of potential ruddy distribution in Mudumalai showing that high probability areas (considered to be areas with > 0.6 probability of presence) accounted for a substantial portion (40 km^2) towards the eastern region in the reserve with the Linear Quadratic Hinge feature type model indicating suitable habitat.

Fig 65. ROC curve of Sensitivity versus Specificity for the habitat model of ruddy mongoose.

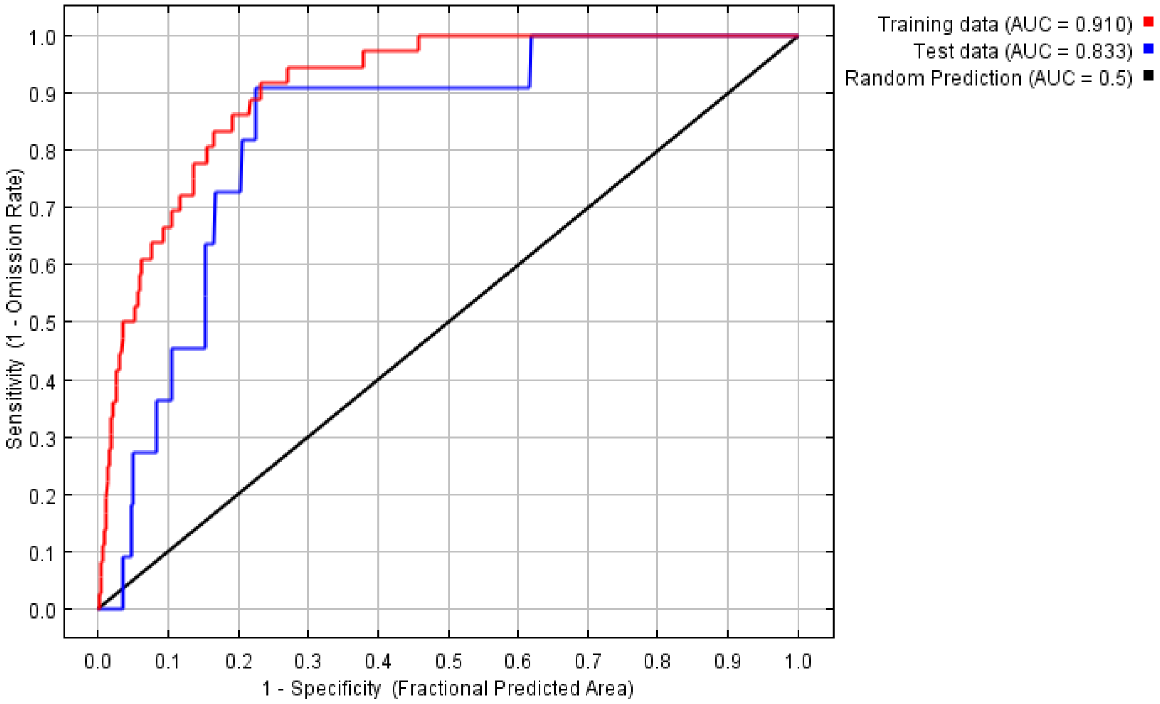


Fig 66. Jackknife analyses of individual predictor variables important in the development of the full model for ruddy mongoose in relation to the overall model quality or the "regularized training gain." Dark blue bars indicate the gain achieved when including only that variable and excluding the remaining variables; light blue bars show how much the gain is diminished without the given predictor variable.

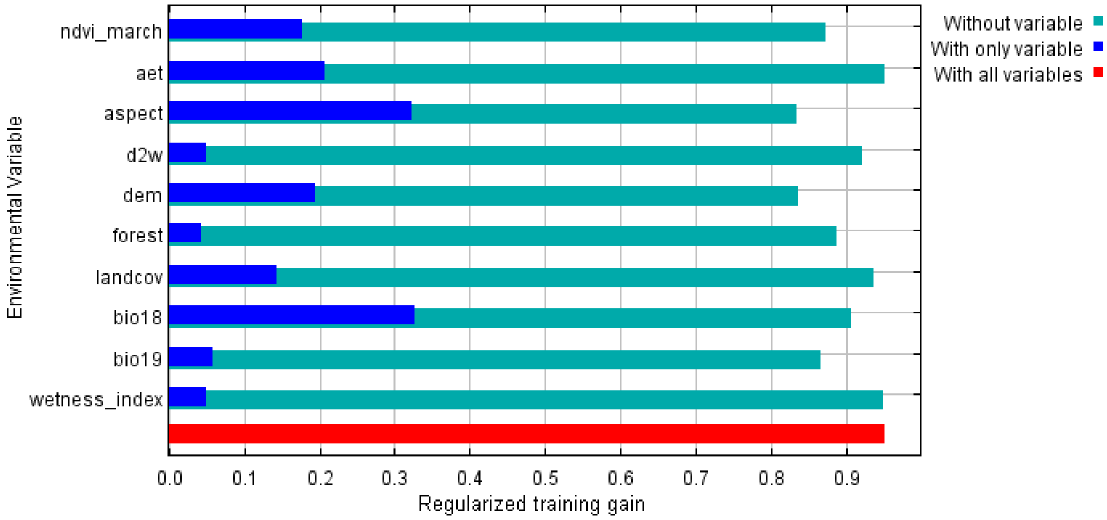
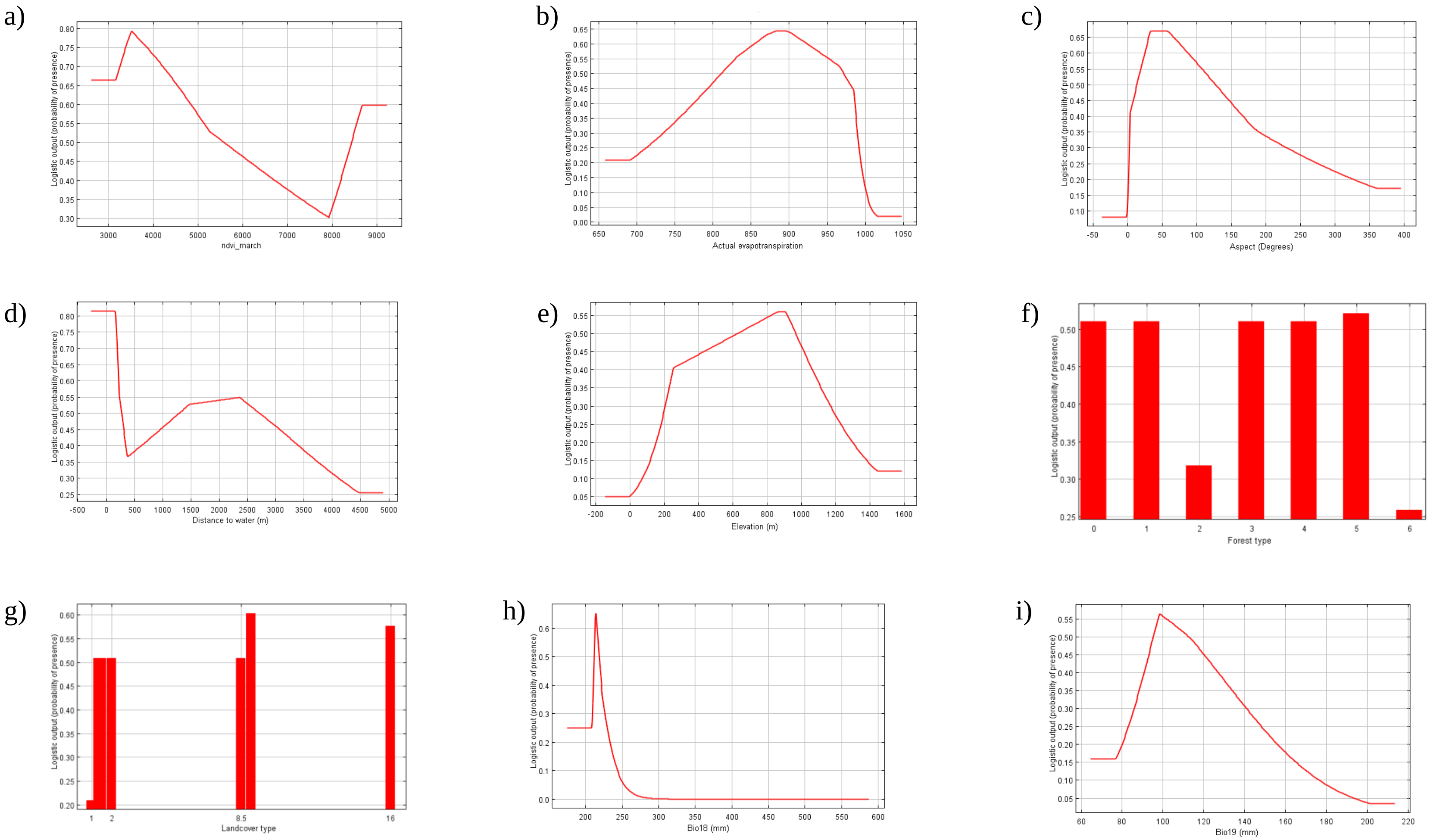


Fig 67. Graphical representation of the relationship between (a) NDVI (March), (b) actual evapotranspiration, c) aspect, d) distance to water e) elevation f) forest type, g) landcover type h) annual precipitation of the warmest quarter i) annual precipitation of the coldest quarter topography wetness index and ruddy mongoose probability of presence (2009-2011). Each of the curves represents a different MaxEnt model created using only the corresponding variable.



j)

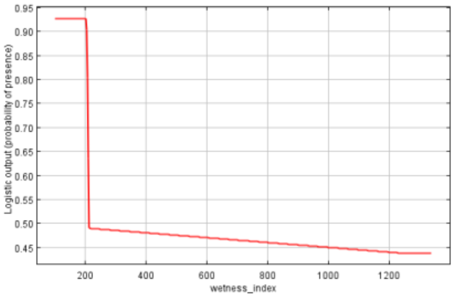
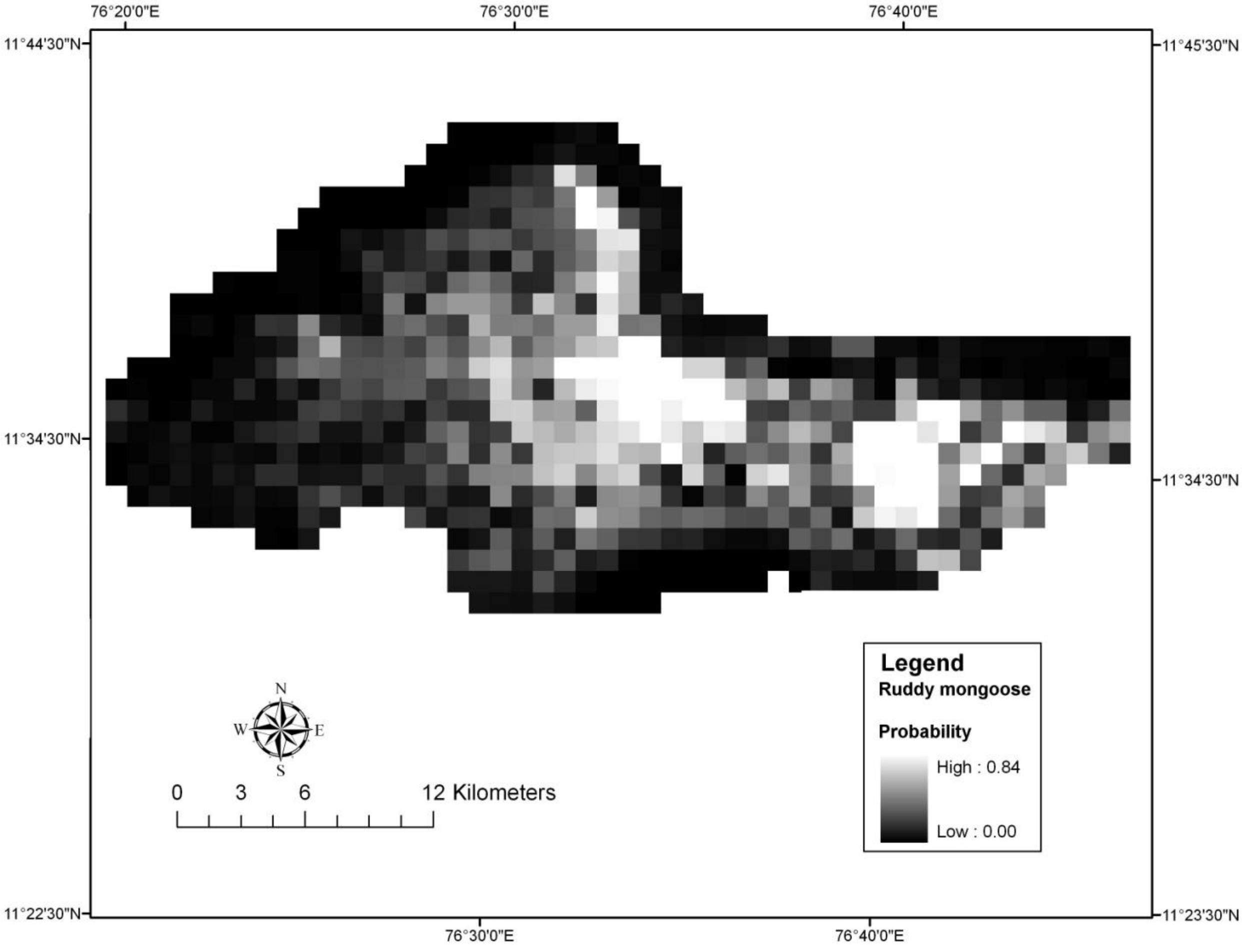


Fig 68. Predicted distribution for ruddy mongoose in Mudumalai Tiger Reserve estimated by MaxEnt modeling (2009-2011). Potential areas are shown in grey shading with the white color indicating higher probabilities of occurrence.



6.3.9 Grey mongoose habitat modeling

Distribution models for grey mongoose performed well based on the high (0.92) AUC value (Fig 69). Annual precipitation of the warmest quarter and landcover categories had the highest predictive power (34.5% and 33.3%) to the model output

(Table 29). The jackknife test of variable importance showed the highest gain when annual precipitation of the warmest quarter was used in isolation which therefore appears to have the most useful information by itself (Fig 70). This variable also decreased the gain when it was omitted showing that it had the most information that was not present in other variables. The annual precipitation of the warmest quarter, landcover type, elevation, actual evapotranspiration, NDVI (June) contributed to 94% for grey mongoose model. The response curve for actual evapotranspiration showed a bell-shaped curve (Fig 71a). Probabilities were skewed towards 900 m of elevation (Fig 71b). High probabilities were achieved for degraded landcover type (Fig 71c), they were skewed towards low annual precipitation of the warmest quarter (200 mm, Fig 71d) and high NDVI (June, 0.72, Fig 71f). Predicted probabilities showed an S-shaped curve from low to medium isothermal temperature (Fig 71e). Probabilities dropped abruptly from low topography wetness index and gradually stabilized at 400 or more (Fig 71g). The MaxEnt model generated a map (predicted probability of occurrence; Fig 72) of potential grey mongoose distribution in Mudumalai showing that high probability areas (considered to be areas with > 0.6 probability of presence) accounted for a small portion (38 km^2) towards the south-eastern region in the reserve with the Linear Quadratic Hinge feature type model indicating suitable habitat.

Fig 69. ROC curve of Sensitivity versus Specificity for the habitat model of grey mongoose.

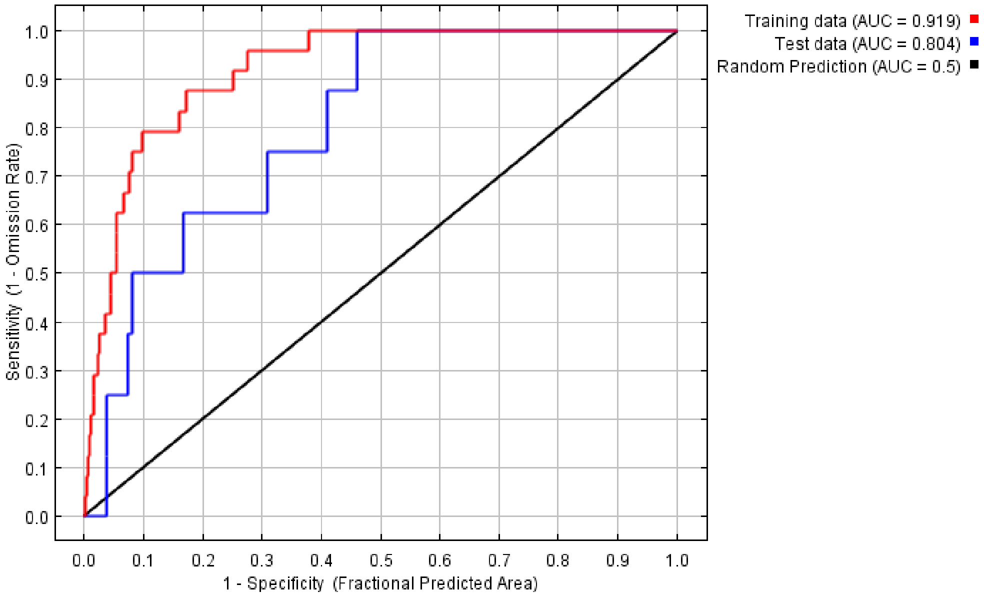


Fig 70. Jackknife analyses of individual predictor variables important in the development of the full model for grey mongoose in relation to the overall model quality or the “regularized training gain.” Dark blue bars indicate the gain achieved when including only that variable and excluding the remaining variables; light blue bars show how much the gain is diminished without the given predictor variable.

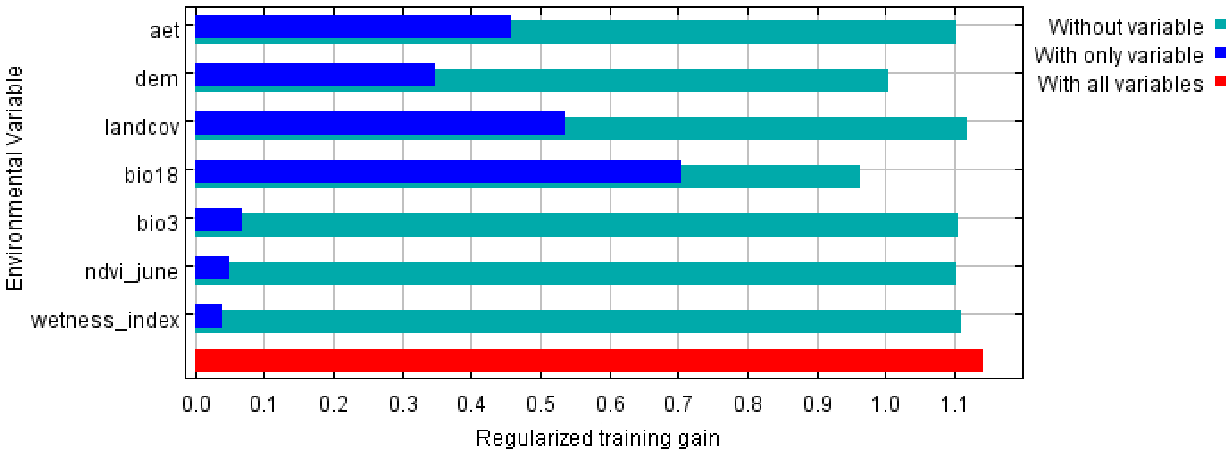
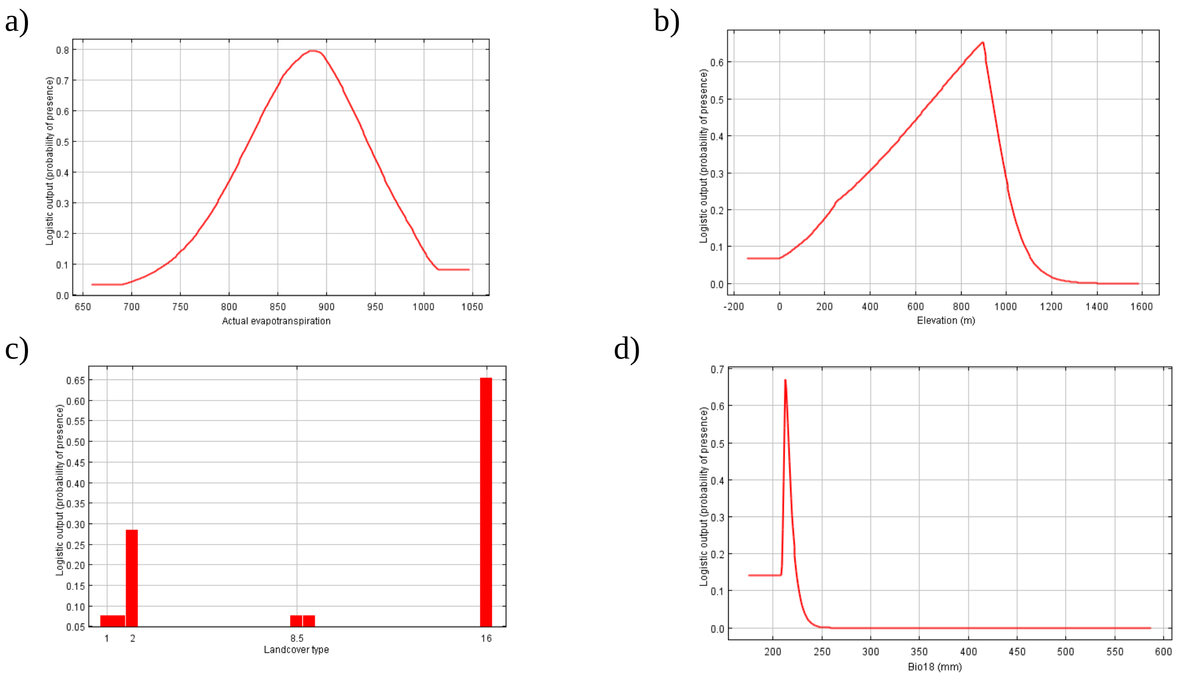


Fig 71. Graphical representation of the relationship between (a) actual evapo-transpiration, (b) elevation, c) landcover type, d) annual precipitation of the warmest quarter e) isothermality f) ndvi (June) g) topography wetness index and grey mongoose probability of presence (2009-2011). Each of the curves represents a different MaxEnt model created using only the corresponding variable.

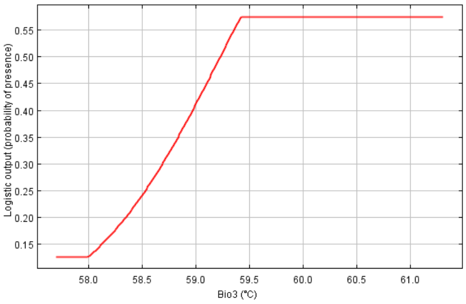

a)

b)

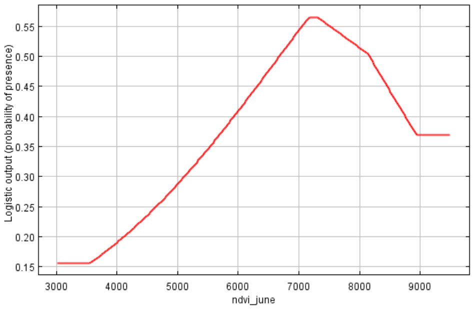
c)

d)

e)


Logistic output (probability of presence)
0.55
0.50
0.45
0.40
0.35
0.30
0.25
0.20
0.15
58.0
58.5
59.0
59.5
60.0
60.5
61.0
Bio3 (°C)

f)


Logistic output (probability of presence)
0.55
0.50
0.45
0.40
0.35
0.30
0.25
0.20
0.15
3000
4000
5000
6000
7000
8000
9000
ndvi_june

g)

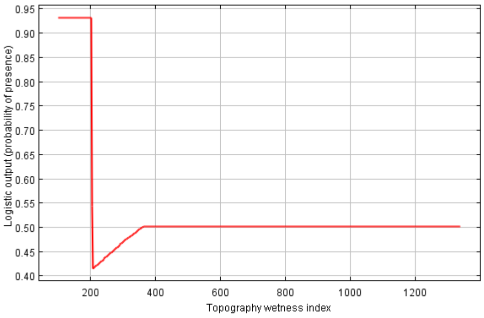

Logistic output (probability of presence)
0.95
0.90
0.85
0.80
0.75
0.70
0.65
0.60
0.55
0.50
0.45
0.40
200
400
600
800
1000
1200
Topography wetness index

Fig 72. Predicted distribution for grey mongoose in Mudumalai Tiger Reserve estimated by MaxEnt modeling (2009-2011). Potential areas are shown in grey shading with the white color indicating higher probabilities of occurrence.

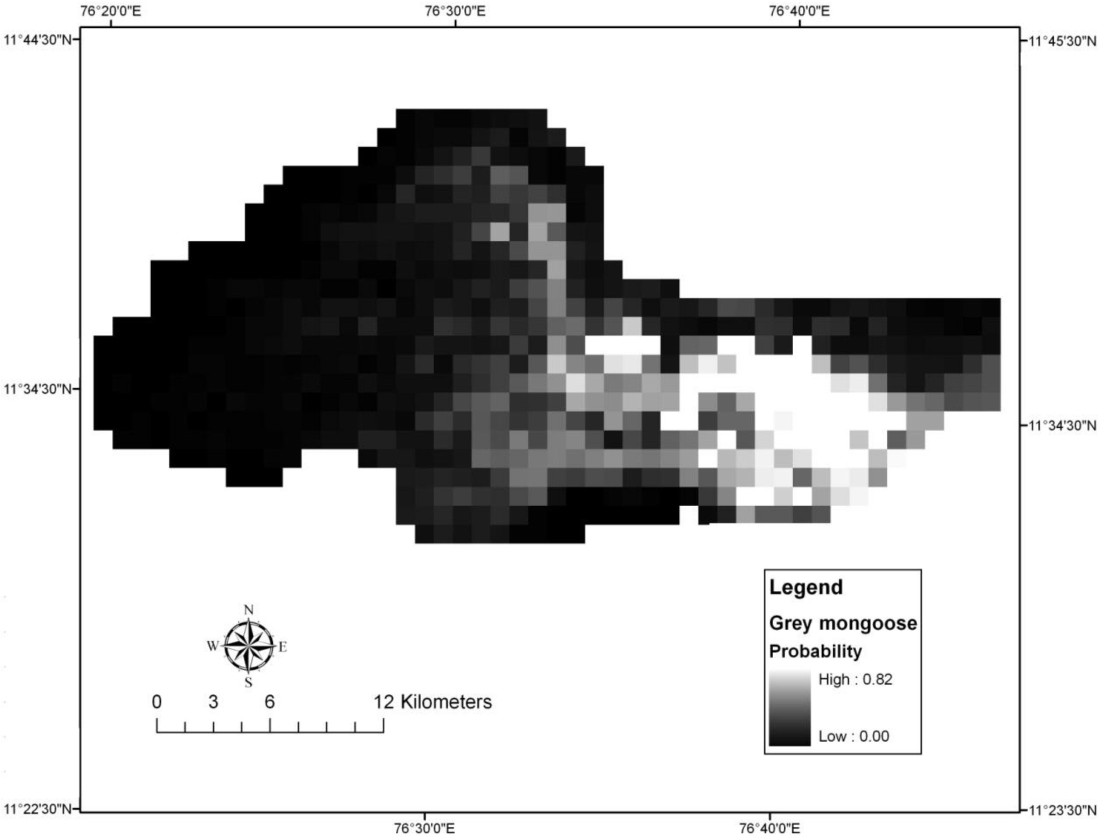


Table 29. Estimates of relative percent contribution (PC) and permutation importance normalized to percentages (PI) for variables used in MaxEnt modeling of small carnivore distribution in Mudumalai Tiger Reserve (2009-2011).

Environmental variables	**MaxEnt model**																	
	Jungle cat		**Leopard cat**		**Rusty-spotted cat**		**Small Indian civet**		**Common palm civet**		**Brown palm civet**		**Stripe-necked mongoose**		**Ruddy mongoose**		**Grey mongoose**	
	PC	**PI**	**PC**	**PI**	**PC**	**PI**	**PC**	**PI**	**PC**	**PI**	**PC**	**PI**	**PC**	**PI**	**PC**	**PI**	**PC**	**PI**
bio18	46	41.2	-	-	19.1	51.6	-	-	39	45.3	39.7	4	2	4.3	12.3	7.2	34.5	75.8
bio19	-	-	-	-	4.5	7.9	9.9	17.6	-	-	9.2	29	13.1	20.1	5.9	26.3	-	-
bio3	-	-	5	20.4	-	-	-	-	-	-	-	-	-	-	-	-	3.4	2.4
dem	19.4	31.3	19.5	7.3	15.2	14.4	31.7	38.7	22.9	20.1	12.7	34.8	41.7	44.3	21.6	30.4	16.6	5.9
ndvi_march	8.3	4.8	-	-	-	-	9.5	10.3	-	-	23	9.3			12	11.3	-	-
forest	7.5	7.3	3.7	8	-	-	15	4.6	8.6	2.1	-	-	4.8	0.5	9.1	5.6	-	-
aspect	5.5	0	-	-	23.7	13.3	18.3	20.1	-	-	4.4	0.4	7.5	16.6	19	14.5	-	-
ndvi_july	5.1	10.7	-	-	-	-	-	-	3.5	10.7	-	-	-	-	-	-	-	-
ndvi_june	-	-	-	-	-	-	-	-	2.4	16.4	1.8	0.3	-	-	-	-	4.1	4.4
d2w	4.2	2.8	-	-	-	-	-	-	-	-	-	-	-	-	2.1	2.7	-	-
landcov	2.4	1	67.8	62.7	30.4	11.1	5.7	0	9.1	1.8	-	-	16.2	3.3	10.8	1	33.3	2.1
wetness_index	1.6	0.9	-	-	7.2	1.7	7.7	5.1	3.8	2.2	-	-	3.6	3.8	1.9	0	2.6	2.2
aet	-	-	4	1.6	-	-	-	-	10.8	1.3	9.2	22.2	4.8	0	5.2	1	5.5	7.1
slope	-	-	-	-	-	-	2.1	3.8	-	-	-	-	6.5	7.2	-	-	-	-

MaxEnt = maximum entropy, bio18 = annual precipitation of the warmest quarter (mm), bio19 = annual precipitation of the coldest quarter (mm), bio3 = isothermality, dem = elevation (m), ndvi_march = Normalized Differentiation Vegetation Index, forest = categorical forest types, aspect = degrees, ndvi_july = , ndvi_June = Normalized Differentiation Vegetation Index, d2w = distance of species presence to nearest water source (m), landcov = categorical land cover type, wetness_index = topography wetness index, aet = actual evapotranspiration, slope = degrees.

6.4. Discussion

Habitat suitability map provided baseline information about the spatial arrangement of potentially suitable habitat for lesser carnivores in Mudumalai. In the spatial model, areas predicted as highly suitable (i.e. probability of occurrence) are clearly delineated, surrounded by areas of lower habitat quality depending upon the species ecological requirements. The modeling results were congruent with our understanding of small carnivore natural history and specifically their habitat preferences. While models developed from coarse-grained landscape variables can predict species distribution effectively, but unfortunately, finer scaled habitat variables, such as food resources, food species, prey distribution, refuge habitat etc., are unlikely to be captured at a landscape level which are essential for small ranging and small sized carnivores that could turn out to be a limitation in this study. Interestingly each of the study species showed a unique distribution pattern; restricted, gradient or random thus indicating the importance of the landscape heterogeneity along with intermediate factors that shape up the distribution of sympatric small carnivore community. Identifying areas of high habitat suitability for lesser carnivores lays the foundation for planning future research and conservation initiatives.

Jungle cat: Its high probability of presence at low precipitation of the warmest quarter and medium elevation, supports the fact that the species prefer open habitats, scrub jungles and agro-ecosystems also supporting its negative relationship with NDVI (March) explaining the species preference towards low canopy areas. The negative relationship of jungle cat with NDVI was also recorded in Sariska Tiger Reserve, North-western India since the area has relatively dry open and scrub forest with sparse vegetation cover (Gupta 2011). High probabilities at sites close to and even away from water sources suggests that this species does not have any specificity towards water. Mudumalai has a high density of large predators (Ramesh 2010) hence it is possible that there could be other variables like capture rates of competing felids (large and small) that could play a major role in its distribution and this must be investigated in detail.

***Rusty spotted cat*:** The negative association of rusty spotted cat with aspect, annual precipitation, of the coldest quarter and wetness index supports that the species preference towards cool and dry areas. Higher probabilities in degraded land explain that the species is tolerant to anthropogenically driven altered habitats. These results support evidence from literature stating that it even occupies abandoned houses in south India and urban cities in the vicinity of forests and amidst agricultural areas (Nowell and Jackson 1996, Mukherjee 1998, Nekaris 2003) although such adaptation could have resulted from the over growing urbanization even in forested areas. Although there are several records of rusty-spotted cats from cultivated and settled areas, it is not known to what degree their populations are able to persist in such areas (Nowell and Jackson 1996) and a large-scale country wise survey would give a better picture.

***Leopard cat*:** The suitability map from this study showed the species suitable habitat to lie mainly in the transition zone from deciduous to evergreen forests. This zone has sites which are relatively moist and dense as depicted from the response curves showing its likelihood towards high elevation, isothermality and evapotranspiration. This species share similar morphological characters with oriental species which may indicate an equal preference for relatively more closed habitats (Mukherjee et al., 2010). A large-scale country wide study based on molecular tools produced suitability maps of various climatic variables with leopard cat locations showing that the maximum temperature in the warmest month (Bio 5) explained leopard cat distribution the most (Mukherjee et al., 2010).

***Small Indian civet*:** This species was likely to be distributed in sites with low and high NDVI and wetness index and medium aspect and elevation with no particular affinity towards forest type. The species seems likely to be found in sites with low precipitation of the coldest quarter and plain terrain. These findings support the fact that across southeast Asia it has been recorded at elevations up to 1500 m, but 88.4% of records were below 600 m with no particular preference for forest type (Jennings and Veron 2011). Small Indian civets occur with a similar frequency in evergreen forest/scrub (47.8%) and deciduous forest/scrub (43.4%) and even in degraded forest (8.7%) in Southeast Asia (Jennings and Veron 2011). Tropical forests in south India are relatively moist and humid than dry semi-arid forests in North-west India which explains why Gupta (2011) reported affinity of the species towards dense canopy cover and water sources.

***Common palm civet*:** This species preferred areas with low precipitation of the warmest quarter and wetness index, medium elevation and evapotranspiration. This species is known to occur in forested areas, urban landscapes and low land areas across the country. Mudappa (2001) did not record the species in rainforests of the Western Ghats thus providing further evidence to our data

***Brown palm civet*:** Brown palm civet preferred areas with high evapotranspiration, aspect, elevation and moderate to high NDVI which characterize rainforests and evergreen forests that are preferred by the species even in its distributional range in the Western Ghats. The key food tree species of the brown palm civet in the reserve as recorded from this study are *Elaeocarpus variablis, Gnetulum ula, Bischovia javanica*, *Piper nigrum* which grow only in the evergreen forests and have also been reported in the species diet from the rainforests of south India (Mudappa et al., 2010).

***Stripe-necked mongoose*:** This species although forest-dwelling, seems to be a generalist due to the availability of sufficient suitable habitat in the study area. Based on the habitat suitability map, the species is likely to be widely distributed across the reserve. The species is likely to be found in areas having low to high evapotranspiration, elevation, aspect and forest type except that it was skewed towards low precipitation and wetness index.

Ruddy mongoose: This species not only has a wide geographical distribution, but it also occurs in varied vegetation types from arid regions in the plains of northern and western India to high altitudes (> 2000 m) of southern India, as well as in human-dominated agricultural landscapes (Choudhury 1997). The response curves for this species is probably typical for mongooses supporting its generalist habits from the fact that predicted probabilities seem to be distributed across from low to high values of other environmental variable except for its likelihood towards low aspect, precipitation and wetness index.

Grey mongoose: The most suitable habitat for this species included areas with medium evapotranspiration, elevation, NDVI and low precipitation of the warmest quarter and wetness index. These results are congruent with available literature where the species is known to thrive in open forests, scrublands, and cultivated fields close to water sources (Choudhury et al., 2011).

The model can be tested by future field surveys accross the Nilgiris Biosphere Reserve for small carnivore presence in (1) areas predicted to have a high probability and (2) areas predicted to have no probability where the model may be wrong. The niche modeling from the present study depicts sympatric spatial distribution pattern between jungle cat, grey mongoose and ruddy mongoose. Although small cats are larger than mongooses, differences in activity pattern might reduce interspecific competition between the two groups. The brown palm civet and common palm civet appear to be spatially separated probably to avoid competition. The common palm civet has never been recorded in the evergreen forests of Western Ghats and hence it can be hypothesized that both species exhibit allopatric distribution in the Western Ghats however co-occurrence analyses would explain this phenomenon in greater detail. Species like the brown palm civet are likely to face extinction as they inhabit a highly specialized habitat. A recent study by Jennings and Veron (2011) revealed the tendency for each civet and mongoose species in Southeast Asia to separate spatially from related species on geographical, habitat, and elevation gradients. Several factors could account for these distribution and niche patterns and explain how these species coexist within Southeast Asia, including interspecific competition, biogeography, and anthropogenic factors. To confirm any interspecific competition, field studies are needed to determine each species' microhabitat usage, behavior and food habits. Further research is needed to determine the ecological sensitivity of small carnivores towards anthropogenic activities.

Overhunting, deforestation, and land conversion have caused range contractions in many mammal species in Southeast Asia, which could account for the absence of some species in fragile ecosystems. Ecological niche modeling highlighted areas with the highest probabilities of occurrence, thereby indicating key localities for long-term conservation of threatened species where further research activities should be prioritized. To confirm my results and further explore the mechanisms responsible for these distribution and niche patterns field studies are needed to gather more data on the distribution, abundance, and ecology of lesser carnivores in India. For instance, insufficient occurrence data exist to investigate spatial or temporal changes in ecological niches, and too little is known about the natural history and ecology of these small cats, civets and mongooses to determine how other biotic factors, such as predation and disease, or the presence of key resources (e.g., den sites, food distribution), also have played a role in determining their distribution patterns.

Literature cited

Abu-Baker, M., Nassar, K., Rifai, L., Qarqaz, M., Al-Melhim, W. and Z. Amr. 2003. On the current status and distribution of the Jungle cat, *Felis chaus*, in Jordan (Mammalia: Carnivora). Zoology in the Middle East 30: 5–10.

Acharjyo, L. N. and S. K. Patnaik. 1987. Occurrence of large Indian civet (*Viverra zibetha*) in Orissa. Journal of the Bombay Natural History Society 84: 201-202.

Allayarov, A. M. 1964. Data on the ecology and geographical distribution of the jungle cat in Uzbekistan. Uzbek Biological Journal: 1-7.

Albignac, R. 1984. The carnivores. In: Key environments: Madagascar. (eds. Jolly, A., Oberle, P. and Albignac, R), Pages 167–182. Pergamon Press, Oxford.

Anderson, D. R. 2008. Model based inference in the life sciences, Springer.

Anderson, D. R., Burnham, K. P., White, G. C. and D. L. Otis. 1983. Density estimation of small mammal populations using a trapping web and distance sampling methods. Ecology 64: 674-680.

Andrewartha, H. G. and L. C. Birch. 1954. The distribution and abundance of animals. The University of Chicago Press, Chicago.

Andrews, P. and E. O'Brien. 2000. Climate, vegetation, and predictable gradients in mammal species richness in southern Africa. Journal of Zoology (London) 251: 205-231.

Anile, S., Bizzarri, L., and B. Ragni. 2009. Camera trapping the European wildcat (*Felis silvestris silvestris*) in Sicily (southern Italy): preliminary results. Hystrix Italian Journal of Mammalogy 20: 55-60.

Anitha, K., Balasubramanian, P. and S. N. Prasad. 2007. Tree community structure and regeneration in Anaikatty hills, Western Ghats. Indian Journal of Forestry 30: 315-324.

Anwar, M., Kumar, H. and J. Vattakavan. 2010. Range extension of rusty-spotted cat to the Indian Terai. Cat News 53: 25–26.

Ashraf, N. V. K., Kumar, A. and A. J. T. Johnsingh. 1993. Two endemic Viverrids of the Western Ghats. Oryx 27: 109-114.

Athreya, V. 2010. Rusty-spotted cat more common than we think? Cat News 53.

Atkinson, R. P. D., Macdonald, D. W. and R. Kamizola. 2002. Dietary opportunism in side-striped jackals *Canis adustus* Sundevall. Journal of Zoology (London) 257: 129–139.

Austin, S. C. and M. E. Tewes. 1999. Observations of viverrids, mustelids and herpestids in Khao Yai NP, Thailand. Small Carnivore Conservation 21: 13–15.

Balakrishnan, M. 2000. Captive breeding of Small Indian Civet (*Viverricula indica*). Department of Forests and Wildlife, Government of Kerala, Thiruvananthapuram, India.

Bali, A., Kumar, A. and J. Krishnaswamy. 2007. The mammalian communities in coffee plantations around a protected area in the Western Ghats, India. Biological Conservation 139: 93-102.

Bartels, E. 1964. On *Paradoxurus hermaphroditus javanicus* (Horsfield, 1824) the common palm civet or toddy cat in western Java: notes on its food and feeding habits, its ecological importance for wood and rural biotopes. Beaufortia 124: 193-201.

Belden, G., Stuebing, R. and M. Nyegang. 2007. Small carnivores in mixed-use forest in Bintulu division, Sarawak, Malaysia. Small Carnivore Conservation 36: 35–37.

Bell, D., Roberton, S. and P. R. Hunter. 2004. Animal origins of SARS coronavirus: possible links with the international trade in small carnivores. Philosophical Transactions of the Royal Society of London, Series B, Biological Sciences 359: 1107-1114.

Bezuijen, M. R. 2003. The flat headed cat in the Merang river region of south Sumatra. Cat News 38: 26–27.

Bisceglia, S. B. C., Pereira, J. A., Teta, P. and R. D. Quintana. 2008. Food habits of Geoffroy's cat (*Leopardus geoffroyi*) in the central Monte desert of Argentina. Journal of Arid Environments 72: 1120-1126.

Bisceglia, S. B. C., Pereira, J. A., Teta, P. and R. D. Quintana. 2011. Rodent selection by Geoffroy's cats in a semi-arid scrubland of central Argentina. Journal of Arid Environments 75: 1024–1028.

Blanford, W. T. 1888–91. The fauna of British India, including Ceylon and Burma. Volume I, Mammalia. Taylor and Francis, London.

Blaum, N., Rossmanith, E., Fleissner, G. and F. Jeltsch. 2007. The conflicting importance of shrubby landscape structures for the reproductive success of the yellow mongoose (*Cynictis penicillata*). Journal of Mammalogy 88: 194–200.

Borchers, D. L. and M. G. Efford. 2007. Spatially Explicit Maximum Likelihood Methods for Capture–Recapture Studies. Biometrics 64: 377–385.

Boulanger, J. and B. N. McLellan. 2001. Closure violation bias in DNA based mark–recapture population estimates of grizzly bears. Canadian Journal of Zoology 79: 642-651.

Buckland, S. T., Anderson, D. R., Burnham, K. P., Laake, J. L. Borchers, D. L. and L. Thomas. 2001. Introduction to Distance Sampling: Estimating Abundance of Biological Populations. Oxford University Press, Oxford.

Burneo, S. F., González-Maya, J. F. and D. G. Tirira. 2009. Distribution and habitat modelling for Colombian Weasel *Mustela felipei* in the Northern Andes. Small Carnivore Conservation 41: 41-45.

Burnham, K. P., Anderson, D. R. and J. L. Laake. 1980. Estimation of density from line transect sampling of biological populations. Wildlife Monograph 72: 1-202.

Burnham, K. P. and D. R. Anderson. 1998. Model Selection and Inference. Springer, New York.

Buskirk, S. W. and R. A. Powell. 1994. Habitat ecology of fishers and American martens. In: Martens, sables, and fishers: biology and conservation (eds. Buskirk, S. W., Harestad, A. S., Raphael, M. G. and R. A. Powell), Pages 283-296. Ithaca, NY: Cornell University Press.

Buskirk, S. W. 1999. Mesocarnivores of Yellowstone. In: Carnivores in ecosystems: The Yellowstone experience. (eds. Clark, T. W., Curlee, A. P., Minta, S. C. and P. M. Kareiva), Pages 167-187. Yale University Press, New Haven, CT.

Buskirk, S. W. 2003. Small and mid-sized carnivores. In: Mammalian community dynamics: Management and conservation in the coniferous forests of western North America. (eds. Zabel, C. J. and R. G. Anthony), Pages 207-249 Cambridge University Press, Cambridge.

Canepuccia, A. D., Martinez, M. M. and A. I. Vassallo. 2007. Selection of waterbirds by Geoffroy's cat: Effects of prey abundance, size, and distance. Mammalian Biology 72: 163-173.

Carbone, C., Christie, S., Conforti, K., Coulson, T., Franklin, N., Ginsberg, J. R., Griffiths, M., Holden, J., Kawanishi, K., Kinnaird, M., Laidlaw, R., Lynam, A., Macdonald, D. W., Martyr, D., McDougal, C., Nath, L., O'Brien, T., Seidensticker, J., Smith, D. J. L., Sunquist, M., Tilson, R. and W. N. Wan Shahruddin. 2001. The use of photographic rates to estimate densities of tigers and other cryptic mammals. Animal Conservation 4: 75–79.

Carbone, C., Cowlishaw G., Isaac, N. J. B. and J. M. Rowcliffe. 2005. How far do animals go? Determinants of day range in mammals. American Naturalist 165: 290-297.

Carbone, C. and J. Gittleman. 2002. A common rule for scaling. Science 295: 2273-2276.

Caro, T. M. and C. J. Stoner. 2003. The potential for interspecific competition among African carnivores. Biological Conservation 110: 67–75.

Cavallini, P. 1993. Spatial organization of the yellow mongoose *Cynictis penicillata* in a coastal area. Ethology, Ecology and Evolution 5: 501-509.

Cavallini, P. and J. A. J. Nel. 1990. Ranging behaviour of the Cape grey mongoose *Galerella pulverulenta* in a coastal area. Journal of Zoology (London) 222: 353–362.

Cavallini, P and J. A. J. Nel. 1995. Comparative behavior and ecology of two sympatric mongoose species (*Cynictis penicillata* and *Galerella pulverulenta*). South African Journal of Zoology 30: 46-49.

Chakraborty, S. 1978. The rusty-spotted cat, *Felis rubiginosa* I. Geoffroy, in Jammu and Kashmir. Journal of Bombay Natural History Society 75: 478-479.

Champion, H. G. and S. K. Seth. 1968. A Revised Survey of the Forest Types of India. The Manager of Publications, Government of India, New Delhi.

Chandrasekhar–Rao, A. and M. E. Sunquist. 1996. Ecology of small mammals in tropical forest habitats of southern India. Journal of Tropical Ecology 12: 561–571.

Choudhury, A. U. 1997. The distribution and status of small carnivores (Mustelids, Viverrids, and Herpestids) in Assam, India. Small Carnivore Conservation 16: 25-26.

Choudhury, A. U. 1999. Conservation of small carnivores (Mustelids, Viverrids, Herpestids and one Ailurid) in north Bengal, India. Small Carnivore Conservation 20: 15-17.

Choudhury, A., Wozencraft, C., Muddapa, D., Yonzon, P., Jennings, A. and V. Geraldine. 2011. *Herpestes edwardsii*. In: IUCN 2012. IUCN Red List of Threatened Species. Version 2012.2. <www.iucnredlist.org>. Downloaded on 19 November 2012.

Cody, M. L. 1974. Competition and the structure of bird communities. Princeton University Press, Princeton, New Jersey, USA.

Colon, C. P. 2002. Ranging behaviour and activity of the Malay civet (*Viverra tangalunga*) in a logged and an unlogged forest in Danum Valley, East Malaysia. Journal of Zoology (London) 257: 473–485.

Colwell, R. K. 2006. EstimateS: Statistical estimation of species richness and shared species from samples. Version 8.0.0. Online at http://purl.oclc.org/estimates.

Condit, R., Ashton, P. S., Baker, P., Bunyavejchewin, S., Gunatilleke, S., Gunatilleke N., Hubbell, S. P., Foster, R. B., Itoh, A., Lafrankie, J. V., Lee, H. S., Losos, E., Manokaran, N., Sukumar, R. and T. Yamakura. 2000. Spatial patterns in the distribution of tropical tree species. Science 288: 1414-1418.

Cooch, E. and G. C. White. 2006. Program MARK: a gentle introduction. 5th Edition. Cornell University, Ithaca. (http://www.phidot.org/software/mark/docs/book/).

Cook, R. D. and J. O. Jacobson. 1979. A design for estimating visibility bias in aerial surveys. Biometrics 35: 735–742.

Corbet, G. B. and J. E. Hill. 1992. Mammals of the Indo-Malayan Region: A Systematic Review. Oxford University Press, Oxford, UK.

Corbett, l. k. 1979. Feeding ecology and social organization of wildcats (*Felis silvestris*) and domestic cats (*Felis catus*) in Scotland, PhD thesis, Department of Zoology, University of Aberdeen.

Corn, J. L. and M. J. Conroy. 1998. Estimation of density of mongooses with capture-recapture and distance sampling. Journal of Mammalogy 79: 1009-1015.

Cossıos, D., Lucherini, M., Ruiz-garcia, M. and B. Angers. 2009. Influence of ancient glacial periods on the Andean fauna: the case of the pampas cat (*Leopardus colocolo*). BMC Evolutionary Biology 9: 68–80.

Creel, S. and M. Hoffmann. 2008. *Helogale parvula*. In IUCN Red List of Threatened Species (Version 2009.2). Retrieved from www.iucnredlist.org.

Creel, S., Spong, G. and N. Creel. 2001. Interspecific competition and the population biology of extinction-prone carnivores. In: Carnivore Conservation, (eds. J. L. Gittleman, Funk, S. M., Macdonald, D. and R. K. Wayne), Cambridge University Press, UK.

Cuéllar, E., Maffei, L., Arispe, R. and A. Noss. 2006. Geoffroy's cats at the northern limit of their range: observations from camera trapping in Bolivian dry forests. Studies on Neotropical Fauna and Environment 41: 169-177.

Dal, S. K. 1954. Animal World of the Armenian Soviet Socialist Republic. Vol. 1: Vertebrate animals.– Academy of Sciences of the Armenian SSR, Yerevan.

de Alwis, W. L. E. 1973. Status of Southeast Asia's small cats. In: The world's cats, Vol. 1: Ecology and conservation. (ed. R. L. Eaton), Pages 198-208. World Wildlife Safari, Portland, Oregon.

de Oliveira, T., Eizirik, E., Schipper, J., Valderrama, C., Leite-Pitman, R. and E. Payan. 2008. *Leopardus tigrinus*. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.2. <www.iucnredlist.org>. Downloaded on 17 March 2012.

Deraniyagala, P. E. P. 1956. A new subspecies of rusty spotted cat from Ceylon. Spoila Zeylanica 28: 113.

Dhungel, S. 1985. Ecology of the Hog Deer in Royal Chitwan National Park, Nepal. Ph.D. Dissertation, University of Montana, USA.

Diaz Behrens, G. and H. Van Rompaey. 2002. The Ethiopian Genet, *Genetta abyssinica* (Rupell 1836) (Carnivora, Viverridae): Ecology and phenotypic aspects. Small Carnivore Conservation 27: 23-28.

Dice, L. R. 1938. Some Census Methods for Mammals. Journal of Wildlife Management 2: 119-130.

Diamond, J. M. 1978. Niche shifts and the rediscovery of interspecific competition. American Scientist 66: 322–331.

Dorcas, M. E. and J. D. Willson. 2009. Innovative methods for studies of snake ecology and conservation. In: Snakes: Ecology and Conservation. (eds. Mullin, S. J. and R. A. Seigel), Pages. 5–37. Cornell University Press, Ithaca, NY,

Dubey, Y. 1999. Sighting of rusty spotted cat in Tadoba Andhari Tiger Reserve, Maharashtra. Journal of the Bombay Natural History Society 96: 310.

Duckworth, J. W. 1997. Small carnivores in Laos: a status review with notes on ecology, behaviour and conservation. Small Carnivore Conservation 16: 1–21.

Duckworth, J. W., Salter, R. E. and K. Khounbline. 1999. Wildlife in Lao PDR: 1999 Status Report. IUCN, Vientiane, Laos.

Duckworth, J. W., Poole, C. M., Tizard, R. J., Walston, J. L. and R. J. Timmins. 2005. The Jungle Cat *Felis chaus* in Indochina: a threatened population of a widespread and adaptable species. Biodiversity and Conservation 14: 1263-1280.

Duckworth, J. W. and W. G. Robichaud. 2005. Yellow-bellied Weasel *Mustela kathiah* sightings in Phongsaly province, Laos, with notes on the species' range in South-East Asia, and recent records of other small carnivores in the province. Small Carnivore Conservation 33: 17–20.

Duckworth, J. W., Steinmetz, R., Sanderson, J. and S. Mukherjee. 2008a. *Felis chaus*. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.1. <www.iucnredlist.org>. Downloaded on 09 November 2011.

Duckworth, J. W., Timmins, R. J. and D. Muddapa. 2008b. *Viverricula indica*. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.1. <www.iucnredlist.org>. Downloaded on 09 November 2011.

Dueser, R. D. and J. H. Shugart. 1978. Microhabitats in a forest-floor small mammal fauna. Ecology 59: 89-98.

Dunham, A. E. 1998. Notes on the behavior of the Ring-tailed mongoose, *Galidia elegans*, at Ranomafana National Park, Madagascar. Small Carnivore Conservation 19: 21-24.

Dunstone, N., Durbin, L., Wyllie, I., Freer, R., Jamett, G. A., Mazzolli, M. and S. Rose. 2002. Spatial organization, ranging behaviour and habitat use of the kodkod (*Oncifelis guigna*) in southern Chile. Journal of Zoology (London) 257: 1-11.

Datta, A., Naniwadekar, R. and M. O. Anand. 2008. Occurrence and conservation status of small carnivores in two protected areas in Arunachal Pradesh, north-east India. Small Carnivore Conservation 39: 1–10.

Dragesco-Joffe, A. 1993. La vie sauvage au Sahara, Delechaux et Niestle, Lausanne.

Eco News. 2011. Highly endangered cat species spotted in Corbett, Zee News Limited, 12 August 2011.

Efford, M. G., Dawson, D. K. and C. S. Robbins. 2004. DENSITY: software for analysing capture–recapture data from passive detector arrays. Animal Biodiversity and Conservation 27: 217–228.

Efford, M. G. 2009. DENSITY 4.45: software for spatially explicit capture–recapture. Department of Zoology, University of Otago, Dunedin, New Zealand. http://www.otago.ac.nz/density. Accessed on 7 June 2011.

Eisenberg, J. E. 1981. The mammalian radiations: an analysis of trends in evolution, adaptation, and behavior. The University of Chicago Press, Chicago, 610 pp.

Ellerman, J. R. and M. T. C. S. Scott. 1951. Checklist of Palaeartic and Indian mammals. British Museum, London.

Elith, J., Graham, C. H., Anderson, R. P., Dudik, M., Ferrier, S., Guisan, A., Hijmans, R. J., Huettmann, F., Leathwick, J. R., Lehmann, A., Li, J., Lohmann, L. G., Loiselle, B. A., Manion, G., Moritz, C., Nakamura, M., Nakazawa, Y., Overton, J. M., Peterson, A.T., Phillips, S.J., Richardson, K., Scachetti-Pereira, R., Schapire, R.E., Soberon, J., Williams, S., Wisz, M. S. and N. E. Zimmermann. 2006. Novel methods improve prediction of species' distributions from occurrence data. Ecography 29: 129–151.

ESRI. 2009. ArcMap v.9.3.1. Copyright 1999-2009.

Feldhamer, G. A., Drickamer, L. C., Vessey, S. H. and J. F. Merritt. 1999. Mammalogy. Adaptation, Diversity and Ecology. WCB McGraw-Hill, Boston. xii+563pp.

Ferrari, N. and J. M. Weber. 1995. Influence of the abundance of food resources on the feeding habits of the red fox *Vulpes vulpes*, in western Switzerland. Journal of Zoology (London) 236: 117–129.

Fischer, J. and B. D. Lindenmayer. 2006. Beyond fragmentation: the continuum model for fauna research and conservation in human-modified landscapes. Oikos 112: 473–480.

García, D. and R. Ortiz-Pulido. 2004: Patterns of resource tracking by avian frugivores at multiple spatial scales: two case studies on discordance among scales. Ecography 27: 187-196.

Gardner, B., J. Reppucci, M. Lucherini, and J. A. Royle. 2010. Spatially-explicit inference for open populations: estimating demographic parameters from camera-trap studies. Ecology 91: 3376–3383.

Gaston, K. J., Blackburn, T. M. and R. D. Gregory. 1999. Does variation in census area confound density comparisons? Journal of Applied Ecology 36: 191–204.

Gaubert, P., Taylor, P. J. and G. Veron. 2005a. Integrative taxonomy and phylogenetic systematics of the genets (Carnivora, Viverridae, genus *Genetta*): a new classification of the most speciose carnivoran genus in Africa. In: African Biodiversity: Molecules, Organisms, Ecosystems. (eds. B.A. Huber, Sinclair, B.J. and K.-H. Lampe) Pages. 371–383. Springer, New York.

Gaubert, P., Wozencraft, W. C., Cordeiro-Estrela, P. and G. Veron. 2005b. Mosaics of convergences and noise in morphological phylogenies: What's in a viverrid-like carnivoran? Systematic Biology 54: 865–894.

Genovesi, P., Besa, M. and S. Toso. 1995. Ecology of a feral cat *Felis catus* population in an agricultural area of northern Italy. Wildlife Biology 1: 233-237.

Gerber, B., Karpanty, S. M., Crawford, C., Kotschwar, M. and J. Randrianantenaina. 2010. An assessment of carnivore relative abundance and density in the eastern rainforests of Madagascar using remotely-triggered camera traps. Oryx 44: 219-222.

Gibbs, J. P. 2000. Monitoring populations. In: Research techniques in animal ecology. (eds. L. Boitani and T. K. Fuller), Pages 213–252. Columbia University Press, New York, New York, USA.

Gittleman, J. L. and P. H. Harvey. 1982. Carnivore home-range size, metabolic needs and ecology. Behavioural Ecology Sociobiology 10: 57-63.

Glas, L. *In press*. *Felis chaus*. In: The Mammals of Africa. (eds. J. S. Kingdon and M. Hoffmann), Academic Press, Amsterdam, The Netherlands.

Goodman, S. M., Kerridge, F. J. and R. C. Ralisoamalala. 2003. A note on the diet of *Fossa fossana* (Carnivora) in the central eastern humid forests of Madagascar. Mammalia 67: 595-597.

Gorman, M. L. 1975. The diet of feral *Herpestes auropunctatus* in the Fijian Island. Journal of Zoology (London) 175: 273-278.

Gorman, M. L. 1979. Dispersion and foraging of the small Indian mongoose *Herpestes auropunctatus*, relative to the evolution of social viverrids. Journal of Zoology (London) 187: 65-74.

Gray, T. N. E. and C. Phan. 2011. Habitat preferences and activity patterns of the larger mammal community in Phnom Prich Wildlife Sanctuary, Cambodia. Raffles Bulletin of Zoology 59: 311-318.

Grassman, L. I. Jr. 1998. Movements and fruit selection of two Paradoxurinae species in a dry evergreen forest in southern Thailand. Small Carnivore Conservation 19: 25-29.

Grassman, L. I. Jr. 2000. Movements and prey selection of the leopard cat (*Prionailurus bengalensis*) in a dry evergreen forest in Thailand. Acta Theriologica 45: 421–426.

Grassman, L. I. Jr., Tewes, M. E., Silvy, N. J. and K. Kreetiyutanont. 2005. Spatial organization and diet of the leopard cat (*Prionailurus bengalensis*) in north-central Thailand. Journal of Zoology (London) 266: 45-54.

Green, R. 1991. Wild cat species of the world. Basset, Plymouth.

Grinnell, J. 1924. Geography and evolution. Ecology 5: 225–229.

Guan, Y., Zheng, B. J., He, Y. Q., Liu, X. L., Zhuang, Z. X., Cheung, C. L., Luo, S. W., Li, P. H., Zhang, L. J., Guan, Y. J., Butt, K. M., Wong, K. L., Chan, K. W., Lim, W., Shortridge, K. F., Yuen, K. Y., Peiris, J. S. M. and L. L. M. Poon. Isolation and characterization of viruses related to the SARS coronavirus from animals in southern China. Science 302: 276–278.

Guisan, A. and N. E. Zimmermann. 2000. Predictive habitat distribution models in ecology. Ecological Modelling 135: 147–186.

Gupta, B. K. 1997. Brown palm civet *Paradoxurus jerdoni*, in Periyar Tiger Reserve, Western Ghats, India. Small Carnivore Conservation 16: 30.

Gupta, B. K. 2004. Killing civets for meat and scent in India. Small Carnivore Conservation 31: 21.

Gupta, S. 2011. Ecology of medium and small sized carnivores in Sariska Tiger Reserve, Rajasthan, India. PhD thesis, Saurashtra University, 156 pp.

Habibi, K. 2003. Mammals of Afghanistan. Zoo Outreach Organisation with assistance from U.S. Fish and Wildlife Service, Coimbatore, India.

Haines, A. M., Grassman, Jr., L. I. and M. E. Tewes. 2004. Survival of radiocollared adult leopard cats *Prionailurus bengalensis* in Thailand. Acta Theriologica 49: 349-356.

Hanfee, F. and A. Ahmed. 1999. Some observations on India's illegal trade in mustelids, viverrids and herpestids. In: ENVIS Bulletin: Wildlife and protected areas. Mustelids, viverrids and herpestids of India, (ed. S.A. Hussain), 2: 113-115.

Haspel, C. and R. E. Calhoon. 1989. Home ranges of free-ranging cats (*Felis catus*) in Brooklyn, New York. Canadian Journal of Zoology 67: 178-181.

Haque, N. M. and V. Vijayan. 1993. Food habits of the fishing cat *Felis viverrina* in Keoladeo National Park, Bharatpur, Rajasthan. Journal of the Bombay Natural History Society 90: 498-500.

Hegde, R. and T. Enters. 2000. Forest products and household economy: a case study from Mudumalai Wildlife Sanctuary, Southern India. Environmental Conservation 27: 250-259.

Heilbrun, R. D., Silvy, N. J., Peterson, M. J. and M. E. Tewes. 2006. Estimating bobcat abundance using automatically triggered cameras. Wildlife Society Bulletin 34: 69–73.

Henschel, P., Abernethy, K. A. and L. J. T. White. 2005. Leopard food habits in the Lope National Park, Gabon, Central Africa. African Journal of Ecology 43: 21-28.

Heptner, V. H. and A. A. Sludskii 1972. Mammals of the Soviet Union. Vol III: Carnivores (Feloidea). Vyssha Shkola, Moscow.

Heptner, V. G., Sludskii, A. A., Komarov, A. and N. Komorov. 1992. Mammals of the Soviet Union: Carnivora, Part 2. BRILL. ISBN 9789004088764. 358 pp.

Herbst, M. and M. G. L. Mills. 2010. The feeding habits of the southern African wildcat (*Felis silvestris cafra*), a facultative trophic specialist, in the southern Kalahari (Kgalagadi Transfrontier Park, South Africa/Botswana). Journal of Zoology (London) 280: 403–413.

Herrera, C. M. 1989. Frugivory and seed dispersal by carnivorous mammals, and associated fruit characteristics, in undisturbed Mediterranean habitats. Oikos 55: 250-262.

Heydon, M. J. and P. Bulloh. 1996. The impact of selective logging upon sympatric civet species (Viverridae) in Borneo. Oryx 30: 31-36.

Hijmans, R. J., Cameron, S. E., Parra, J. L., Jones, P. G. and A. Jarvis. 2005. Very high resolution interpolated climate surfaces for global land areas. International Journal of Climatology 25: 1965-1978.

Hinde, G. and L. Hunter. 2005. Cats of Africa: behaviour, ecology, and conservation. Struick. ISBN 978-1-77007-063-9.

Hiscocks, K. and M. R. Perrin. 1991. Den selection and use by dwarf mongooses and banded mongooses in South Africa. South African Journal of Wildlife Research 21: 119-122.

Hoagland, D. B., G. R. Horst, and C. W. Kilpatrick. 1989. Biogeography and population biology of the mongoose in the West Indies. In: Biogeography of the West Indies. (ed. C. A. Woods), Pages 611–634.. Sand Hill Crane Press, Gainesville, Florida, USA.

Holden, J. 2001. Small cats in Kerinci Seblat National Park, Sumatra, Indonesia. Cat News 35: 11-14.

Honacki, J. H., Kinman, K. E. and J. W. Koeppl. 1982. Mammal's species of the world. Allen Press Inc. and The Association of Systematic Collections. Lawrence Kanas, USA.

Horst, G. R., Hoagland, D. B. and C. W. Kilpatrick. 2001 The mongoose in the West Indies: the biogeography and population biology of an introduced species. In: Biogeography of the West Indies: patterns and perspectives, vol. 2 (eds. C. A. Woods and F. E. Sergile), Pages. 409–424. Boca Raton, FL: CRC Press.

Howard, J. G. and D. E. Wildt. 1990. Ejaculate and hormonal characteristics in the leopard cat (*Felis bengalensis*), and sperm function as measured by in vitro penetration of zona-free hamster ova and zona-intact domestic cat oocytes Molecular Reproduction and Development 26: 163—174.

Hume, I. D. 2005. Concepts of digestive efficiency. In: Physiological and Ecological Adaptations to Feeding in Vertebrates. (eds. Stark, J. M. and T. Wang), Pages. 43–58. Science Publishers Inc, New Heaven, USA.

Hurlbert, S. H. 1978. The measurement of niche overlap and some relatives. Ecology 59: 67–77.

Hutchinson, G. E. 1957. Concluding remarks. Cold Spring Harb. Symposia on Quantitative Biology 22: 415–427.

Hutton, A. F. 1949. Notes on the snakes and mammals of the High Wavy Mountains, Madura district, south India. Journal of Bombay Natural History Society 48: 681-694.

IUCN. 1996. Wild cats, status survey and conservation action plan. Gland: IUCN.

IUCN. 2007. European Mammal Assessment. IUCN, Gland, Switzerland and Cambridge, UK.

IUCN. 2008. IUCN Red List of Threatened Species. Available at: http://www.iucnredlist.org. (Accessed on 5 October 2008).

Izawa, M., Doi, T., Okamura, M., Nakanishi, N., Murayama, A., Hiyama, T., Oh, D., Teranishi, A. and A. Suzuki. 2007. Toward the survival of two endangered felid species of Japan. In: Felid biology and conservation conference. (eds. J. Hughes and R. Mercer), 17-20 September: Abstracts, Pages. 120-121. WildCRU, Oxford, UK.

Jacobs, J. 1974. Quantitative measurement of food selection– a modification of the forage ratio and Ivlev's electivity index. Oecologia 14: 413–417.

Jayaram, K. C. 1999. The Freshwater Fishes of The Indian Region. Narendra Publishing house, New Delhi, 396 pp.

Jennelle, C. S., Runge, M. C. and D. I. MacKenzie. 2002. The use of photographic rates to estimate densities of tigers and other cryptic mammals: A comment on misleading conclusions. Animal Conservation 5: 119–120.

Jennings, A. P. and G. Veron. 2011. Predicted distributions and ecological niches of 8 civet and mongoose species in Southeast Asia. Journal of Mammalogy 92: 316-327.

Jerdon, T. C. 1874. A Handbook of the Mammals of India (Reprint 1984 by Mittal Publications, Delhi).

Jhala, Y. V., Gopal, R. and Q. Qureshi. (eds) 2008. Status of tigers, co-predators, and prey in India. National Tiger Conservation Authority and Wildlife Institute of India, Dehra Dun, TR08/001:1–164.

Jordan, M. J., Barrett, R. H. and K. L. Purcell. 2011. Camera trapping estimates of density and survival of fishers *Martes pennant*. Wildlife Biology 17: 266-276.

Joseph, S., Reddy, C. S., Pattanaik, C. and S. Sudhakar. 2008. Distribution of plant communities along climatic and topographic gradients in Mudumalai Wildlife Sanctuary (southern India). Biological Letters 45: 29-41. Available online at http://www.biollett.amu.edu.pl

Joshi, A. R., Smith, J. L. D. and F. J. Cuthbert. 1995. Influence of Food Distribution and Predation Pressure on Spacing Behavior in Palm Civets. Journal of Mammalogy 76: 1205-1212.

Joshi, N. V., Suresh H. S., Dattaraja H. S. and R. Sukumar. 1997. The spatial organization of plant communities in a deciduous forest: A computational-geometry-based analysis. Journal of the Indian Institute of Science 77: 365-375.

Johnsingh, A. J. T. 1983. Large mammalian prey-predators in Bandipur. Journal of the Bombay Natural History Society 80: 1-57.

Johnsingh, A. J. T. 1986. Diversity and conservation of carnivorous mammals in India. Proceedings of the Indian Academy of Science, Bangalore, Karnataka, India.

Johnson, W. E. and W. L. Franklin. 1991. Feeding and spatial ecology of *Felis geoffroyi* in southern Patagonia. Journal of Mammalogy 72: 815-820.

Johnson, W. E, Eizirik, E., Pecon-Slattery, J., Murphy, W.J., Antunes, A., Teeling E. and S. J. O'Brien. 2006. The Late Miocene Radiation of Modern Felidae: A Genetic Assessment. Science 311: 73–77.

Johnson, A, Vongkhamheng, C. and T. Saithongdam. 2008. The diversity, status and conservation of small carnivores in a montane tropical forest in northern Laos. Oryx 43: 626-633.

Kalle, R. 2011. Wildlife, a treasure or a delicacy? Hornbill 32-33.

Kalle, R., Ramesh, T., Qureshi, Q. and K. Sankar. 2011. Density of tiger and leopard in a tropical deciduous forest of Mudumalai Tiger Reserve, southern India, as estimated using photographic capture–recapture sampling. Acta Theriologica 56: 335–342.

Kanchanasakha, B., Simcharoen, S. and U. T. Than. 1998. Carnivores of Mainland South East Asia. Bangkok: Siam Tong Kit Printing Co. Ltd.

Karanth, U. 1986. A possible sighting record of the Malabar civet (*Viverra megaspila* Blyth) from Karnataka. Journal of the Bombay Natural History Society 83: 192-193.

Karanth, K. U. 1988. Analysis of predator-prey balance in Bandipur Tiger Reserve with reference to census reports. Journal of the Bombay Natural History Society 85: 1-8.

Karanth, K. U. 1993. Predatory-prey relationships among large mammals of Nagarhole National Park (India). PhD thesis. Mangalore University, India.

Karanth, K. U. 1995. Estimating tiger *Panthera tigris* populations from camera trap data using capture-recapture models. Biological Conservation 71: 333–338.

Karanth, K. U. and J. D. Nichols. 1998. Estimating tiger (*Panthera tigris*) populations from camera-trap data using capture–recaptures. Ecology 79: 2852–2862.

Karanth, K. U. and J. D. Nichols. 2002. Monitoring Tigers and their Prey. A Manual for Researchers, Managers and Conservationists in Tropical Asia. Center for Wildlife Studies, Bangalore, India.

Kerridge, F. J., Ralisoamalala, R. C., Goodman, S. M. and S. D. Pasnick. 2003. *Fossa fossana*, Malagasy striped civet, fanaloka. In: Natural history of Madagascar. (eds. Goodman, S. M. and J. P. Benstead), Pages. 1363–1365. The University of Chicago Press, Chicago.

Khan, A. A. and M. A. Beg. 1986. Food of some mammalian predators in the cultivated areas of Punjab. Pakistan Journal of Zoology 18: 71-79.

Khan, J. and S. Mukherjee. 2008. *Prionailurus rubiginosus*. IUCN Red List of Threatened Species. Version 2010.4. International Union for Conservation of Nature.

Kiltie, R. A. 1988. Intraspecific size regularities in tropic felid assemblages. Oecologia 76: 97-105.

Kjellander, P. and J. Nordstro¨ m. 2003. Cyclic voles, prey switching in red fox, and roe deer dynamics—a test of the alternative prey hypothesis. Oikos 101: 338–344.

Kittle, A. and A. Watson. 2004. Rusty-spotted cat in Sri Lanka: observations of an arid zone population. Cat News 40: 17–19.

Kodandapani, N., Cochrane, M. A. and R. Sukumar. 2004. Conservation threat of increasing fire frequencies in the Western Ghats, India. Conservation Biology 18: 1553-1561.

Kotliar, N. B. and J. A. Wiens. 1990. Multiple scales of patchiness and patch structure: a hierarchical framework for the study of heterogeneity. Oikos 59: 253–260.

Krebs, C. J. 1989. Niche overlaps and diet analysis. In: Ecological Methodology, (ed. C. Krebs), p. 654. Harper & Row, New York.

Krishnakumar, H. and M. Balakrishnan. 2003. Feeding ecology of the Common Palm Civet *Paradoxurus hermaphroditus* (Pallas) in semi-urban habitats of Trivandrum, India. Small Carnivore Conservation 28: 10-11.

Krishnan, M. 1972. An ecological survey of the larger mammals of Peninsular India. Journal of the Bombay Natural History Society 69: 26-54.

Kumar, A., Chellam, R., Choudhury, B. C., Mudappa, D., Vasudevan, K., Ishwar, N. M. and B. R. Noon. 2002. Impact of rainforest fragmentation on small mammals and herpetofauna in the Western Ghats, south India. WII-USFWS Collaborative Project Final Report, Wildlife Institute of India, Dehra Dun. 146 pp.

Kumar, A. and G. Umapathy. 1999. Home range and habitat use by Indian grey mongoose and small Indian civets in Nilgiri Biosphere Reserve, India. In: ENVIS Bulletin: Wildlife and protected areas. Mustelids, viverrids and herpestids of India, (ed. S. A. Hussain), 2: 87-91.

Kumar, A. and K. Yoganand. 1999. Distribution and abundance of small carnivores in Nilgiri Biosphere Reserve, India. In: ENVIS Bulletin: Wildlife and protected areas, mustelids, viverrids and herpestids of India, (ed. S. A. Hussain), 2: 74-86.

Kumara, H. N. and M. Singh. 2007. Small carnivores of Karnataka: distribution and sight records. Journal of the Bombay Natural History Society 104: 155-162.

Kumara, H. N., Sharma, A. K., Kumar, M. A. and M. Singh. 2000. Road kills of wild fauna in Indira Gandhi Wildlife Sanctuary, Western Ghats, India: Implications for management. Biosphere Conservation 3: 41-47.

Laliberte, A. S. and W. J. Ripple. 2004. Range contractions of North American carnivores and ungulates. BioScience 54: 123-138.

Lancia, R. A., Nichols, J. D. and K. H. Pollock. 1994. Estimating the number of animals in wildlife populations. Pages. 215-253. In: Research and management techniques for wildlife and habitats. (ed. T.A. Bookhout). The Wildlife Society, Bethesda (Md.).

Lantschner, M. V., Rusch V. and J. P. Hayes. 2012. Habitat use by carnivores at different spatial scales in a plantation forest landscape in Patagonia, Argentina. Forest Ecology and Management 269: 271–278.

Lekagul, B. and J. A. Mcneely. 1977. Mammals of Thailand. Association for the Conservation of Wildlife, Bangkok, Thailand, 958 pp.

Lekagul, B. and J. A. McNeely. 1988. Mammals of Thailand. 2nd ed. Saha Karn Bhaet, Bangkok.

Le Roux, A., Cherry, M. I. and M. B. Manser. 2008. The effects of population density and sociality on scent marking in the yellow mongoose. Journal of Zoology (London) 275: 33–40.

Levins, R. 1968. Evolution in changing environment. Princeton University Press. Princeton, 120 pp.

Levin, S. A. 1992. The problem of pattern and scale in ecology. Ecology 73: 1943-1967.

Le Xuan Canh, Pham Trong Anh, Duckworth, J. W., Vu Ngoc Thanh and Lic Vuthy. 1997. A survey of large mammals in Dak Lak Province, Vietnam. Unpublished report to IUCN and WWF. Hanoi, Vietnam.

Liberg, O. 1980. Spacing patterns in a population of rural free-roaming domestic cats. Oikos 35: 336-349.

Lidsay, H. M. 1929. A note on *Viverra civettina* Blyth. Journal of the Bombay Natural History 83: 192-193.

Loveridge, A. J. and D. W. Macdonald. 2003. Niche separation in sympatric jackals (*Canis mesomelas* and *Canis adustus*). Journal of Zoology (London) 259: 143-153.

Lozano, J., Moleón, M. and E. Virgós. 2006. Biogeographical patterns in the diet of the wildcat, *Felis silvestris* Schreber, in Eurasia: factors affecting the trophic diversity. Journal of Biogeography 33: 1076–1085.

Lynam, A. J., Round, P., and W. Y. Brockelman. 2006. Status of birds and large mammals of the Dong Phayayen-Khao Yai Forest Complex, Thailand. Biodiversity Research and Training Program and Wildlife Conservation Society, Bangkok, Thailand.

MacArthur, R. H. 1972. Geographical ecology. Princeton University Press, Princeton, New Jersey, USA.

Macdonald, D. 1992. The Velvet Claw: A Natural History of the Carnivores. BCA: London.

Macdonald, D. W. and M. J. Wise. 1979. Notes on the behavior of the Malay civet *Viverrra tangalunga* Gray. Sarawak Museum Journal 48: 295–299.

Mackenzie, D. I. and J. D. Nichols. 2004. Occupancy as a surrogate for abundance estimation. Animal Biodiversity and Conservation 27: 461–467.

MacKenzie, D. I. and L. L. Bailey. 2004. Assessing the fit of site occupancy models. Journal of Agricultural, Biological, and Environmental Statistics 9: 300-318.

Mackenzie, D. I., Nichols, J. D., Lachman, G. B., Droege, S., Royle, J. A. and C. A. Langtimm. 2002. Estimating site occupancy rates when detection probabilities are less than one. Ecology 83: 2248–2255.

MacKenzie, D. I., Nichols, J. D., Royle, J. A., Pollock, K. P., Bailey, L. L. and J. E. Hines. 2006. Occupancy estimation and modeling: inferring patterns and dynamics of species occurrence. Academic Press, San Diego, California, USA.

Maffei, L., Cuellar, E. and A. Noss. 2004. One thousand jaguars (*Panthera onca*) in Bolivian Chaco? Camera trapping in the Kaa-Iya National Park. Journal of Zoology (London) 262: 295–304.

Maffei, L. and A. J. Noss. 2007. How small is too small? Camera trap survey areas and density estimates for ocelots in the Bolivian Chaco. Biotropica 40: 71–75.

Magurran, A. E. 2004. Measuring Biological Diversity. Blackwell Publishing, Malden, MA.

Mahmood, T., Hussain, I., and M. S. Nadeem. 2011. Population estimates, habitat preference and the diet of small Indian mongoose (*Herpestes javanicus*) in Potohar Plateau, Pakistan. Pakistan Journal of Zoology 43: 103-111.

Malo, A. F., Lozano, J., Huertas, D. L. and E. Virgo´s. 2004. A change of diet from rodents to rabbits (*Oryctolagus cuniculus*). Is the wildcat (*Felis silvestris*) a specialist predator? Journal of Zoology (London) 263: 401–407.

Manakadan, R. and S. Sivakumar. 2006. Rusty-spotted cat on India's east coast. Cat News 45: 26.

Manel, S., Buckton, S. T. and S. J. Ormerod. 2000. Problems and possibilities in large-scale surveys: the effects of land use on the habitats, invertebrates and birds of Himalayan rivers. Journal of Applied Ecology 37: 756–770.

Manfredi, C., Soler, L., Lucherini, M. and E. B. Casanave. 2006. Home range and habitat use by Geoffroy´s cat (*Oncifelis geoffroyi*) in wet grassland in Argentina. Journal of Zoology (London) 268: 361-387.

Manser, M. B. and M. B. Bell. 2004. Spatial representation of shelter locations in meerkats, Suricata suricatta. Animal Behaviour 68: 151-157.

Marcum, C. L. and D. O. Loftsgaarden. 1980. A non-mapping technique for studying habitat preferences. Journal of Wildlife Management 44: 963-968.

Marino, J., Bennett, M., Cossios, D., Iriarte, A., Lucherini, M., Pliscoff, P., Sillero-Zubiri, C., Villalba, L. and S. Walker. 2011. Bioclimatic constraints to Andean cat distribution: a modelling application for rare species. Diversity and Distributions 17: 311–322.

Marino, J., Lucherini, M., Villalba, L., Cossios, D., Iriarte, A. and C. Sillero-Zubiri. 2010. Highland cats: ecology and conservation of the rare and elusive Andean cat. In: Biology and conservation of wild felids (eds. D. W. Macdonald and A. J. Loveridge), Pages. 581–596. Oxford University Press, Oxford, United Kingdom.

Martin, A. C., Zim, H. S. and A. L. Nelson. 1951. American wildlife and plants: a guide to wildlife food habits. Dover Publications, New York, USA.

Martinoli, A., Preatoni, D. G., Chiarenzi, B., Waulterts, L. A. and G. Tosi. 2001. Diet of stoats (*Mustela erminea*) in an Alpine habitat: The importance of fruit consumption in summer. Acta Oecologia 22: 45-53.

Medway, L. 1977. Mammals of Borneo. Monograph of the Malaysian Branch of the Royal Asiatic Society 7: 1-172.

Medway, L. 1978. The wild mammals of Malaya and Singapore. Second edition. Oxford University Press. Oxford.

Menon, V. 2003. A Field Guide to Indian Mammals. Handbook, Dorling Kindersley (India) Pvt. Ltd. 201 pages.

Menon, S. and K. S. Bawa. 1997. Applications of Geographical Information Systems, remote sensing, and a landscape ecology approach to biodiversity conservation in Western Ghats. Current Science 73: 134-145.

Mills, M. G. L. 1984. Prey selection and feeding habits of the large carnivores in the southern Kalahari. Koedoe, supplement: 281–294.

Mitra, S. 2005. Gir Forest and the Saga of the Asiatic Lion. New Delhi: Indus Publishing Company. 254 pp.

Moegenburg, S. M. and D. J. Levey. 2003. Do frugivores respond to fruit harvest? An experimental study of short-term responses. Ecology 84: 2600–2612.

Mohanan, M. and N. P. Balakrishnan. 1991. Endangered orchids of Nilgiri Biosphere Reserve, India. In: Proceedings of the symposium on rare, endangered and endemic plants of the Western Ghats. Kerala Forest Department, Thiruvanthapuram.

Monterroso, P., Brito, J. C., Ferreras, P. and P. C. Alves. 2009. Spatial ecology of the European wildcat in a Mediterranean ecosystem: dealing with small radio–tracking datasets in species conservation. Journal of Zoology (London) 279: 27–35.

Morris, D. W., Davidson, D. L. and C. J. Krebs. 2000. Measuring the ghost of competition: insights from density-dependent habitat selection on the co-existence and dynamics of lemmings. Evolutionary Ecology Research 2: 41-67.

Morruzzi, T. L., Fuller, T. K., DeGraaf, R. M., Brooks, R. T., and W. Li. 2002. Assessing remotely triggered cameras from surveying carnivore distribution. Wildlife Society Bulletin 30: 380–386.

Mudappa, D. 1998. Use of camera-traps to survey small carnivores in the tropical rain forest of Kalakad-Mundanthurai Tiger Reserve, India. Small Carnivore Conservation 18: 9-11.

Mudappa, D. 1999. Lesser-known carnivores of the Western Ghats. In: ENVIS Bulletin: Wildlife and protected areas, mustelids, viverrids and herpestids of India, (ed. S. A. Hussain) 2(2): 65-70.

Mudappa, D. 2001. Ecology of the Brown Palm Civet *Paradoxurus jerdoni* in the tropical rainforests of the Western Ghats, India. PhD thesis, Division of Conservation Biology, Salim Ali Centre for Ornithology and Natural History, Coimbatore.

Mudappa, D. 2002. Observations of small carnivores in the Kalakad-Mundanthurai Tiger Reserve, Western Ghats, India. Small Carnivore Conservation 27: 4-5.

Mudappa, D. and R. Chellam. 2001. Capture and immobilization of wild brown palm civets in Western Ghats. Journal of Wildlife Diseases 37: 383–386.

Mudappa, D., Kumar, A. and R. Chellam. 2010. Diet and fruit choice of the brown palm civet *Paradoxurus jerdoni*, a viverrid endemic to the Western Ghats rainforest, India. Tropical Conservation Science 3: 282-300.

Mudappa, D., Noon, B. R., Kumar, A. and R. Chellam. 2007. Responses of small carnivores to rainforest fragmentation in the southern Western Ghats, India. Small Carnivore Conservation 36: 18–26.

Mukherjee, S. 1989. Ecological separation of three sympartric carnivores in Keoladeo Ghana National Park, Rajasthan, India. MSc dissertation, Saurashtra University, Rajkot.

Mukherjee, S. 1998. Habitat use in sympatric small carnivores in Sariska Tiger Reserve, Rajasthan, Western India. PhD thesis, University of Saurashtra, Biosciences department, India.

Mukherjee, S., Goyal, S. P., Johnsingh, A. J. T. and M. R. P. L. Pitman. 2004. The importance of rodents in the diet of jungle cat (*Felis chaus*), caracal (*Caracal caracal*) and golden jackal (*Canis aureus*) in Sariska Tiger Reserve, Rajasthan, India. Journal of Zoology (London) 262: 405-411.

Mukherjee, S. and C. Groves. 2007. Geographic variation in jungle cat (*Felis chaus* Schreber, 1777) (Mammalia, Carnivora, Felidae) body size: is competition responsible? Biological Journal of the Linnean Society 92: 163–172.

Mukherjee, S., Krishnan, A., Tamma, K., Home, C., Navya, R., Joseph, S., Das, A. and U. Ramakrishnan. 2010. Ecology Driving Genetic Variation: A Comparative Phylogeography of Jungle cat (*Felis chaus*) and Leopard cat (*Prionailurus bengalensis*) in India. PLoS ONE 5: e13724.

Myers, N., Mittermeier, R. A. and C. G. Mittermeier. 2000. Biodiversity hotspots for conservation priorities. Nature 403: 853-858.

Nag, K. 2008. Assessing animal abundance from photographic capture data using an occupancy approach. Post-Graduate Programme in Wildlife Biology & Conservation, Centre for Wildlife Studies. (Master's thesis), The Manipal Academy of Higher Education, Manipal University, India.

Nakashima, Y. and A. S. Jumrafiah. 2010. Importance of common palm civets (*Paradoxurus hermaphroditus*) as a long-distance disperser for large-seeded plants in degraded forests. Tropics (*in press*)

Nakashima, Y., Inoue, E., Inoue-Murayam, M. and J. R. Abd Sukor. 2010. Functional uniqueness of a small carnivore as seed dispersal agents: a case study of the common palm civets in the Tabin Wildlife Reserve, Sabah, Malaysia. Oecologia 164: 721-730.

Napolitano, C., Bennett, M., Johnson, W. E., O'Brien, S. J., Marquet, P. A., Barría, I., Poulin, E. and A. Iriarte. 2008. Ecological and biogeographical inferences on two sympatric and enigmatic Andean cat species using genetic identification of faecal samples. Molecular Ecology 17: 678–690.

Nayar, T. S., Rasiya Beegam, A., Mohanan, N. and G. Rajkumar. 2006. Flowering plants of Kerala – a handbook. Tropical Botanic Garden and Research Institute, Thiruvananthapuram, India.

Neal, E. 1986. The natural History of badgers. Helm, London, 238. p.

Negro˜es, N., Sarmento, P., Cruz, J., Eira, C., Revilla, E., Fonseca, C., Sollmann, R., Toˆrres, N. M., Furtado, M. M., Ja´como, A. T. A. and L. Silveira. 2010. Use of camera-trapping to estimate puma density and influencing factors in central Brazil. Journal of Wildlife Management 74: 1195–1203.

Nekaris, K. A. I. 2003. Distribution and behaviour of three small wild cats in Sri Lanka. Cat News 38: 30-32.

Nellis, D. W. and C. O. R. Everard. 1983. The biology of the mongoose in the Caribbean. Studies Fauna Curacao Other Caribbean. lslands. 195: 1-162.

Nichols, J. D., Hines, J. E., Sauer, J. R., Fallon, F. W., Fallon, J. E., and P. J. Heglund. 2000. A double-observer approach for estimating detection probability and abundance from point counts. Auk 117: 393–408.

Nixon, A. M. A., Sidharth, R., Karthik, K., Ashraf, N. V. K. and V. Menon. 2010. Civet Chronicles - Search for the Malabar civet (*Viverra civettina*) in Kerala and Karnataka. Wildlife Trust of India, New Delhi.

Novaro, A. J. 2011. Population density of Geoffroy's cat in scrublands.of central Argentina. Journal of Zoology (London) 283: 37–44.

Nowak, R. M. 1999. Walker's mammals of the world, 6th edn. Johns. Hopkins University Press, Baltimore, MD.

Nowell, K. and P. Jackson. 1996. Wild Cats. Status Survey and Conservation Action Plan. IUCN/SSC Cat Specialist Group, Gland, Switzerland and Cambridge, UK.

Nozaki, E., Azuma, S., Sasaki, H. and H. Torii. 1994. Home range of the Malay Civet (*Viverra tangalunga*) in Teluk Kaba area, Kutai National Park, East Kalimantan, Indonesia. Kyoto University. Over. Rep. Spec. Num. 85–94.

O'Brien, T. G., Kinnaird, M. F. and H. T. Wibisono. 2003. Crouching tigers, hidden prey: Sumatran tiger and prey populations in a tropical forest landscape. Animal Conservation 6: 131–139.

Olff, H. and M. Ritchie. 2002. Fragmented nature: consequences for biodiversity. Landscape and Urban Planning 58: 83-92.

Olson, D. M. and E. Dinerstien. 1998. The global 200: a representative approach to conserving the earth’s most biologically valuable ecoregions. Conservation Biology 12: 502-515.

Osborn, D. J. 1998. The Mammals of Ancient Egypt. Aris and Phillips Ltd., Warminster.

Otis, D. L., Burnham, K. P., White, G. C., and D. R. Anderson. 1978. Statistical inference from capture data on closed animal populations. Wildlife Monographs 62: 1–135.

Papes, M. and P. Gaubert. 2007. Modelling ecological niches from low numbers of occurrences: assessment of the conservation status of poorly known viverrids (Mammalia, Carnivora) across two continents. Diversity and Distributions 13: 890–902.

Parihar, A. S. 1989. Caracal (*Felis caracal* Sohrober) sighted in Panna Forests. Journal of the Bombay Natural History Society 86: 237-237.

Parthasarathy, N. 1999. Tree diversity and distribution in undisturbed and human impacted sites of tropical wet evergreen forest in southern Western Ghats, India. Biodiversity and Conservation 8: 1365-1381.

Patel, K. and P. Jackson. 2005. Rusty-spotted cat in India: new distribution data. Cat News 42: 27.

Patel, K. 2006. Observations of rusty-spotted cat in eastern Gujurat. Cat News 45: 27–28.

Patel, K. 2010. New distribution record data for rusty-spotted cat from Central India. Cat News 53.

Pathak, B. J. 1990. Rusty spotted cat *Felis rubiginosa* Geoffroy: A new record for Gir Wildlife Sanctuary and National Park. Journal of the Bombay Natural History Society 87: 8.

Pearson, R. G., Raxworthy, C. J., Nakamura, M. and A. T. Peterson. 2007. Predicting species distributions from small numbers of occurrence records: a test case using cryptic geckos in Madagascar. Journal of Biogeography 34: 102-117.

Pereira, J. A., Fracassi, N. G., and M. M. Uhart. 2006. Numerical and spatial responses of Geoffroy's cat (*Oncifelis geoffroyi*) to a severe decline in prey abundance in the Monte desert, Argentina. Journal of Mammalogy 87: 1132–1139.

Pereira, J. A. 2009. Efectos del manejo ganadero y disturbios asociados sobre la ecologı´a tro´fica y espacial y la demografı´a del gato monte´s (*Leopardus geoffroyi*) en el desierto del Monte, Argentina. PhD thesis, Universidad de Buenos Aires.

Pereira, J. A., Di Bitetti, M. S., Fracassi, N. G., Paviolo, A., De Angelo, C. D., Di Blanco, Y. E., and W. Peter. 2003. Nepal Development of Participatory Biodiversity Monitoring Concept and Methodology. Report of the Churia Forest Department Nepal. 156 pages.

Peterson, A. T. and R. D. Holt. 2003. Niche differentiation in Mexican birds: using point occurrences to detect ecological innovation. Ecology Letters 6: 774-782.

Phillips, S. 2006. A brief tutorial on Maxent. AT and T Research. Available at: http://www.cs.princeton.edu/~schapire/maxent/tutorial/tutorial.doc

Phillips, S. J. and M. Dudik. 2008. Modeling of species distributions with Maxent: new extensions and a comprehensive evaluation. Ecography 31: 161–175.

Phillips, S. J., Anderson, R. P. and R. E. Schapire. 2006. Maximum entropy modeling of species geographic distributions. Ecological Modelling 190: 231-259.

Phillips, S. J., Dudík, M., J. Elith, Graham, C. H., Lehmann, A., Leathwick, J., and S. Ferrier. 2009. Sample selection bias and presence-only distribution models: implications for background and pseudo-absence data. Ecological Applications 19: 181–197.

Phillips, W. W. A. 1935. Manual of the mammals of Sri Lanka, 2d revised edn., Part III. Wildlife and Nature Protection Society of Sri Lanka, Colombo (1984).

Phillips W. W. A. 1984. Mannual of the mammals of Sri Lanka. Wildlife and Nature Protection Society, Sri Lanka.

Phillips, W. W. A. 1935. Manual of the mammals of Sri Lanka, 2nd revised edn., Part III. Wildlife and Nature Protection Society of Sri Lanka, Colombo (1984).

Pianka, E. R. 1973. The structure of lizard communities. Annual Review of Ecology and Systematics 4: 53–74.

Pielou, E. C. 1966. The measurement of diversity in different types of biological collections. Journal of Theoretical Biology 13: 131-144.

Pillay, R. 2009. Observations of small carnivores in the southern Western Ghats, India. Small Carnivore Conservation 40: 36–40.

Pimentel, D. 1955. Biology of the Indian Mongoose in Puerto Rico. Journal of Mammalogy 36: 62-68.

Pledger, S. and M. Efford. 1998. Correction of bias due to heterogeneous capture probability in capture-recapture studies of open populations. Biometrics 54: 888-898.

Plotkin, J. B., Potts, M. D., YU D. W., Bunyavejchewin, S., Condit, R., Foster, R., Hubbell, S., Lafrankie, J., Manokaran, N., Lee, H. S., Sukumar, R., Nowak, M. A., and P. S. Ashton. 2000. Predicting species diversity in tropical forests. Proceedings of Natural Acadamy of Science 97: 10850-10854.

Pocock, R. I. 1939. The Fauna of British India, including Ceylon and Burma. Mammalia. – Volume 1. Taylor and Francis, Ltd., London. Pp 276–280.

Pocock, R. I. 1941. Fauna of British India, including Ceylon and Burma. Mammalia. Volume 2. London, Taylor and Francis.

Pollock, K. H., Nichols, J. D., Brownie, C. and J. E. Hines. 1990. Statistical inference for capture-recapture experiments. Wildlife Monographs 107: 1–97.

Prakash, I. 1959. Food of some Indian desert mammals. Journal of Biological Sciences 2: 100-109.

Prater, S. H. 1948. The Book of Indian Animals. Bombay Natural History Society, Bombay.

Prater, S. H. 1971. The book of Indian animals. III edition. Bombay Natural History Society, Oxford University Press, Bombay, 324 pp.

Prisazhnyuk, B. E. and A. E. Belousova 2007. Species account for *Felis chaus*. Available at: http://www.biodat.ru/db/rb/rb.php?src=1&vid=389.

Prugh, L. R. 2005. Coyote prey selection and community stability during a decline in food supply. Oikos 110: 253–264.

Rabinowitz, A. R. 1990. Notes on the behavior and movements of the leopard cat, *Felis bengalensis*, in a dry tropical forest mosaic in Thailand. Biotropica 22: 397–403.

Rabinowitz, A. R. 1991. Behaviour and movements of sympatric civet species in Huai Kha Khaeng Wildlife Sanctuary, Thailand. Journal of Zoology (London) 223: 281-298.

Rai, N. D. and A. Kumar. 1993. A pilot study on the conservation of Malabar civet (*Viverra civettina* (Blyth, 1862). Small Carnivore Conservation 9: 3-7.

Rajamani, N., Mudappa, D. and H. Van Rompaey. 2002. Distribution and status of the brown palm civet in the Western Ghats, South India. Small Carnivore Conservation 27: 6 – 9.

Rajaratnam, R. 2000. Ecology of the leopard cat *Prionailurus bengalensis* in Tabin Wildlife Reserve, Sabah, Malaysia. PhD thesis, Universiti of Kabangsaan, Malaysia.

Rajaratnam, R., Sunquist, M., Rajaratnam, L. and L. Ambu. 2007. Diet and habitat selection of the leopard cat (*Prionailurus bengalensis borneoensis*) in an agricultural landscape in Sabah, Malaysian Borneo. Journal of Tropical Ecology 23: 209-217.

Ramachandran, K. K. 1985. A note on the scavenging behaviour of the stripe-necked mongoose on tiger's kill. Journal of the Bombay Natural History Society 82: 182-183.

Ramesh, T. 2010. Prey selection and food habits of large carnivores: tiger *Panthera tigris*, leopard *Panthera pardus* and dhole *Cuon alpinus* in Mudumalai Tiger Reserve, Tamil Nadu. PhD thesis, Saurashtra University, Gujarat, Rajkot.

Ramesh, T., Sridharan, N., Sankar, K., Qureshi, Q., Selvan, K.M., Gokulakkannan, N., Francis, P., Narasimmarajan, K., Jhala, Y.V. and R. Gopal. 2012. Status of large carnivores and their prey in tropical rainforests of South-western Ghats, India. Tropical Ecology 53: 137-148.

Rasa, O. A. E., Wenhold, B. A., Howard, P., Marais, A. and J. Pallett. 1992. Reproduction in yellow mongoose revisited. South African Journal of Zoology 27: 192-195.

Rathore, F. S. and V. Thapar. 1984. Behavioral observations of leopard and jungle cat in Ranthambhor National Park and Tiger Reserve, Rajasthan. In: The plight of the cats: proceedings of the meeting and workshop of the IUCN/SSC Cat Specialist Group at Kanha National Park, Madhya Pradesh, India, 9-12 April 1984. Pp 136-139 Unpublished report, IUCN/SSC Cat Specialist Group, Bougy-Villars, Switzerland.

Renjithkumar, C. R., Harikrishnan, M. and B. M. Kurup. 2011. Exploited fisheries resources of the Pampa River, Kerala, India. Indian Journal of Fisheries 58: 13-22.

Reppucci, J., Gardner, B. and M. Lucherini. 2011. Estimating detection and density of the Andean cat in the high Andes. Journal of Mammalogy 92: 140-147.

Rexstad, E. and K. P. Burnham. 1991. User’s guide for interactive program CAPTURE. Colorado Cooperative Fish and Wildlife Research Unit. Colorado State University, Fort Collins

Roberts, T. J. 1977. The Mammals of Pakistan. Ernest Benn, London.

Roberton, S. I. 2007. The status and conservation of small carnivores in Vietnam. School of Biological Sciences, PhD thesis, University of East Anglia, Norwich, U.K.

Rode, K. D. and C. Robbins. 2000. Why bears consume mixed diets during fruit abundance. Canadian Journal of Zoology 78: 1640–1645.

Rood, J. P. 1986. Ecology and social evolution in the mongooses. In: Ecological aspects of Social Evolution. (ed. D. I. Rubenstein and R. W. Wrangham). Princeton University Press. Princeton. pp 131-152.

Rosalino, L. M., MacDonald, D. W. and M. Santos-Reis. 2004. Spatial structure and land-cover use in a low density Mediterranean population of Eurasian badgers. Canadian Journal of Zoology 82: 1493–1502.

Ross, S., Munkhtsog, B. and S. Harris. 2010. Dietary composition, plasticity, and prey selection of Pallas's cats. Journal of Mammalogy 91: 811-817.

Rovero, F. and A. R. Marshall. 2009. Camera trapping photographic rate as an index of density in forest ungulates. Journal of Applied Ecology 46: 1011–1017.

Rowcliffe, J. M., Field, J., Turvey S. T. and C. Carbone. 2008. Estimating animal density using camera traps without the need for individual recognition. Journal of Applied Ecology 45: 1228 – 1236.

Royle, J. A. 2004. N-mixture models for estimating population size from spatially replicated counts. Biometrics 60: 108-115.

Royle, J. A. and J. D. Nichols. 2003. Estimating abundance from repeated presence-absence data or point counts. Ecology 84: 777–790.

Royle, J. A., Nichols, J. D. and M. Kery. 2005. Modelling occurrence and abundance of species when detection is imperfect. Oikos 110: 353–359.

Rosenweig, M. L. 1966. Community structure in sympatric Carnivora. Journal of Mammalogy 47: 602-612.

Ryley, K. V. 1913. Bombay Natural History Society' Mammal Survey of India. Report No. 11. Coorg. Journal of the Bombay Natural History Society 22: 486–513.

Sarmento, P., Cruz, J., Tarroso, P. and C. Fonseca. 2006. Space and habitat selection by female European wild cats (*Felis silvestris silvestris*). Wildlife Biology in Practice 2: 79-89.

Sarmento, P, Cruz, J., Eira, C. and C. Fonseca. 2009. Spatial colonization by feral domestic cats *Felis catus* of former wildcat *Felis silvestris silvestris* home ranges. Acta Theriologica 54: 31–38.

Sarmento, P. B., Cruz, J. P., Eira, C. I. and C. Fonseca. 2010. Habitat selection and abundance of common genets *Genetta genetta* using camera capture-mark-recapture data. European Journal of Wildlife Research 56: 59-66.

Sanei, A. and M. Zakaria. 2010. Possible first jungle cat record from Malaysia. Cat News 53: 13–14.

Santiapillai, C. M. 2000. The status of mongooses (Family: Herpestidae) in Ruhuna National Park, Sri Lanka. Journal of Bombay Natural History Society 97: 208–214.

Schauenberg, P. 1974. Donne´es nouvelles sur le chat des sables *Felis margarita*, Loche, 1858. Revue Suisse de Zoologie 81: 949-969.

Schauenberg, P. 1978. Reproduction in the Jungle Cat *Felis chaus*. Mammalia 42: 355–358.

Schauenberg, P. 1979. La reproduction du chat des marais, Felis chaus (Guldenstadt, 1776). [Reproduction of the jungle cat Felis chaus.] (in French, English summary). Mammalia 43: 215-223.

Schmidt, K., Nakanishi, N., Okamura, M., Doi, T. and M. Izawa. 2003. Movements and use of home range in the Iriomote cat (*Prionailurus bengalensis irimotensis*). Journal of Zoology (London) 261: 273-283.

Schreiber, A., Wirth, R., Riffel, M. and H. Van Rompaey. 1989. Weasels, civets, mongooses, and their relatives: An action plan for the conservation of mustelids and viverrids. IUCN, Gland, Switzerland.

Seaman, G. A. 1952. The mongoose and Caribbean wildlife. Transactions of the North American Wildlife and Natural Resources Conference 17: 188-197.

Seber, G. A. F. 1982. The estimation of animal abundance and related parameters, Second edn. Charles Griffin, London, UK.

Seshadri, K. S. and T. Ganesh. 2011. Faunal mortality on roads due to religious tourism across time and space in protected areas: A case study from south India. Forest Ecology and Management 262: 1713-1721.

Shanahan, M., So, S., Compton, S. G. and R. T. Corlett. 2001. Fig-eating by vertebrate frugivores : a global review. Biological Reviews 76: 529-572.

Shanker, K. and R. Sukumar. 1998. Community ecology and demography of small mammal populations in insular montane forests in southern India. Oecologia 116: 243–251.

Sharma S. D., Shetty B.V., Vivekanandan K. and N. C. Radhakrishnan. 1978. Flora of Mudumalai Wildlife Sanctuary, Tamil Nadu. Journal of the Bombay Natural History Society 75: 13-42.

Sih, A., Englund, G. and D. Wooster. 1998. Emergent impacts of multiple predators on prey. Trends in Ecology and Evolution 13: 350–355.

Silori, C. S. and B. K. Mishra. 2001. Assessment of livestock grazing pressure in and around the elephant corridors in Mudumalai Wildlife Sanctuary, south India. Biodiversity and Conservation 10: 2181–2195.

Sliwa, A. 2006. Seasonal and sex-specific prey composition of black-footed cats *Felis nigripes*. Acta Theriologica 51: 195–204.

Silver, S. C., Ostro, L. E., Marsh, L. K., Maffei, L., Noss, A. J., Kelly, M. J., Wallace, R. B., Gómez, H. and G. Ayala. 2004. The use of camera traps for estimating jaguar *Panthera onca* abundance and density using capture/recapture analysis. Oryx 38: 148–154.

Silveira, L. 1995. Notes on the distribution and natural history of the pampas cat, *Felis colocolo*, in Brazil. Mammalia 59: 284-288.

Smallwood, K. S. 1997. Interpreting puma (*Puma concolor*) population estimates for theory and management. Environmental Conservation 24: 283–289.

Sokal, R. and J. Rohlf. 1995. Biometry, 3rd ed. W. H. Freeman & Co. San Francisco. 887 p.

Sody, H. J. V. 1931. Six new mammals from Sumatra, Java, Bali and Borneo. Natuurkundig Tijdschrift voor Nederlandsch-Indië 91: 349-360.

Soisalo, M. K. and S. M. C. Cavalcanti. 2006. Estimating the density of a jaguar population in the Brazilian Pantanal using camera-traps and capture–recapture sampling in combination with GPS radio-telemetry. Biological Conservation 129: 487–496.

Solanki, G. S. and P. Chutia. 2004. Ethno Zoological and Socio-cultural Aspects of Monpas of Arunachal Pradesh. Journal of Human Ecology 15: 251-254.

SPSS, 2007. SPSS for Windows. Version 16.0. SPSS, Chicago, IL, USA.

Sterndale, R. A. 1884. Natural history of the Mammalia of India and Ceylon. thacker, Spink & Co., Calcutta, India.

Stallcup, R. 1991. Cats: a heavy toll on songbirds, a reversible catastrophe. Point Reyes Bird Observatory Observer 91: 8-9.

Sukumar, R., Dattaraja, H. S., Suresh H. S., Radhakrishnan J., Vasudeva R., Nirmala S. and N. V. Joshi 1992. Long-term monitoring of vegetation in a tropical deciduous forest in Mudumalai, Southern India. Current Science 62: 608-616.

Sum, E. I. 2011. Microhabitat analysis and population parameters of small carnivores in Sarawak planted forest, with emphasis on the Malay civet *Viverra tangalunga*. Master's dissertation, Department of Bioscience and Chemistry, Universiti Tunku Abdul Rahman, Malaysia, 110 pp.

Su, Su. 2005. Small carnivores and their threats in Hlawga Wildlife Park, Myanmar. Small Carnivore Conservation 33: 6-13.

Su, S. and J. Sale. 2007. Niche differentiation between Common Palm Civet *Paradoxurus hermaphroditus* and Small Indian Civet *Viverricula indica* in regenerating degraded forest, Myanmar. Small Carnivore Conservation 36: 30–34.

Suyanto, A., Yoneda, M., Maryanto, I., Maharadatunkamsi, H. S. and J. Sugardjito. 1998. Checklist of the mammals of Indonesia. Indonesia: LIPI & JICA.

Suen, W.K.-Y. 2002. Lost Mammals. KLH Co. Ltd, Hong Kong.

Sunquist, M. and F. Sunquist. 2002. Wild cats of the World. Chicago: University of Chicago Press. pp. 60–66. ISBN 0-226-77999-8.

Suresh, H. S., Dattaraja, H. S. and R. Sukumar. 1996. Tree flora of Mudumalai Sanctuary, Tamil Nadu, Southern India. Indian Forester 122: 507-519.

Su, Su. 2005. Small carnivores and their threats in Hlawga Wildlife Park, Myanmar. Small Carnivore Conservation 33: 6-13.

Swanson, W. F. 2006. Application of assisted reproduction for population management in felids: The potential and reality for conservation of small cats. Theriogenology 66: 49–58.

Takele, S., B. Afework, B. Gurja and M. Balakrishnan. 2011. A comparison of rodent and insectivore communities between sugarcane plantation and natural habitat in Ethiopia. Tropical Ecology 52: 61-68.

Tehsin, H. R. 1996. Induced Emesis by Jungle Cat (*Felis chaus*), (905), IUCN Cat Specialist Group Digital Cat Library.

Tehsin, R. and F. Tehsin. 1988. Jungle Cat *Felis chaus* and Grey Junglefowl *Gallus sonneratii* (904), IUCN Cat Specialist Group Digital Cat Library.

Terborgh, J. 1986. Keystone plant resources in the tropical forest. Conservation Biology (ed. M. Soule), pp. 330–344. Sinauer Press, Sunderland, MA, USA.

Than Zaw, Saw Htun, Saw Htoo Tha Po, Myint Maung, Lynam, A. J., Kyaw Thinn Latt and J. W. Duckworth. 2008. Status and distribution of small carnivores in Myanmar. Small Carnivore Conservation 38: 2–28.

Thomas, L., Laake, J. L., Strindberg, S., Marques, F. F. C., Buckland, S. T., Borchers, D. L., Anderson, D. R., Burnham, K. P., Hedley, S. L. and J. H. Pollard. 2009. Distance 6.0, Release 2. UK: Research Unit for Wildlife Population Assessment, University of St. Andrews.

Thompson, I. D. and P. W. Colgan. 1990. Prey choice by marten during a decline in prey abundance. Oecologia 83: 443-451.

Thompson, W. L. 2004. Sampling rare or elusive species: concepts, designs, and techniques for estimating population parameters. Island Press, Washington, D.C.

Thorn, J. S., Nijman, V., Smith, D. and K. A. I. Nekaris. 2009. Ecological niche modelling as a technique for assessing threats and setting conservation priorities for Asian slow lorises (Primates: Nycticebus). Diversity and Distributions 15: 289-298.

Tikader, B. K. 1983. Threatened animals of India. Zoological Survey of India, Calcutta.

Tilman, D. 1982. Resource competition and community structure. Princeton University Press, Princeton, New Jersey, USA.

Treves, A., Mwina, P., Plumptre, A. J. and S. Isoke. 2010. Camera trapping forest-woodland wildlife of western Uganda reveals how gregariousness biases estimates of relative abundance and distribution. Biological Conservation 143: 521–528.

Trolle, M., Noss, A. J., Lima, E. D. S. and J. C. Dalponte. 2007. Camera-trap studies of maned wolf density in the Cerrado and the Pantanal of Brazil. Biodiversity and Conservation 16: 1197–1204.

Tsegay, G. and B. Afework. 2006. Population dynamics of pest rodents of Bilalo area, Arsi, Ethiopia. Ethiopian Journal of Biological Sciences 9: 63-74.

van Gruisen, J. and T. Sinclair. 1992. Fur trade in Kathmandu: implications for India. TRAFFIC India, New Delhi.

Van Rompaey, H. and M. N. Jayakumar. 2003. The Stripe-necked Mongoose, *Herpestes vitticollis*. Small Carnivore Conservation 28: 14-17.

Varman, K. S. and R. Sukumar. 1995. The line transect method for estimating densities of large mammals in a tropical deciduous forest: An evaluation of models and field experiments. Journal of Biosciences 20: 273-287.

Vereshchagin, N. K. 1959. The Mammals of the Caucasus: A History of the Evolution of the Fauna. Academy of Sciences of the USSR.

Venkataraman, M., Shanker, K. and R. Sukumar. 2005. Small mammal communities of tropical forest habitats in Mudumalai Wildlife Sanctuary, southern India. Mammalia 69: 349-358.

Vuillermoz, P and A. Sapoznikow. 1998. Hábitos alimenticios y selección de presas de los carnívoros medianos en la Reserva de Vida Silvestre “Campos del Tuyú”. Fundación Vida Silvestre Argentina, Boletín Técnico 44.

Vyas, V. R., Lakhmapurkar, J. J. and D. Gavali. 2007. Sighting of rusty-spotted cat from new localities in central Gujurat. Cat News 46: 18.

Wang, H. and T. K. Fuller. 2001. Notes on the ecology of sympatric small carnivores in southeastern China. Mammalian Biology 66: 251-255.

Wang, H. and T. K. Fuller. 2003. Food habits of four sympatric carnivores in southeastern China. Mammalia 67: 513 – 519.

Wang, P. C., Sheng, H. L. and H. J. Lu. 1976. The diet analysis and captive care of lesser oriental civet. Chinese. Journal of Zoology (London) 2: 39-40.

Wang, S. and Y. Xie. 2004. China Species Red List. Vol. 1 Red List. Higher Education Press, Beijing, China.

Webb-Peploe, C. G. 1947. Field notes on the mammals of south Tinnevelly, south India. Journal of the Bombay Natural History Society 46: 629–644.

Wemmer, C. M. and D. Watling. 1986. Ecology and status of the Sulawasi palm civet, Macrogalidia musschenbroekii Schegel. Biological Conservation 35: 1-17.

Wendeln, M. C., Runkle, J. R., and E. K. V. Kalko. 2000. Nutritional value of 14 fig species and bat feeding preferences in Panama. Biotropica 32: 489– 501.

Wiens, J. A. 1976. Population response to patchy environments. Annual Review of Ecology and Systematics 7: 81–120.

Whelan, C. J., Brown, J. S., Schmidt, K. A., Steele, B. B. and M. F. Willson. 2000. Linking consumer–resource theory and digestive physiology: application to diet shifts. Evolutionary Ecology Research 2:911–934.

White, G. C., Anderson, D. R., Burnham, K. P. and D. L. Otis. 1982. Capture-recapture and removal methods for sampling closed populations. Los Alamos National Laboratory, Los Alamos, New Mexico.

White, P. J., Ralls, K. and C. A. Vanderbilt White. 1995. Overlap in habitat and food use between coyotes and San Joaquin kit foxes. Southwestern Naturalist 40: 342–349.

Wilcoxon, F. 1945. Individual comparisons by ranking methods. Biometrics Bulletin 1: 80–83.

Williams, B. K., Nichols, J. D. and M. J. Conroy. 2002. Analysis and management of animal populations. Academic Press, San Diego, California, US.

Wilson D. E. and D. M. Reeder. 2005. Mammal Species of the World: a Taxonomic and Geographic Reference, 3rd edn. Johns Hopkins University Press, Baltimore, Maryland.

Wilson K. R. and D. R. Anderson. 1985. Evaluation of two density estimators of small mammal population size. Journal of Mammalogy 66: 13-21.

Wilting, A., Cord, A., Hearn, A. J., Hesse, D., Mohamed, A., Traeholdt, C., Cheyne, S. M., Sunarto, S., Jayasilan, M. A., Ross, J., Shapiro, A. C., Sebastian, A., Dech, S., Breitenmoser, C., Sanderson, J., Duckworth, J. W. and H. Hofer. 2010. Modelling the species distribution of flat-headed cats (*Prionailurus planiceps*), an endangered south-east Asian small felid. PLoS ONE 5: e9612.

Wintle, B. A., M. A. McCarthy, C. T. Volinsky and R. P. Kavanagh. 2003. The use of Bayesian model averaging to better represent uncertainty in ecological models. Conservation Biology 17: 1579–1590.

Worah, S. 1991. The ecology and management of a fragmented forest in south Gujarat, India: the Dangs. PhD thesis, University of Pune, Pune.

Workeneh, G., B. Afework, B. Gurja and M. Balakrishnan. 2006. Home range and reproduction of rodents in Maynugus irrigation field, northern Ethiopia. SINET: Ethiopian Journal of Science 29: 57-62.

Wozencraft, W. C. 1989. The phylogeny of the Re-cent carnivora. Pp. 495-535, in Carnivore behavior, ecology, and evolution (J. L. Gittleman, ed.). Cornell University Press, Ithaca, New York, 620 pp.

Wozencraft, W. C. 2005. Order Carnivora. In: D. E. Wilson and D. M. Reeder (eds), Mammal Species of the World: A taxonomic and geographic reference. Third Edition, pp. 532-628. Smithsonian Institution Press, Washington, DC, USA.

Yamakura T. 2000. Spatial patterns in the distribution of tropical tree species. Science 288: 1414- 1418.

Yoganand, T. R. K. and A. Kumar. 1995. The distributions of small carnivores in the Nilgiri Biosphere Reserve, southern India: a preliminary report. Small Carnivore Conservation 13: 1-2.

Yasuda, M., Matsubayashi, H., Rustam, Numata, S., Sukor, J. R. A. and S. Abu Bakar. 2007. Recent records by camera traps in Peninsular Malaysia and Borneo. Cat News 47: 14-16.

Zar, J. H. 1999. Biostatistical Analysis. 4 ed. Upper Saddle River, Prentice Hall.

Zhou, Y. B., Zhang, J., Slade, E., Zhang, L., Palomares, F., Chen, J., Wang, X. and S. Zhang, 2008. Dietary shifts in relation to fruit availability among Masked Palm Civets (*Paguma larvata*) in Central China. Journal of Mammalogy 89: 43-437.

Zielinski, W. J., Truex, R. L., Dunk, J. R. and T. Gaman. 2006. Using forest inventory data to assess fisher resting habitat suitability in California. Ecological Applications 16: 1010–1025.

APPENDICES

Appendix 1. Different forest types in Mudumalai Tiger Reserve.

a) Dry thorn forest

b) Dry deciduous forest

c) Moist deciduous forest

d) Semi-evergreen forest

e) Moist Bamboo brakes

f) Riparian fringe forest

Appendix 2. Small carnivores camera-trapped in Mudumalai Tiger Reserve (2009-2011).

a) Jungle cat b) Rusty-spotted cat

c) Leopard cat d) Small Indian civet

e) Common palm civet f) Brown palm civet

g) Grey mongoose

h) Ruddy mongoose

i) Stripe-necked mongoose

Appendix 3. Indirect signs of small carnivores

a) Small cat track

b) Jungle cat faeces

c) Civet track

d) Civet faeces

e) Mongoose track

f) Mongoose faeces

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Diet of mongoose in Mudumalai Tiger Reserve, southern India.

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Abstract

Feeding ecology in most of the Indian small carnivores is poorly documented. As a baseline study, the diet of mongooses in Mudumalai Tiger Reserve was investigated by faecal analysis from Jan 2010 to December 2011. Mongooses preyed upon hare, rodents, shrew, birds, reptiles, molluscs, crustaceans, pilli millipedes, millipedes, scorpion, five orders of insects, eight species of fruits, reptile egg and carrion. In terms of relative frequency of occurrence, invertebrates were the most common food items, accounting for 48.49% of the materials in faecal samples followed by small mammals (18.29%), fruits (16.01%), reptiles (8.12%), birds (3.56%), other vegetative matter (3.26%), carrion (1.14%) and fish (1.06%). Trophic diversity was highest in deciduous forest followed by dry thorn forest and least in semi-evergreen forest across months. It was evident that mongooses are generalists and as opportunistic feeders they exhibit the ability to switch between different food sources that are readily available in the ecosystem.

Keywords: diet, faecal analyses, mongoose, trophic diversity, Western Ghats.

INTRODUCTION

Herpestids are a family of mongooses, feeding on a wide range of prey including small mammals and birds (including bird eggs), reptiles (especially snakes), insects, and crabs. Some species also include vegetable material in their diets, feeding on tubers, fruits, and berries (Feldhamer *et al.,* 1999). Mongooses are small carnivores with body lengths ranging from around 230 mm to over 750 mm, and weighing from ≤ 1 kg to around 5 or 6 kg (Paradiso, 1975). Herpestids tend to have small heads, pointed snouts, and short, rounded ears that are not as conspicuously erect or pointed as those of viverrids. Some herpestids are gregarious, occurring in colonies or in pairs while some are solitary. Most species are terrestrial, often making intricate burrow systems, but a few also include arboreal habitats in their foraging. Herpestids are found in a wide variety of habitats, ranging from desert to tropical forests. Although they must respond at some level to site productivity, the tremendous variety of food items they eat is produced in such a wide range of environment which is so complex to measure, that the apparent food abundance is not highly predictive of the distribution or abundance of most species.

Distribution pattern of mongooses in India are primarily driven by climatic gradients, temperature and landscape types. The Indian Grey Mongoose (*Herpestes edwardsii*) is mostly recorded in disturbed areas, dry secondary forests, and thorn forests (Gupta, 2011). They are easily targeted by animal trappers and are later sold as pets in local markets; they are still in great demand for the wildlife trade (Hanfee and Ahmed, 1999; Kalle, 2011). The Ruddy Mongoose (*Herpestes smithii*) is found in North-west India (Gupta, 2011), Central India, Southern India in the Nilgiri Biosphere Reserve (Yoganand and Kumar, 1995), and Sri Lanka. The Ruddy Mongoose occurs in forested areas including dry forests, dry thorn forests, and disturbed forests, although there are also a few records from open areas and secluded rice paddy fields (Choudhury *et al.,* 2012). It is thought to live a partially arboreal life, as it hunts, feeds, and rests in trees. The Stripe-necked Mongoose (*Herpestes vitticollis*) is the largest Asian mongoose. It is found in Southwest India (Mudappa, 1998) and Sri Lanka (Santiapillai, 2000). In India, this species is found particularly in the Western Ghats and other hill tracts in the Nilgiris from Coorg (now Kodagu) to Travancore (Pocock, 1939; Prater, 1971; Phillips, 1984; Corbet and Hill, 1992; Mudappa, 1998; Kumara and Singh, 2007; Pillay, 2009) and Dharwar as well as near Bombay (Blanford 1888-1891) to Cape Comorin (Van Rompaey and Jayakumar, 2003). It has been recorded in deciduous and evergreen forest, swamps, plantations, open scrub and along watercourses (Webb-Peploe, 1947; Bali *et al.,* 2007). It feeds on small mammals, birds, eggs, reptiles, fish, insects, grubs, and roots (Van Rompaey and Jayakumar, 2003). The Ruddy and Stripe-necked Mongoose are crepuscular. Unlike Grey and Ruddy Mongoose, the Stripe-necked Mongoose is more common in the hills than in the lowlands, and has been found upto 2,200 m (Van Rompaey and Jayakumar, 2003).

Mongooses are hunted for meat by several tribes and for its hair that is used for making shaving brushes, paint brushes, and good luck charms (Hanfee and Ahmed, 1999). Although mongooses are capable of accommodating a variety of habitats (natural and man-made), it makes it all the more difficult to accurately

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quantify their food abundance. Some predators switch between primary and alternative prey items as the availability of food resources change. Factors affecting food choice by predators generally differ across landscapes, and this knowledge is essential to determine the ability of predators to deal with different prey compositions and habitat characteristics. The mongooses comprise a carnivore group exhibiting considerable diversity in ecology and behaviour. Literature on their feeding habits in India is quite scarce. Most published information on their diet is from the Small Indian Mongoose (*Herpestes javanicus*) (Cavallini and Serafini, 1995; Gorman, 1975; Yamada and Sugimura, 2004; Rana *et al.,* 2005). Several field studies have revealed that the Small Indian Mongoose is primarily an insectivore, though it also feeds opportunistically on small vertebrates (Cavallini and Nel 1995). Ecological information on most generalist species are lacking from the Indian sub-continent. Keeping in mind the dearth of ecological data available on herpestids, the present study aimed to address basic feeding ecology of mongooses from Western Ghats, India through analysis of faecal contents.

STUDY AREA

Our study was conducted in Mudumalai Tiger Reserve (11° 32′–11°43′N; 76°22′–76°45′E), within the Nilgiri Biosphere Reserve of South India. Mudumalai is situated at the junction of the three states of Tamil Nadu, Karnataka, and Kerala with Wayanad Wildlife Sanctuary on the west, Bandipur Tiger Reserve in the north, and the Nilgiri North Forest Division in the south. The reserve spans 321 km^2 of dry thorn, dry deciduous, moist deciduous, semi-evergreen, moist bamboo brakes, and riparian fringe forests (Champion and Seth, 1968). Mudumalai Tiger Reserve receives rainfall from the southwest monsoon which starts in May and ends in August and the northeast monsoon which starts in September and ends in December. Based on the climate of the area, there are three distinct seasons: dry season (January to April), first wet season (May to August), and second wet season (September to December; Varman and Sukumar, 1995).

METHODS

Sample collection and faeces identification

Intensive surveys were carried out from January 2010 to December 2011. Thirty-six trails (each 2-4 km) were surveyed systematically twice a month. During the first visit to each trail all faeces were removed to ensure that only fresh ones were collected during later visits. Faeces were collected occasionally along roads, trails, dried streambeds and around termite mounds, opportunistically. Faeces were identified from their characteristic shape, size, nearby tracks, foraging signs, and frequent defecation sites. Identification characters include their bulky appearance, cylindrical shape with staggered ends (≤ 3 cm in diameter) and a typical odour. Collectively, these procedures suggest that the probability of misidentification of mongoose faeces with other sympatric small carnivores in the study area would be minimal.

Identification of food items

After collection, faeces were placed in plastic bags, tagged and air-dried. Faeces were washed in water through a nylon mesh sieve to separate food remains and later sun-dried. Dietary components were identified to species/taxa based on pertinent anatomical elements such as hair, mandibles, tooth rows, scales, feathers, wings, elytra, fruit cuticles, and seeds (Buskirk *et al.,* 1996). Hair remains were identified with the help of reference slides at the Research Laboratory of Wildlife Institute of India. Experts from the Zoological Survey of India helped us in the identification of rodent specimens collected from the field. To obtain estimates of rodent consumption by the carnivores, we first identified rodent species from the jaws found in faeces by comparing the mandibles with known specimens collected from the field, along with photographs and descriptions from Roberts (1977), Nowak (1999) and Menon (2003). We then calculated the number of individuals of each rodent species consumed by counting jaw parts (jaws were usually present as 2 halves of upper and 2 halves of lower jaws, hence 4 parts would make 1 individual). However in some, all parts were not present, and in these cases the presence of even a single portion or fragment of jaw with dentition was recorded as 1 individual. Wherever possible rodents were identified to the species level, otherwise they were categorized as unidentified rodents. Seeds were identified with the help of experts from the herbarium section at the Wildlife Institute of India and a reference collection maintained in the field.

We estimated diet diversity using the Shannon diversity index (Magurran, 2004), randomizing the original order of scat samples (1000 iterations) using the software EstimateS (Colwell, 2006). We expressed diet composition in 2 complementary ways: percentage of occurrence (PO), and relative frequency of occurrence (RFO).

The PO of prey items in the diet was calculated as number of occurrences of species or taxonomic group x 100/total number of faecal samples (Genovesi *et al.,* 1996; Silva *et al.,* 2005)

The RFO (Loveridge and Macdonald, 2003) was defined as: number of occurrences of species or taxonomic group x 100/total occurrences of all food types in all samples.

The two measures must be treated cautiously because faeces collected in series can contain common elements from a single large meal, or locally similar items and small items can be overestimated (Atkinson *et al.*, 2002). Wilcoxon paired sample test was used to determine differences in observed frequencies of occurrence between food items and to determine differences in frequency occurrence between habitats (Zar, 1999). Diet composition was compared between habitats by χ^2 contingency table analysis using frequency occurrence of major food items (prey items were pooled) and at the taxon/species level (Zar, 1999). SPSS 16 statistical package (SPSS, 2007) was used to process data. All statistical tests were evaluated ≤ 0.05 significance level unless otherwise stated.

RESULTS

Adequacy of sample size

A total of 708 mongoose faeces were collected during the two year study from three habitats; dry thorn (n = 281), deciduous (n = 333) and semi-evergreen (n = 94) forests. The number of faeces collected in each month varied between four and 85 (Fig. 1). A total of 1318 food items were detected in mongoose faeces; the mean ± SD number of food items per faecal sample was 1.89 ± 1.16 (range = 1-5). Totally eight species of fruits, seven species of small mammals, bird, snake, *Calotes spp*, skink, fish, molluscs, arthropods, reptile eggs, and vegetative matter were recorded in mongoose faeces (Table. 1). The number of species appeared to reach an asymptote at about 70 samples in dry thorn and deciduous forest while it did not stabilize completely in semi-evergreen forests probably due to the small sample size (Fig. 2).

Fig 1. Number of mongoose faecal samples collected on a monthly basis during the study period in Mudumalai Tiger Reserve (2010 and 2011).

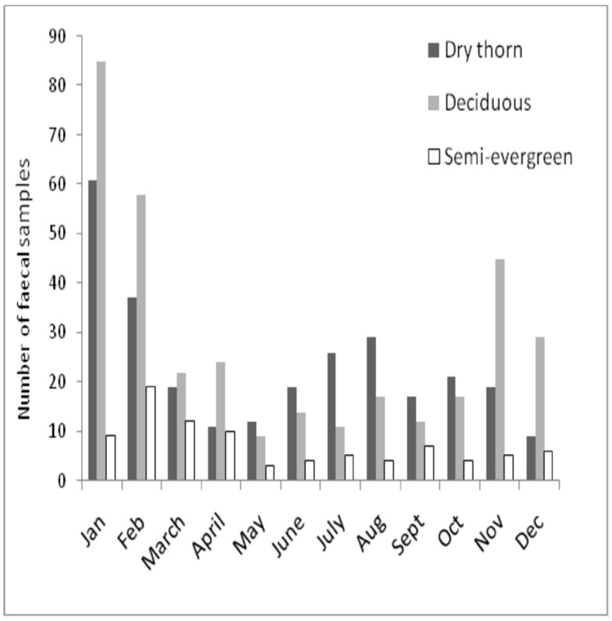
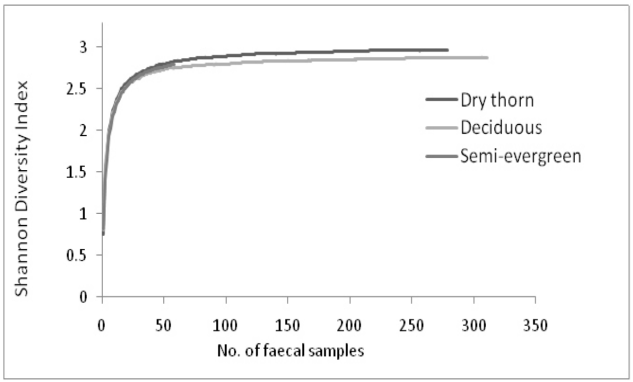


Fig 2. Cumulative dietary diversity indexed by the Shannon diversity index, for mongoose against increasing number of faecal samples for the years 2010–2011 in Mudumalai Tiger Reserve.



Diet composition of mongoose

Animal prey included hare, rodents, shrew, birds, reptiles, molluscs, crustaceans, pilli millipedes, millipedes, scorpion, five orders of insects, eight species of fruits, reptile egg and carrion (Table. 1). The frequency occurrence of major food items did not differ between habitats (Contingency table analysis: dry thorn and deciduous χ^2= 5.7, df = 7, P = 0.57; dry thorn and semi-evergreen χ^2= 6.57, df = 7, P = 0.47; deciduous and semi-evergreen χ^2 = 4.94, df = 7, P = 0.55; Fig. 3). However frequency occurrence of individual prey category at the taxon/species level differed between habitats (Contingency table analysis: dry thorn and deciduous χ^2= 84.66, df = 32, P <0.0001; dry thorn and semi-evergreen χ^2= 110.08, df = 33, P <0.0001; deciduous and semi-evergreen χ^2= 94.46, df = 31, P <0.0001). Invertebrates were the most common food items, accounting for more than 48.49% of the materials in faecal samples followed by small mammals that accounted for about 18.29% of the materials in faecal

Fig 3. Relative frequency occurrence of food items of the mongoose in the dry thorn, deciduous and semi-evergreen forests.

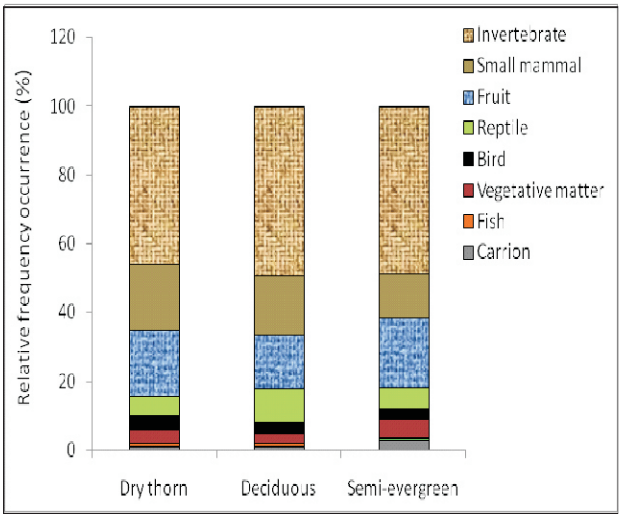


Table 1. Diet composition of mongoose based on fecal analysis in Mudumalai Tiger Reserve

Food item	n	PO	RFO
Small mammal	**241**	**34.04**	**18.29**
Unidentified rodent	84	11.86	6.37
Lepus nigricollis	9	1.27	0.68
Mus spp	62	8.76	4.70
Cremnomys blanfordi	33	4.66	2.50
Rattus spp	28	3.95	2.12
Suncus montanus	22	3.11	1.67
Millardia meltada	2	0.28	0.15
Tatera indica	1	0.14	0.08
***Axis axis* (carrion)**	**15**	**2.12**	**1.14**
Bird	**47**	**6.64**	**3.56**
Unidentified bird	45	6.36	3.41
Quail	2	0.28	0.15
Reptile	**107**	**15.11**	**8.12**
Calotes spp	61	8.62	4.63
Snake	34	4.80	2.58
Skink	12	1.69	0.91
Fish	**14**	**1.98**	**1.06**
Invertebrates	**639**	**90.25**	**48.49**
Mollusc	8	1.13	0.61
Crustacean	43	6.07	3.26
Pilli millipede	228	32.20	17.30
Millipede	35	4.94	2.66
Dragonfly	1	0.14	0.08
Dung beetle	233	32.91	17.68
Red ant	2	0.28	0.15
Scorpion	29	4.10	2.20
Cricket	9	1.27	0.68
Termite	24	3.39	1.82
Grasshopper	7	0.99	0.53
Moth	1	0.14	0.08
Unknown insect	19	2.68	1.44
Reptile egg	**1**	**0.14**	**0.08**
Fruit	**211**	**29.8**	**16.01**
Zizyphus eonophilia	65	9.18	4.93
Unidentified fruit	54	7.63	4.10
Grewia tiliifolia	38	5.37	2.88
Cordia obliqua	21	2.97	1.59
Lantana camara	17	2.40	1.29
Bridelia crenulata	8	1.13	0.61
Zizyphus mauritiana	3	0.42	0.23
Syzigium cumini	3	0.42	0.23
Ficus spp	2	0.28	0.15
Vegetative matter	**43**	**6.07**	**3.26**

n = sample size, PO = Percentage occurrence, RFO = relative frequency occurrence

samples. Fruit presented 16.01% of the materials. Reptiles formed 8.12% of materials and birds formed 3.56% of the materials in fecal samples. Fish represented poorly in the diet accounting for 1.06% of the dietary materials.

Invertebrates and small mammals were consumed throughout the year in the three habitats. Overall invertebrates were consumed significantly higher than small mammals, bird, reptile, fish, vegetative matter, carrion (Wilcoxon paired sample test: z = -3.06; P = 0.002) and fruit (Wilcoxon paired sample test: z = -2.98; P = 0.003; Fig 4a). Small mammal remains occurred higher than birds, fish, vegetative matter, (Wilcoxon paired sample test: z = -3.06; P = 0.002) and reptile (Wilcoxon paired sample test: z = -2.98; P = 0.003; Fig 4a). Monthly occurrence of bird remains was higher in thorn forest than semi-evergreen forest (Wilcoxon paired sample test: z = -2.29; P = 0.02; Fig 4b and d). Reptile occurrence was higher in deciduous than thorn forest (Wilcoxon

Fig 4a) Monthly variations in the main food types ingested by mongoose; a) overall b) dry thorn c)deciduous d) semi-evergreen forests. Food types are expressed as relative percentage occurrence.

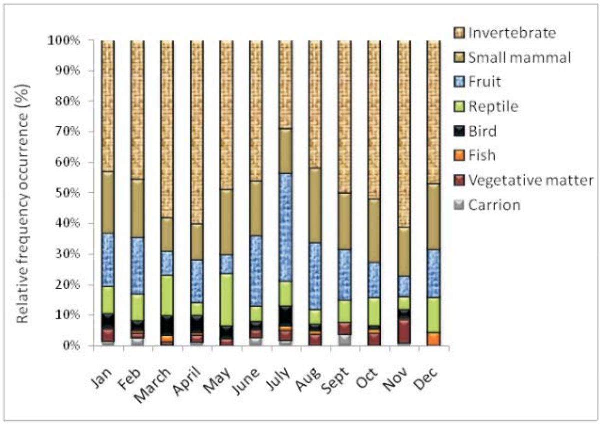


Fig 4 b)

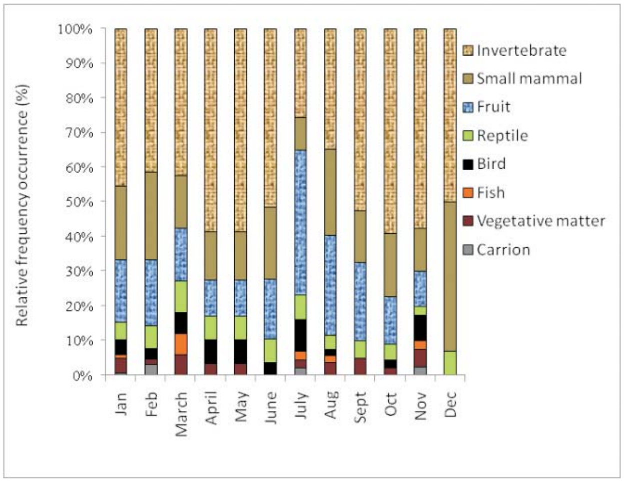


Fig 4 c)

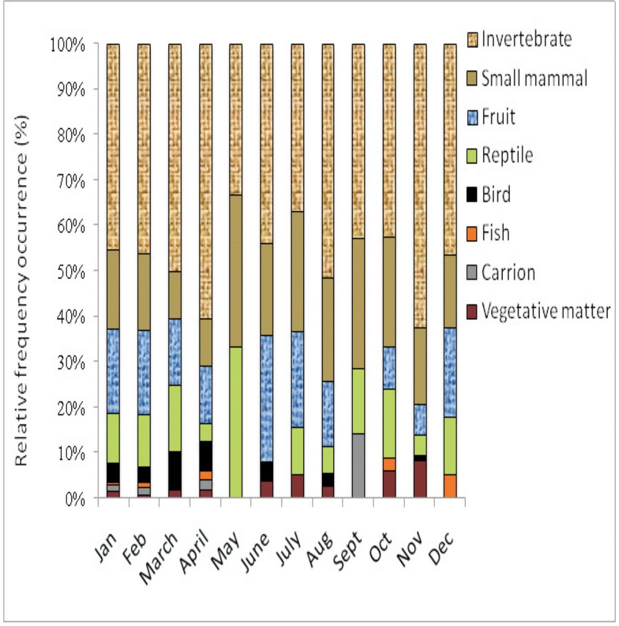
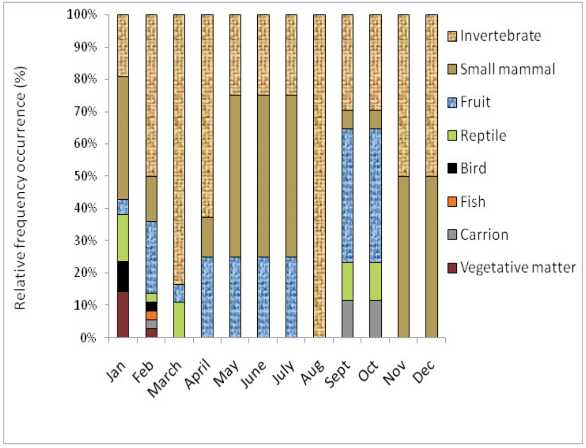


Fig 4 d)



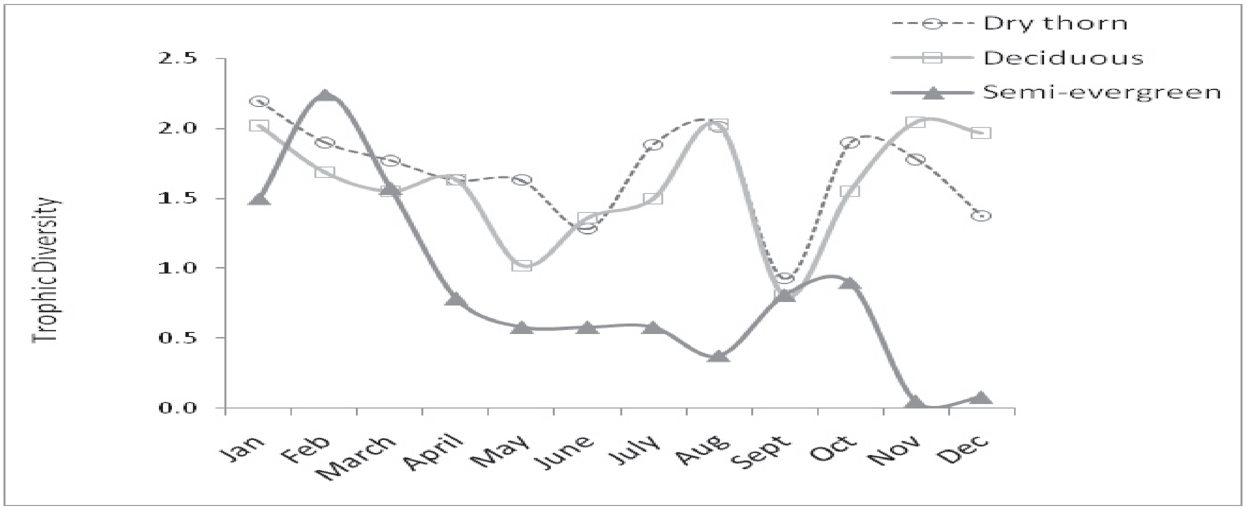
paired sample test: z = -2.12; P = 0.03; Fig 4b and c) and semi-evergreen forest (Wilcoxon paired sample test: z = -2.62; P = 0.009; Fig 4d).

Trophic diversity was significantly higher in deciduous than semi-evergreen forest (Wilcoxon paired sample test: z = -2.43; P = 0.01) and significantly higher in thorn forest than semi-evergreen forest (Wilcoxon paired sample test: z = -2.82; P = 0.005; Fig. 5). In general, trophic diversity was < 0.5 during Aug, Nov and Dec in semi-evergreen forest, while in other habitats diversity was > 0.5 throughout the year (Fig. 5).

DISCUSSION

Validity of the diet data should be considered cautiously, bearing in mind that the composition of diet generally changes according to the available resources regulated by the time of year. Evaluation parameters such as frequency have a descriptive function that allows investigation of the exclusive differences among trophic categories. The most important food for mongooses was invertebrates. The data obtained indicate that they are the specialist consumers of subsoil prey. In any case, preliminary observations on this aspect allow us to formulate that the edaphic fauna resource available is the best the region can offer; it is also reasonably stable and abundant which explains why these burrowing animals have developed specific strategies to exploit them. Mongooses have an elongated, narrow snout and sharp dentition; their foraging behaviour is related to these characteristics since they spend a great deal of time searching for food, excavating the soil looking for small prey. In these activities, the long snout is particularly useful, together with the prey they ingest is exclusively represented by arthropods. Such morphological characters supporting their adaptations correspond to insectivory. Besides the consumption of principal food items, they also supplemented their diet with other food items. Small mammals contribute to the major biomass of numerous small carnivores which was also evident from this study. Bird species that spend time mainly on the ground are also potential sources of

Fig 5. Temporal variation in diet diversity of mongoose in Mudumalai Tiger Reserve from 2010-2011



food for mongoose. Small mammals are generally abundant in the ecosystem which increases the encounter rate with predators. They typically contain a greater percentage of digestible matter with respect to similar-sized birds or reptiles (Johnson and Hansen, 1979).

The presence of grass/herbs leaves in the faeces is consistent with other studies, mostly concluding a possible role of leaves in scouring the intestine during digestion. Invertebrates were the main dietary component of mongoose followed by small mammals. Mongooses consumed bird remains more in the thorn forest than semi-evergreen forest while reptiles were consumed higher in the deciduous than thorn and semi-evergreen forest. Trophic diversity was higher in deciduous forest since the type of food preferred (mainly invertebrates) could be most abundant due to the humidity and moist sub-soil conditions favouring dung beetles and millipedes. The general feeding habits appear to confirm results from available studies. A recent study by Rahman *et al.,* (2012) reported that invertebrate taxonomic richness was found highest in moist-deciduous and semi-evergreen forests of the Nilgiri Biosphere Reserve. Mongooses are ground-foraging, burrowing species that often live in underground burrows or in dense shrub cover. Some species like the Small Indian Mongoose are largely insectivorous while others may largely consume fruits for parts of the year (Seaman and Randall, 1962). The percent frequency of the food items from the faecal pellets has shown that the major food component (more than 65%) of the diet of the small Indian mongoose were rodents (Mahmood *et al.,* 2011). Eggs are a favoured delicacy in mongooses (Prater, 1971; Gorman, 1975; Paradiso, 1975; Roberts, 1977; Siddiqui *et al.,* 2003) which was evident in our study. Other species like the Spanish Mongoose (*Herpestes ichneumon*) makes up the greater part of its diet by consuming young rabbits and supplements it with reptiles, insects, mushrooms, etc. (Delibes, 1976). Parts of insects and a scorpion were found in the stomachs of two small Indian mongooses collected from Rajasthan Desert in India (Prakash, 1959). So the findings in literature confirm results of the present study that mongooses were opportunistic with varied diet. The high frequency occurrence of pilli millipedes and insects in the diet of mongoose was probably due to their dentition designed to crush food items that are calcareous. Large mammal remains found in faeces probably represent carrion from dead animals in most cases. It was hard to observe foraging signs of mongooses due to the high understory cover and overgrown weeds, *Lantana camara* and *Eupatorium odoratum* formed wide green stretches along the banks of streams throughout the forest. Quite often the first author would observe holes dug out from the soft soil with mongoose tracks along dried streambeds indicating the species look out for sub-soil prey. However, few direct observations of mongoose foraging were recorded during the study. Once, the first author observed a Ruddy Mongoose feeding on crabs over a cemented ledge of the Marvakandy dam in the region. On several occasions she has observed pairs of Stripe-necked Mongoose looking for prey along dried streambeds.

The food items selected by lesser carnivores are generally abundant in the environment yet not all food items can be quantified easily due to high diversity. Future research must include quantification of invertebrates for clarification in their true prey selection by small carnivores, although this would still be quite challenging and suffer from technical limitations. The analysis of faecal samples is a method that permits estimates of the trophic requirements of the species of interest without the need to sacrifice, manipulate or directly observe the animals. Nevertheless, the method has limitations in detecting the totality of food ingested initially by the animal. The identifiable remains in an excrement sample are no more than the fragments of those foods that are ingested totally, or that remain as normal residues of the digestive process. The combination of small size, diurnal activity rhythms, and insectivorous diet is a necessary (although not sufficient) condition for group living in herpestids (Rood, 1986).

Future studies addressing prey activity and their influence on feeding habits and use of space and time could shed light upon the mechanisms involved in both prey and predators. Prey abundance data would better represent the energetic contribution of each food category, without relinquishing its descriptive nature. Although this study could not address feeding habits at the species level, it was useful in providing information on dietary trends at the community level. In the future, feeding ecology should be addressed at the species through radio-tracking or DNA identification of faeces to reveal precise feeding strategies exhibited by different mongoose species. Further species wise diet data would also aid in understanding how dental adaptations play a major in selection of food items.

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REFERENCES

Atkinson, R.P.D., Macdonald, D.W. and Kamizola, R. 2002. Dietary opportunism in side-striped jackals *Canis adustus* Sundevall. *J. Zool. Lond.,* 257: 129–139.

Bali, A., Kumar, A. and Krishnaswamy, J. 2007. The mammalian communities in coffee plantations around a protected area in the Western Ghats, India. *Biol. Conserv*., 139: 93-102.

Blanford, W.T. 1888–91. The fauna of British India, including Ceylon and Burma. Volume I, Mammalia. Taylor and Francis, London.

Buskirk, S.W., Ma, Y.Q., Xu, L. and Jiang, Z.W. 1996. Diets of, and prey selection by, sables (*Martes zibellina*) in northern China. *J. Mammal.,* 77: 725–730.

Cavallini, P and Nel, J.A.J. 1995. Comparative behavior and ecology of two sympatric mongoose species (*Cynictis penicillata* and *Galerella pulverulenta*). *S. Afr. J. Zool.,* 30: 46-49.

Cavallini, P and Serafani, P. 1995. Winter diet of the small Indian mongoose (*Herpestes auropunctatus*) on an Adriatic Island. *J. Mammal.,* 76: 569-574.

Champion, H.G. and Seth, S.K. 1968. A revised survey of the forest types of India. The Manager of Publications, Government of India, New Delhi.

Choudhury, A., Wozencraft, C., Muddapa, D. and Yonzon, P. 2008. *Herpestes smithii*. In: IUCN 2012. IUCN Red List of Threatened Species. Version 2012.1.<www.iucnredlist.org>. Downloaded on 29 September 2012.

Colwell, R.K. 2006. EstimateS: Statistical estimation of species richness and shared species from samples. Version 8.0. Online at http://purl.oclc.org/estimates

Corbet, G.B. and Hill, J.E. 1992. Mammals of the Indo-Malayan Region: A Systematic Review. Oxford University Press, Oxford, UK.

Delibes, M. 1976. The diet of the Spanish mongoose (*Herpestes ichneumon*) in Spain. *Saeugetierkd Mitt*., 24: 38-42.

Genovesi, P., Secchi, M. and Boitani, L. 1996. Diet of stone martens: An example of ecological flexibility. *J. Zool. Lond.,* 238: 545–555.

Gorman, M.L. 1975. The diet of feral *Herpestes auropunctatus* in the Fijian Island. *J. Zool. Lond.,* 175: 273-278.

Gupta, S. 2011. Ecology of medium and small sized carnivores in Sariska Tiger Reserve, Rajasthan, India. Ph.D. dissertation, Saurashtra University. P. 156.

Feldhamer, G.A., Drickamer, L.C., Vessey, S.H. and Merritt, J.F. 1999. Mammalogy. Adaptation, Diversity and Ecology. WCB McGraw-Hill, Boston. P. xii+563.

Hanfee, F. and Ahmed, A. 1999. Some observations on India's illegal trade in mustelids, viverrids and herpestids. *In*: Hussain, S.A. (Ed.), ENVIS Bulletin: Wildlife and protected areas. Mustelids, viverrids and herpestids of India, 2(2): 113-115.

Johnson, M.K. and Hansen, R.M. 1979. Coyote food habits on the Idaho National Engineering Laboratory. *J. Wildl. Manage.,* 43: 951–956.

Kalle, R. 2011. Wildlife, a treasure or a delicacy? *Hornbill,* P. 32-33.

Kumara, H.N. and Singh, M. 2007. Small carnivores of Karnataka: distribution and sight records. *J. Bombay Nat. Hist. Soc.*, 104: 155-162.

Loveridge, A.J. and Macdonald, D.W. 2003. Niche separation in sympatric jackals (*Canis mesomelas* and *Canis adustus*). *J. Zool. Lond*., 259: 143–153.

Mahmood T., Hussain I. and Nadeem M.S. 2011. Population estimates, habitat preference and the diet of small Indian mongoose (*Herpestes javanicus*) in Potohar Plateau, Pakistan. *Pak. J. Zool.,* 43: 103-111.

Magurran, A.E. 2004. Measuring Biological Diversity. Blackwell Publishing, Malden, MA.

Menon, V. 2003. A field guide to Indian mammals. Dorling Kindersley (India), Delhi.

Mudappa, D. 1998. Use of camera-traps to survey small carnivores in the tropical rain forest of Kalakad-Mundanthurai Tiger Reserve, India. *Small. Carniv. Conserv.,* 18: 9-11.

Nowak, R.M. 1999. Walker''s Mammals of the World. Sixth Edition. Volume I. Johns Hopkins University Press, Baltimore.

Paradiso, J.L. 1975. Walker's Mammals of the World, Third Edition. Johns Hopkins University Press, Baltimore.

Phillips, W.W.A. 1984. Mannual of the mammals of Sri Lanka. Wildlife and Nature Protection Society, Sri Lanka.

Pillay, R. 2009. Observations of small carnivores in the southern Western Ghats, India. *Small. Carniv. Conserv.,* 40: 36–40.

Pocock, R.I. 1939. The Fauna of British India, including Ceylon and Burma. Mammalia. – Volume 1. Taylor and Francis, Ltd., London. P. 276–280.

Prakash, I. 1959. Food of some Indian desert mammals. *J. Biol. Sci.,* 2: 100-109.

Prater, S.H. 1971. The book of Indian animals. III edition. Bombay Nat. Hist. Soc. Oxford University Press, Bombay, P. 324.

Rahman, P.M., Varma, R.V. and Sileshi, G.W. 2012. Abundance and diversity of soil invertebrates in annual crops, agroforestry and forest ecosystems in the Nilgiri Biosphere Reserve of Western Ghats, India. *Agrofor. Sys.,* 85: 165-177.

Rana, S., Smith, S.M. and Siddiqui, M.J.I. 2005. Scat Analysis of Small Indian Mongoose (*Herpestes auropunctatus*) Feeding on Fauna of Some High and Relatively Low Input Crop Fields. *Int. J. Agric. Biol.,* 7: 777-780.

Roberts, T.J. 1977. The Mammals of Pakistan. Ernest Benn, London.

Rood, J.P. 1986. Ecology and social evolution in the mongooses. *In:* Rubenstein, D.I. and Wrangham, R.W. (Eds.), Ecological aspects of Social Evolution. Princeton University Press. Princeton. P. 131-152.

Santiapillai, C. 2000. The status of mongooses (Family: Herpestidae) in Ruhuna National Park, Sri Lanka. *J. Bombay. Nat. Hist. Soc.,* 97: 208-214.

Seaman, G. and Randall, J. 1962. The mongoose as a predator in the Virgin islands. *J. Mammal.*, 43: 544-546.

Siddiqui, M.J.I., Rana, N. and Rana, S.A. 2004. Analysis of the scats of small Indian mongoose (*Herpestes auropunctatu*) with special reference to insect fauna in croplands of Faisalbad (Pakistan). *Pak. Entomol*., 26: 95-99.

Silva, S.I., Bozinovic, F. and Jaksic. F.M. 2005. Frugivory and seed dispersal by foxes in relation to mammalian prey abundance in a semiarid thornscrub. *Austral. Ecol.,* 30: 739–746.

SPSS, 2007. SPSS for Windows. Version 16.0. SPSS, Chicago, IL, USA.

Varman, K.S. and Sukumar, R. 1995. The line transect method for estimating densities of large mammals in a tropical deciduous forest: An evaluation of models and field experiments. *J. Biosc.,* 20: 273-287.

Van Rompaey, H. and Jayakumar, M.N. 2003. The Stripe-necked Mongoose, *Herpestes vitticollis*. *Small. Carniv. Conserv.,* 28: 14-17.

Webb-Peploe, C.G. 1947. Field notes on the mammals of south Tinnevelly, south India. *J. Bombay. Nat. Hist. Soc.,* 46: 629–644.

Yamada, F, and Sugimura, K. 2004. Negative Impact of an Invasive Small Indian Mongoose *Herpestes javanicus* on Native Wildlife Species and Evaluation of a Control Project in Amami-Ohshima and Okinawa Islands, Japan. *Global. Environ. Res.,* 8: 117-124.

Yoganand, T.R.K. and Kumar, A. 1995. The distributions of small carnivores in the Nilgiri Biosphere Reserve, southern India: a preliminary report. *Small. Carniv. Conserv.,* 13: 1-2.

Zar, J.H. 1999. Biostatistical analysis. Prentice Hall, Upper Saddle River.

HABITAT OCCUPANCY OF SMALL FELIDS, VIVERRIDS AND HERPESTIDS IN MUDUMALAI TIGER RESERVE, TAMIL NADU

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Small carnivores represent some of the most threatened species, but also the least known because of their rarity, elusiveness and cryptic habits. Conservation efforts commonly use habitat models as a surrogate of empirical data on monitoring the status of multi-species when documenting species occurrence as well as to predict the efficacy of management interventions. Occupancy surveys offer a potentially efficient and robust means of assessing elusive carnivore populations. We evaluated the influence of habitat models on site occupancy rates of small felids, viverrids and herpestids using presence/absence data in a capture-recapture framework from systematically placed remote camera traps (1.25 km x 1.25 km grids) in Mudumalai Tiger Reserve, Tamil Nadu over a period of 6 months from November 2009 to April 2010. We identified 64 sites covering three different habitat types (dry thorn forest, deciduous forest and semi-evergreen forest) which were repeatedly surveyed for 30 days in the dry season amounting to 1920 trap nights. The habitat models increased site occupancy rates of certain small carnivores. The dry thorn forest influenced site occupancy rate of jungle cat (*psi* = $0.19 \pm 0.06_{SE}$), small Indian civet (*psi* = $0.26 \pm 0.09_{SE}$), ruddy mongoose (*psi* = $0.14 \pm 0.06_{SE}$) and grey mongoose (*psi* = $0.13 \pm 0.07_{SE}$) while the semi-evergreen forest increased site occupancy of brown palm civet (*psi* = $0.33 \pm 0.27_{SE}$). The study revealed that habitat type contributed to the distribution of small carnivores indicating species-habitat relationships. Occupancy estimates from this study provide a benchmark for long-term monitoring of small carnivore populations in the future.