IMPACTS OF HABITAT FRAGMENTATION ON THE TERRESTRIAL SMALL MAMMAL COMMUNITIES IN THE TROPICAL RAIN FOREST OF THE ANAIMALAI HILLS IN THE WESTERN GHATS, SOUTH INDIA

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by
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CERTIFICATE

This is to certify that the thesis, entitled **"Impacts of Habitat Fragmentation on the Terrestrial Small Mammal Communities in the Tropical Rain Forest of the Anaimalai Hills in the Western Ghats, South India"** is a record of original research work done by **Mr. A. Prabhakar** in the Division of Conservation Biology, Sálim Ali Centre for Ornithology and Natural History, as a full time Reasearch Scholar during the period of study 1994-1998 under my guidance and supervision for the award of the Degree of Doctor of Philosophy in Zoology. I further certify that this research work has not previously formed the basis for the award of any other degree or Diploma or Associateship or Fellowship or other similar title to any candidate of this or any other University.

Signature of the Guide

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DECLARATION

I do hereby declare that the thesis entitled **"Impacts of Habitat Fragmentation on the Terrestrial Small Mammal Communities in the Tropical Rain Forest of the Anaimalai Hills in the Western Ghats, South India"** submitted to the Bharathiar University, Coimbatore, for the award of the Degree of Doctor of Philosophy in Zoology, is a record of original and independent research work done by me during 1994-1998 under the supervision and guidance of **Dr. Ajith Kumar**, Principal Scientist, Division of Conservation Biology, Sálim Ali Centre for Ornithology and Natural History, Coimbatore, and it has not previously formed the basis for the award of any Degree, Diploma, Associateship, Fellowship or other similar title to any candidate of any University.

Signature of the Candidate

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Head of the department

SUMMARY

The loss and fragmentation of the species-rich tropical rain forests is the most serious conservation crisis currently facing us. In the Western Ghats in south India as much as 40% of the forests was lost between 1920s and 1990s, with a 17-fold increase in forest patches. Such fragmentation of forests leads in the long run to loss of species due to the problems faced by small isolated populations, the cascading effects of the changes in the micro and macro climate, and the effects of invading species. The goal of this study was to make an assessment of the changes in the small mammal community due to the fragmentation of their rain forest habitat in the Western Ghats. The study animals included rodents of the Families Muridae and Muscardinidae, and shrews of the Order Insectivora. The specific objectives were: (a) to evaluate the differences among rain forest fragments in the community structure of small mammals, in relation to macrohabitat features of the forest fragments; (b) to examine the synchrony among forest fragments in the seasonal variation of several demographic parameters; and (c) to identify species differences in microhabitat preferences and examine their relevance to the changes in community structure due to habitat fragmentation.

This study was carried out in the Indira Gandhi Wildlife Sanctuary (Tamil Nadu State) in the Anaimalais Hills of the Western Ghats. During a preliminary survey I identified 25 rain forest fragments. For this study, I selected seven forest fragments which varied in area (from less than 10 ha to more than 2,000 ha) and other habitat features. Small mammals were sampled using Sherman live traps, each trapping session lasting for five days. The number of traps per session varied from 50 to 200 depending on the area of the fragment. Seventeen microhabitat parameters were measured from 1m^2 quadrats centred on all sprung trap stations, and from every alternative trap station. The sampling was replicated in four seasons.

Live trapping covered 10, 595 trap nights with an overall trapping success of 5.40 animals per 100 trapnights. I captured six species from the Family Muridae, one endemic dormouse of the Family Muscardinidae (the Malabar spiny dormouse, *Platacanthomys lasiurus*) and one endemic shrew (*Sorex niger*) from the Order Insectivora. The most abundant species *R. r. wroughtoni* formed nearly 50% of the total captures, followed by *S. niger* (30%), *M. booduga* (15%) and *M. musculus* (1.76%). The other four species formed less than 4% of the captures.

A principal component analysis identified two factors that accounted for 77% of the total variance in macrohabitat parameters and their Coefficient of Variations (CVs). Factor 1 represented the status of the fragment in terms of area, tree density *etc.,* while Factor 2 represented spatial heterogeneity. Factor 1 was linearly related to area of the fragment while Factor 2 had a quadratic relation. Seasonal species richness showed a nonlinear relationship with some of the habitat parameters and habitat status (Factor 1), and a linear relationship with habitat heterogeneity (Factor 2). The mean seasonal species richness was highest in medium sized fragments, which were moderately disturbed and had the highest spatial heterogeneity. This was due to the greater abundance of invaders which are commensal to man.

The total abundance of rodents and shrews did not show any clear relationship with habitat parameters, even though individual species did. Thus, the abundance of *R. r. wroughtoni* increased with habitat status and decreased with habitat heterogeneity, while *M. booduga* and *M. musculus* showed an opposite relation. *S. niger* showed no clear relationship. Species overlap between fragment pairs was generally high and was not related to similarity in habitat parameters. Fragments with similar spatial heterogeneity had similar species composition. The proximity of human settlements seemed to affect the similarity in species occurrence among fragments.

The subadult to adult ratio over all the fragments indicated that the breeding period for all three species was in the late summer and southwest monsoon. This might be correlated with peak fruit and foliage production in the habitat. There was no difference between forest fragments of four size classes in the age ratio in the case of *R. r. wroughtoni*. The differences in the other two species did not show any consistent pattern with fragment area. There was some synchrony among fragments in the seasonal variation of age ratio only in the case of *S. niger*.

The adult sex ratio did not show any consistent pattern with fragment area in *R. r. wroughtoni* and *S. niger*, while in *M. booduga* it seemed to decrease with fragment area. Synchrony was evident only in the case of *S. niger*. When data from fragments were pooled, the body weights of all species were greatest during the southwest monsoon. The body weight of *R. r. wroughtoni* increased with increasing tree density, decreasing CV of tree density, and increasing Factor 1 score. *M. booduga* showed an opposite relation with the same parameters while *S. niger* did not show significant relationship. The body weight of all species was greater in the wet seasons in most fragments.

The lack of demographic synchrony among forest fragments in the case of *R. r. wroughtoni* and *M. booduga* might be related to the differences among fragments in plant species composition and the lack of dispersal between fragments. Invertebrates, the major food of *S. niger*, may show less variation among fragments. Moreover, this species might be also able to disperse between fragments. These two factors might promote greater demographic synchrony among forest fragments in the case of *S. niger*.

Bivariate analysis showed considerable overlap in the responses of *R. r. wroughtoni* and *S. niger* to variation in almost all microhabitat variables. *M. booduga* also showed a similar response, but captures were greatest at medium values of some variables such as shrub density and herb cover. In contrast, captures of *M. musculus* were greater at low values of most of these variables, and at mid values of the others. Logistic regression segregated these species better with reference to their response to the microhabitat variables. *R. r. wroughtoni* showed significant response to the presence of rattans, buttressed trees and shrub density. *S. niger* responded to shrub density, rock cover, litter cover and herb cover. *M. booduga* responded to shrub density, rock cover, herb cover and canopy cover, the last of these negatively. Shrub density was the only variable to which all three species responded. However, nonlinear response of *M. booduga* to some variables may not be represented adequately in the logistic regression.

Since *R. r. wroughtoni* and *M. musculus* are almost totally segregated in their microhabitat selection, competition among them is unlikely to be a factor in the decline of *R. r. wroughtoni* in forest fragments. Overlap with *M. booduga* in microhabitat selection can lead to competition among them in moderately disturbed fragment, and thus to the decline of *R. r. wroughtoni.* However, loss of optimal microhabitats of *R. r. wroughtoni* could also be a reason. The insectivorous *S. niger* is more generalist than rodents, and is unlikely to be affected by competition with invading or resident rodents, in spite of considerable overlap among them in microhabitat preferences.

The low species richness among the rodents, mice, and shrews in the Western Ghats may reflect the faunal impoverishment in these taxa in India. However, lack of taxonomical expertise and systematic sampling might also be reasons. Human settlements near forest fragments have introduced several species of commensal rodents into forest fragments, while two endemics (*P. lasiurus* and *R. r. wroughtoni*) seem to have declined. Changes in the community structure of small mammals could have major ecological implications. Being major seed predators, changes in the abundance of rodents could affect the long term vegetation dynamics of their habitat. Being a major preybase, changes in the abundance of small mammals could affect the communities of avian, reptilian and mammalian predators. Rodents are also major carriers of several pathogens and parasites. The invasion of several generalist species into forest fragments might introduce pathogens and parasites to which the native species might be ill adapted. Thus, changes in the small mammal community due to forest fragmentation can have a cascading effect on the ecosystem.

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1. INTRODUCTION AND OBJECTIVES

1. 1 INTRODUCTION

1. 1. 1 Tropical Forests

Tropical rain forests are rich in faunal and floral species (Prance 1988; Myers 1988; Terborgh 1986; Richards 1996; Whitmore 1997). The factors which contribute to such rich diversity have been a subject of considerable debate among ecologists. Several hypotheses relate this diversity to the stability, disturbance, latitudinal, geological and environmental gradients, woody nature of plants, high rainfall and solar radiation (Richards 1996). In general, the tropical rain forests lie below an altitude of 1300 m. Latitudinal range of rain forests is between 28° N in southern China and at 34-47° S in New Zealand, where it is often referred to as subtropical rain forest (Richards 1996). Tropical rain forests are generally found in the ever-wet tropics, where there is only minimal seasonal water shortage. They consist mainly of broad-leaved tall evergreen trees with characteristic buttresses, cauliflory, thick stemmed lianas, and mostly herbaceous epiphytes (Richards 1992, 1996; Whitmore 1997).

Rain forests occupy lowland areas with an annual rainfall ranging from 1800 mm in the rain forest of Amazon to as high as 9000 mm in Choco forest in South America (Prance 1988; Richards 1992, 1996; Whitmore 1997). The mountain tops and cliffs are occupied by the characteristic wet montane forests (Richards 1996).

The tropical rain forests occur in three major geographical zones: a) the South American rain forest comprises nearly 60% of world's rain forests. It is found predominantly in the Amazon basin (Richards 1996; Whitmore 1997); b) the Indo-Malayan (Thailand, Myanmar, South China, Malaysia, Indonesia, Philippines, south-west Sri Lanka, the Andaman and Nicobar Islands, north-east India and the Western Ghats of India) rain forests make up the second largest percentage of the world's rain forests; and c) the African rain forest, centred in the Congo (Zaïre) basin and continuing into Gabon, Cameroon and extending with a narrow belt along the north coasts of the gulf of Guinea and Uganda, is the least extensive and poorest in species (Richards 1996; Whitmore 1997). Apart from these three major rain forest regions, there are small patches on the east coast of Madagascar, and the islands of Mauritius, Reunion, Rodriguez and the Seychelles. However, these do not extend very far (Whitmore 1997). The subtropical rain forest reaches as far as down the eastern seaboard of Australia (through Papua and New Guinea) and northern New South Wales (Richards 1992; Whitmore 1997).

1. 1. 2 Loss of World's Tropical Forest

The tropical rain forests are being extensively depleted due to the extension and intensification of human land requirements (Myers 1988; Richards 1992; Whitmore 1997). In 1990, tropical wet evergreen forests covered about eight million km^2 (Myers 1991). Recent estimates have reported an annual loss between 160,000 and 200,000 km^2 suggesting that the annual depletion rate has doubled, compared to 1980 (WRI 1990). It is predicted that at this rate of forest depletion in the next 33 years, 90% of forest cover could possibly be removed (Terborgh 1992).

1. 2 THE SOUTH INDIAN RAIN FOREST

1.2.1 Distribution

In the Asian continent, the rain forest distribution is confined to Vietnam, Laos, Cambodia, the southern part of Thailand, Myanmar, South China, Malaysia, Indonesia, the Philippines, south-west Sri Lanka, the Andaman and Nicobar islands, north-east India and the Western Ghats of India (Nair 1991; Whitmore 1997). The south Indian rain forest occurs along the 1600 km long Western Ghats which runs from the southern tip of Tamil Nadu to the southern part of Gujarat (from 8° N to 21° N), covering 160,000 km^2 in six states: Kerala, Tamil Nadu, Karnataka, Goa, Maharashtra and Gujarat. The altitude ranges from mean sea level to 2695 m (Anaimudi), with 14 peaks >2000 m and a 30 km wide gap which is popularly known as the Palghat Gap (Pascal 1988; Nair 1991). The monsoon winds from the south-west are responsible for most of the seasonal rainfall. The western slopes of the Ghats receive very heavy rainfall, ranging from 2000 mm to 6000 mm per year. Rain decreases as the hill range extends eastward; the eastern slopes and foothill are almost a rain shadow area (Pascal 1988; Nair 1991).

The peak rainy season along the western face of the Ghats is from June to September (south-west monsoon) and from October to December (north-east monsoon) along the eastern slope. The Western Ghat tropical rain forests are at present mostly confined to the hill tops and plateaux, reaching 2000 m or more in elevation (Pascal 1988; Nair 1991).

1. 2. 2 Geology of the Western Ghats

The Western Ghats has three major rock formations. The Pre-Cambrian rock regions, considered to be the oldest rocks in India, are found predominantly in the eastern most part of Karnataka, Maharashtra and the northern part of Goa (Pascal 1988). Dark or light ferro-magnesian rocks, generally known as Charnokite rocks, are found in the eastern part of Tamil Nadu and Andhra Pradesh, and comprise a major portion of the Nilgiris and Palani hills (Mathur 1991). In coastal Kerala, recent sedimentary rocks are found in shallow water littoral deposits. These deposits are believed to have been formed during the Miocene, while the `Quilon beds' sediments and the `Warkali beds' sediments were formed in Mio-Pliocene age (Pascal 1988). During the late Tertiary Period and into the Quaternary, the western part of the peninsula was uplifted and several small and medium faults formed, forming the Western Ghats as they are today (Pascal 1988; Mathur 1991).

During the Tertiary period (70 to 1.8 million years ago), some floral families seem to have evolved exclusively on the Gondwanaland (region of land mass of three different continents Asia, Africa and South America before continental drift moved these land masses apart). Miocene period fossils (25 to 11 million years ago) reveal that regions of Gondwanaland were mainly occupied by tropical evergreen forests (Meher-Homji 1989). Impact and upliftment of the Himalayas during the mid-Miocene to the Pliocene Period was the primary cause of climate change in the Western Ghats during the Pliocene. As a result of this climate change, the dry season increased, causing evergreen species to be replaced with deciduous species (Meher-Homji 1989).

Till recently (late Pliocene) much of peninsular India had a humid climate and was covered with tropical rain forest. Pleistocene glacial and inter glacial periods saw the repeated withdrawal and advancement of grassland and rain forest in south India (Sukumar *et al.* 1993). However, the upliftment of the Himalaya and Western Ghats set on the monsoon climate. This resulted in the loss or regression of tropical rain forest in south India to the Western Ghats, only where conditions persisted for such a vegetation.

1. 2. 3 Species Richness and Endemism in the Western Ghats

The Western Ghats is one of the 18 biodiversity hot spots, covering only 5% of India's land but with 30% of the species. Most of the endemic fauna of Western Ghats is largely of Pre-Himalayan origin and comprises older taxonomic groups such as lower vertebrates, especially amphibians and reptiles. These oldest characteristic faunas of the Indian peninsula are referred to as 'Gondwana faunal derivatives' and mainly include peninsular autochthonous endemics (Mani 1974). Due to continental drift, they are presently confined to the Indian peninsula and their close allies are found in Madagascar (Lemurian fauna), South Africa (Ethiopian fauna) and rare forms in South America (Mani 1974).

In general, the species richness and rate of endemism in the Western Ghats increases as the taxonomic order decreases. For instance, in India nearly 1501 butterflies are reported, of which 330 are from South India and nearly 18% of these are endemic (Gaonkar in press.). Endemism is particularly high in the lower vertebrates, especially amphibians and reptiles (Inger and Dutta 1986). In India nearly 216 species of amphibians, including anurans and caecilians have been reported, of which 50% are endemic (Inger and Dutta

1986). The highest concentrations of endemic amphibians (93 species) are found in the Western Ghats. Out of 19 species of Gymnophiona from India, 17 are from the Western Ghats. Among reptiles, out of about 487 species from India, about 190 are from the Western Ghats, and nearly 50% of these are endemic (Kumar *et al.* 1998). About 245 species of fishes have been reported from the Western Ghats, of which 103 species (42%) are endemic (Kumar *et al.* 1998). The flora of the Western Ghats, especially the flowering plants of wet evergreen forest are species rich in endemics. For instance, out of about 5,000 species of flowering plants, nearly 1, 400 are endemic from 58 plant genera, out of which 42 are monotypic. Endemism is high among tree species with 308 out of 490 species being endemics.

Endemism is only about 10% among mammals. The Order Rodentia is relatively species poor in the Western Ghats. The endemics are the spiny dormouse (*Platacanthomys lasiurus*), jungle striped squirrel (*Funambulus tristriatus*), and the Travancore flying squirrel (*Petinomys fuscocapillus*); at the subspecies level the dusky striped palm squirrel (*F. sublineatus*), Layard's striped squirrel (*F. layardi*), grizzled giant squirrel (*Ratufa macroura*) and the white-bellied rat (*Rattus rattus wroughtoni*) are endemics. The Order Insectivora is also species poor in the Western Ghats with only five out of 29 species reported from India (Kumar *et al.* 1998). Other endemic mammals are the Malabar civet (*Viverra civettina*), the brown palm civet (*Paradoxurus jerdoni*), the lion-tailed macaque (*Macaca silenus*), the Nilgiri langur (*Trachypithecus johnii*), and the Nilgiri tahr (*Hemitragus hylocrius*), and a subspecies of the brown mongoose (*Herpestis fuscus fuscus*), stripenecked mongoose (*H. vitticollis vitticollis*), and the Nilgiri marten (*Martes gwatkinsi*). The Nilgiri tahr is the only large endemic mammal in the Western Ghats.

The high species richness and endemism in the Western Ghats are largely due to two factors, its geological history and the vastness of the rain forest. The former includes the continental drift; the collision of peninsular India with the Eurasian plate which formed the Himalayas. This left the Western Ghats insulated and isolated through delimitation by the sea to the west, the Vindhya and Satpura ranges to the north, and the arid Deccan plateau to the east (Nair 1991). Within the Western Ghats, the vast stretch of rain forest offered a wide range of suitable niches favourable to speciation through species specific habitat selection and landscape isolation (Mani 1974).

1. 2. 4 The Loss and Fragmentation of Rain Forest in the Western Ghats

Much of India's forest is undergoing large scale depletion due to population pressure (Myers 1980). Large scale deforestation began in the Western Ghats during the 1860's, with the establishment of tea, coffee, and cardamom estates. Later such forest exploitation was intensified for the construction of roads, hydro electric projects and railway and for monoculture plantations such as teak and eucalyptus (Pascal 1988; Nair 1991). About 40% of the forest cover in the Western Ghats was lost due to aforesaid activities between 1920 and 1990 (Menon and Bawa 1997). In Kerala, the annual forest loss was estimated at 0.27% between 1905 and 1965, while the loss was 0.6% during 1965 and 1973 (Chattopadhyay 1985).

According to Pascal (1988), among the wet evergreen forest patches in the Western Ghats only a few are more than 200 km^2 in area. These areas include Agasthyamalai Hills, Cardamom Hills, Silent Valley-New Amarambalam Forests and southern parts of the

Dakshin Kannada District in Karnataka State. Currently, there are several proposed hydel projects that could potentially submerge some of the best lowland wet rain forests, for instance Pooyamkutty Hydel Project in Kerala. The extension of roads across the forests can also heighten the speed of fragmentation (*e.g.*, a proposed metal road from Papanasam to Tiruvananthapuram across Agasthyamalai Hills and the railroad to Sabarimala Temple in Kerala).

The habitat loss described above leads in due course to habitat fragmentation. Apart from the anthropogenic pressures that have caused the subdivision of contiguous habitats into several small fragments (Lord and Norton 1990; Saunders *et al.* 1991b), on a small scale habitat fragmentation also occurs due to natural processes such as fire and windfalls (Pickett and Thompson 1978). According to Nair (1991), forest plantations such as teak often cause a serious fire hazard which substantially contributes to further forest fragmentation (see also Hill and Read 1984).

1. 2. 5 The Impacts of Forest Fragmentation

Several processes lead to species loss due to forest fragmentation. However, forest fragmentation has two major impacts: those that result from the isolation of a contiguous population into several small populations, and the progressive degradation of forest fragments through several path ways such as changes in micro and macro-climates, changes in vegetation structure, invasion by non-native species, and changes in the matrix surrounding the fragment.

The small populations are vulnerable to extinction from demographic and environmental stochasticity, and loss of genetic variability and pathogenic diseases (Gilpin and Soulé 1986; Quinn and Hastings 1987; Saltz 1996). Pathogens and parasites that are transmitted by invading animals into the fragments often determine the survival of the isolated small populations (Kozakiewicz 1993; Loye and Carrol 1995; McCallum and Dobson 1995; Combes 1996). Small populations are vulnerable to extinction from inbreeding (Simberloff and Abele 1982) which results in decreased variability in the gene pool (Bennett 1990; Kozakiewicz and Konopka 1991; Kattan *et al.* 1994) and produces undesirable characteristics in the offspring. Uninhabitable matrices surrounding the fragments act as barriers to the dispersal of individuals from animal communities living in the fragment (Kozakiewicz and Jurasińska 1989; Bennett 1990; Hobbs 1993b; Szacki *et al.* 1993; Fahrig and Merriam 1994).

Changes in the micro and macro habitats are perhaps the most obvious effect of fragmentation (see Saunders *et al.* 1991b; Hobbs 1993a; Saunders *et al.*1993). Changes in the microclimate that cause physical modification of a habitat include altered radiation fluxes and wind patterns, water run off, soil moisture and atmospheric humidity. The removal of vegetation, along with either the subsequent replacement or regrowth of secondary successional species, will increase the amount of solar radiation reaching the ground. Furthermore, an increase in night-radiation of heat changes the energy balance of a fragmented landscape (Saunders *et al.* 1991b). Hobbs (1993a) reported that the alteration of native vegetation reduces evapo-transpiration, while the altered soil encourages heavy water flow. The result is a rise in the water table, bringing salinity to the surface. These changes, coupled with altitude and landscape position, contribute to an `edge effect' (see Lovejoy *et al.*1986; Laurance 1991; Fiedler 1993). These factors together with the surrounding vegetation, replace the native vegetation along the forest edges further to the interior of the fragment. This leads to loss of native species (see Laurance 1991; Young and Mitchell 1994; Mills 1995). In Amazon, changes were observed in the relative humidity and air temperatures within a few years of forest fragmentation (Lovejoy *et al.* 1986; Kapos 1989).

Animals with higher dispersal capabilities often survive for longer periods of time in a fragmented landscape, while animals with a higher habitat preference and sensitivity to changes in the environment often fail to survive (Karr 1982; Laurance 1994). In addition to this, the influx of non-native animals and plants from the secondary forests often replaces many native species. The secondary forest animals are usually well adapted to the changes in the isolates than are the native animals (Lovejoy *et al.* 1986). The invading species often also include predators (Johns 1985; Wilcove *et al.* 1986), brood parasites (Robinson *et al.* 1992; Haskell 1995) and pathogens (Saltz 1996) to which the native species are not adapted. Altered macro habitats, such as greater canopy opening, and greater stress, for example due to food shortage could further enhance the susceptibility to predation and pathogens. This could further shift the balance in favor of the invading species which are in competition with native species.

The island biogeographic theory (MacArthur and Wilson, 1967), predicts that the more habitat islands are isolated, the less likely these will be colonized by new inhabitants. Initially, when habitat islands are formed `species compaction' increase the absolute number of species in the island. The rate of immigration of species will be high because any colonizing individual represents a species new to that island. As the number of resident species increases, the rate of immigration of new, unrepresented species declines. As a result, `faunal collapse' or extinction of insular species will occur. The extinction rate at this point is perhaps greater. Largely because of more number of species in the above stated conditions, competitive exclusion becomes more likely, and the population size of a species is on the average smaller, making them more vulnerable to extinction. However, this theory has been the subject of debate among ecologists, and it has been of little practical importance in terms of design and management of forest resources (Gilbert 1980; Zimmerman and Bierregaard 1986).

In a species-area comparison of the occurrence of bird species in fragmented and unfragmented chaparral habitats (Bolger *et al.* 1991), it was found that the rate of extinction of resident bird species was greater in the fragmented habitat than in the unfragmented habitat. In addition, the more abundant species persisted longer in the fragments. Cook (1995) explained the relationships between nested subset and species diversity in fragmented and unfragmented forests. He pointed out that under conditions of close nestedness, a strong overlap is found in species composition among sites; smaller areas tend to contain only a few species while larger areas have new species in addition to the species in smaller areas.

1. 2. 6 Community Structure

A community is defined in terms of various interactions of biotic and abiotic factors in the environment (Andrewartha and Birch 1984), in which the position of a species and its niche mean more than just where it occurs, the habitat or community in which it may be found. A community structure also implies a consideration of the role a species actually plays within the community (Putman and Wrattan 1984). A community's complexity of structure can be seen by the identification of species guilds within a community (see Dueser and Shugart 1978, 1979; Adler 1988; Bowers and Flanagan 1988; Gore 1988; Canova and Fasola 1991; Woodman *et al.*1995). Extreme ecological similarity and competition for the same resources lead those species to essentially evolve different specializations to meet their needs, thereby reducing competition (Putman and Wrattan 1984). The relative abundance of individuals of a species may represent a particular influence on the nature and function of the community (Putman and Wrattan, 1984). Species-abundance relationships in a community are affected not only by the actual relationships between species, but by the relative number of organisms of those different species. Leigh *et al.* (1982), suggested several factors that affect community structure. Even though his work was based on plant communities, these factors are also applicable to animal communities. The important factors are: a) heterogeneity of habitat, b) environmental variability in time, c) environmental variability produced by disturbances to the uniform structure of the forest, and d) rate of production (immigration or speciation) of competitively equivalent species. The relative importance of microhabitat requirements and competition in shaping community structure has been considerably debated for several decades (Bowers and Flanagan 1988; Gore 1988;

Canova and Fasola 1991). Both the factors are perhaps important, their relative importance depending on the particular ecological context (Laurance 1994; Shanker 1998).

The impact of isolation on the community structure may depend on fragment area, since the local extinction of a species is dependent on both the size of the local population and the area it occupies (Begon *et al.* 1990; Fiedler 1993). However, Andrén (1994) has argued that size and isolation are not the only factors that affect community structure, the habitat structure and surrounding matrix also playing a considerable role in structuring community. For instance, the isolated fragments are also a part of the surviving landscape mosaic, to which species respond differently, the generalists being more adapted to it. Under these circumstances, the forest dependent species are often replaced by commensal species (Ricklefs 1990; Kattan *et al.*1994). This generalized occupancy of commensal species often leads to local extinction of small population in fragmented landscapes (Andrén 1994).

Area of the fragment is nonetheless an important factor in determining the occurrence of many, especially larger or specialist species. For example, the diversity and relative abundance of temperate birds in Chile were associated to fragment area (Wilson *et al.* 1994). Newmark (1991) has reported from the Usambara mountains of Tanzania, that the extinction of birds from fragments was related to rarity. However, for many species it is habitat structure and the surrounding matrix is important. In Amazon, Bierregaard *et al.* (1992) found that primate communities are sensitive to fragment area, whereas the small bodied mammals and lepidoptera were hyper-sensitive to the vegetation structure in and around the fragments. The disappearance of the dormouse (*Muscardinus avellanarius*) from forest fragments in the United Kingdom, is largely due to habitat degradation and microhabitat loss rather than to fragment area (Bright and Morris 1996). However, from Australia, Laurance (1994) reported that the forest dependent species were caught only from unfragmented forest.

Invading habitat generalists are often introduced by human. The invasion of these commensal species into fragments is a major consequence of habitat fragmentation (Kattan *et al.* 1994). The invaders can also change the species composition by bringing new parasites and pathogens to which resident species are more vulnerable (McCallum and Dobson 1985).

1. 2. 7 Studies on Rodents and Shrews

There have been several studies in India on rodent pest management (Barnett and Prakash 1975; Bhat 1992). However, ecological studies of this taxa and shrews in India have been relatively few and mostly restricted to the arid and semi arid areas have addressed burrow structure (Ghosh 1975; Goyal 1982; Goyal and Ghosh 1993), population ecology (Prakash 1975), niche alteration (Prakash in press) and community ecology (Prakash *et al.* 1995). Ecological studies of this taxa in southern India have been mostly limited to population change and reproductive biology in cultivated landscapes (*e.g.,* Rajagopalan 1972; Chandrahas 1974; Balakrishnan and Alexander 1977; Bhat and Mathew 1984; Bhat and Sujatha 1986; Balakrishnan 1987; Bhat *et al.* 1987).

Until recently studies on terrestrial rodents and shrews have not been attempted in the rain forest in India. Chandrasekar-Rao and Sunquist (1996) conducted live trapping in three vegetation types the Indira Gandhi Wildlife Sanctuary. Among the five species captured, *R. r. wroughtoni* was the most dominant in the natural forest while a commensal *M. platythrix* was the most dominant in teak plantations. Overall species richness and biomass were greatest in the moist deciduous forest. In the same sanctuary, the very large and large fragments, where disturbance was comparatively low, had all five species of diurnal arboreal mammals, while small and medium fragments had only one or two species each (Kumar *et al.* 1995; Umapathy 1998). The abundance of the Malabar giant squirrel (*Ratufa indica*) and two species of flying squirrels seemed to increase with disturbance. In the montane shola fragments in Nilgiri Hills, there was a positive relationship between species number and fragment size. Forest-specific species were trapped from large contiguous forest (Shanker 1998). In the relatively undisturbed and contiguous rain forest in Kalakkad-Mundanthurai Tiger Reserve, among five species of trapped rodents and shrews, the *R. r. wroughtoni* was the most abundant (Anon. 1997).

Outside of India, there have been several studies on the community structure and micro habitat preferences of small mammals, and the impact of forest fragmentation; in Australia (Laurance 1991; Saunders *et al.* 1991b; Saunders and Hobbs 1991a; Arnold *et al.* 1995; Monamy 1995; Smith and Quin 1996), Amazon (Lovejoy *et al.* 1986; Nitikman and Mares 1987; Malcolm 1991 Bieregaard *et al.* 1992; Stallings 1988 and others), temperate countries (Dueser and Shugart 1978, 1979; Bujalka 1986; see USDA 1988; Kozakiewicz and Jurasińska 1989; Jurgiel 1992; Kozakiewicz 1993) and Southeast Asia (Kemper and Bell 1985; Walker and Rabinowitz 1992)

In Amazon, Nitikman and Mares (1987) reported several rodents as being habitat specialists while the one species of marsupials (*Marmosa agilis*) was trapped randomly in all microhabitat types. In Australia, the velvet-furred rat (*Rattus lutreolus velutinus*) was trapped throughout the study period in sites with dense ground cover (Monamy 1995). Iyawe (1989) pointed out that in Ogba Forest Reserve in Madagascar the species richness was greater in primary forest than secondary forest. Kemper and Bell (1985) found that in a lowland rain forest of Malaysia there were no differences in trapping success between regenerating sites and primary forest sites. However, vegetation cover and under storey cover influenced the number of species trapped. Rats and mice were captured often in dense tree stands and tall grass in Western Mauna Kea, Hawaii (Amarasekare 1994). In England, the occurrence of grey squirrel was more likely when the forest fragments were larger and closer to other woodlands (Churchfield and Brown 1997). The harvest mouse and the wood mouse were habitat specialists and showed a high preference to habitats, the former to sites with abundant forbs and the latter to sites with small trees and shrubs. However, a few generalist species of shrews and *Microtus agretis* showed no clear habitat preferences (*op.cit.*).

Saunders *et al.* (1993) found that in Australian rain forests intensive clearings and the introduction of exotic species heightened the loss of native flora and fauna, while cattle grazing and other disturbances amplified the rate of species extinction. In some areas, edge or disturbance-tolerant species increased in fragmented forests, while in unfragmented forest the assemblages of ecologically similar communities were more structured (Laurance 1994). The elephant shrew's distribution and abundance in a Kenyan coastal forest suggested that the elephant shrew (*Rhynchocyon chrysopygus*) was not essentially a forest-dependent species as it was also found in scrub and degraded habitats (FitzGibbon 1994). However, this species was considered to be threatened due to its restricted range, small isolated populations and an indiscriminate loss of habitats.

Seasonal abundances, dietary habit, body weight and age/sex composition of terrestrial small mammals have been extensively studied elsewhere in the world. For instance, Stephanson (1994) observed the highest species richness in Madagascar during summer when all species were active. Adler (1994) found that densities were lowest during the early rainy season and highest during late rainy season. In addition, the proportion of adults peaked in mid-season but were lower in the rainy season. There have been several studies on community structure in small mammals in the temperate regions (Bujalka 1986; Smith and Quin 1996; USDA 1988). Bujalka (1986) found that in the bank vole (*Clethrionomys glareolus*) in northeastern Poland, adult females were more common than adult males, due to a lower maturation rate than females. Studies on feeding habits of small mammals conducted by Varty (1990) in Somalia revealed that all the trapped rodents heavily relied on seeds and that their dietary overlap was high. Smith and Quin (1996) hypothesized that body size and rate of reproduction were important determinants of the decline and extinction of Australian conilurine rodents. Additionally, small carnivores (especially feral cats) also played a decisive role in the decline of populations. The impact of introduced rodents (especially *Mus musculus* and *Rattus rattus*) which have invaded forest fragments, contiguous forests, and islands have been examined in Australia (Laurance 1994), Hawaii (Amarasekare 1994) and North America (Soulé *et al.* 1992). Even though these invading species now form substantial part of the small mammal biomass, their impacts on other small mammals are far from clear.

1.3. OBJECTIVES

A review of the studies on small mammals reveals that very few studies have addressed the ecology of rodents and shrews in India, even more so the impact of habitat fragmentation on them. Studies elsewhere have noted substantial changes in the community structure of small mammals in forest fragments, related to the nature of changes in the micro and macro habitats, the invasion of generalists and introduced species, and the biology and life history of the species concerned. The species-rich tropical rain forest of Western Ghats needs to be assessed in this context, since it has undergone extensive loss and fragmentation.

Rodents and shrews were selected as the study animals for several reasons. As major seed predators rodents play very important roles in the long term vegetation dynamics of their habitat. They are also a major prey base for many reptilian (*e.g.*, snakes), avian (*e.g.*, raptors) and mammalian (*e.g.*, small carnivores) predators. Any changes in the rodent and shrew community are thus likely to have a cascading impact on the ecosystem in which they live. Studies elsewhere have shown that rodents and shrews show two major impacts of habitat fragmentation, that of species loss and invasion. Finally, among mammals it is this taxa that have contributed most to our current understanding of community structure and processes.

This study was carried out with the following objectives:

- to evaluate the variation in community structure in rodents and shrews in relation to various features of the rain forest fragments;
- to examine the seasonal variation and synchrony among forest fragments in population abundance, age/sex ratio and body weight of rodents and shrews in relation to habitat features; and
- to identify species differences in microhabitat selection and to examine whether such differences explain the variation in community structure among forest fragments.

This thesis is divided into six Chapters including this Chapter. Chapter 2 gives an account of the study area, study animals and methods. Chapter 3 examines the community structure in rodents and shrews in seven forest fragments, in relation to several macro habitat features. Chapter 4 examines the variation and synchrony among forest fragments in several demographic parameters and attempts to identify factors that govern the variation and synchrony. Chapter 5 examines interspecific variation in microhabitat selection and the extent to which this can account for variation in community structure among forest fragments. The final Chapter presents a concluding discussion.

2. STUDY AREA, ANIMALS AND METHODS

2.1 STUDY AREA

2.2.1 Geography and Climate

This study was conducted in the Indira Gandhi Wildlife Sanctuary (previously Anaimalai Wildlife Sanctuary), Tamil Nadu, south India. Anaimalai Hills, at the 'T' function just south of the Palghat gap, was chosen as the study area. This area is a typical example of the extent of loss and fragmentation that tropical rainforest has been subjected to in the Western Ghats(Figure 2.1). In the Anaimalai Hill, the Indira Gandhi Wildlife Sanctuary was chosen as the study area. This sanctuary and adjoining private forests and estates now retain the remnant of what was once the most contiguous and extensive tropical rainforest in the Western Ghats. The landscape of this area consisting of several forest fragments, estates (tea, coffee and cardamom), plantations (teak, cinchona and eucalyptus), a chain of reservoirs, a vast network of roads, and several small to large human settlements was a typical representative of all the complexity within which impacts of forest fragmentation operated.

This sanctuary (10° 13' 2" - 10° 33' 3" N-76° 49' 3"-77° 21' 4" E), covering an area of 987 km^2, and is about 100 km south of Coimbatore City. Three major public roads pass through the Sanctuary; from Pollachi town to Parambiculam, Chalakkudi and Munnar. The Sanctuary, with Chinnar Wildlife Sanctuary and Eravikulam National Park in the south, Parambiculam Wildlife Sanctuary in southwest, Nelliampathy Reserve Forest and

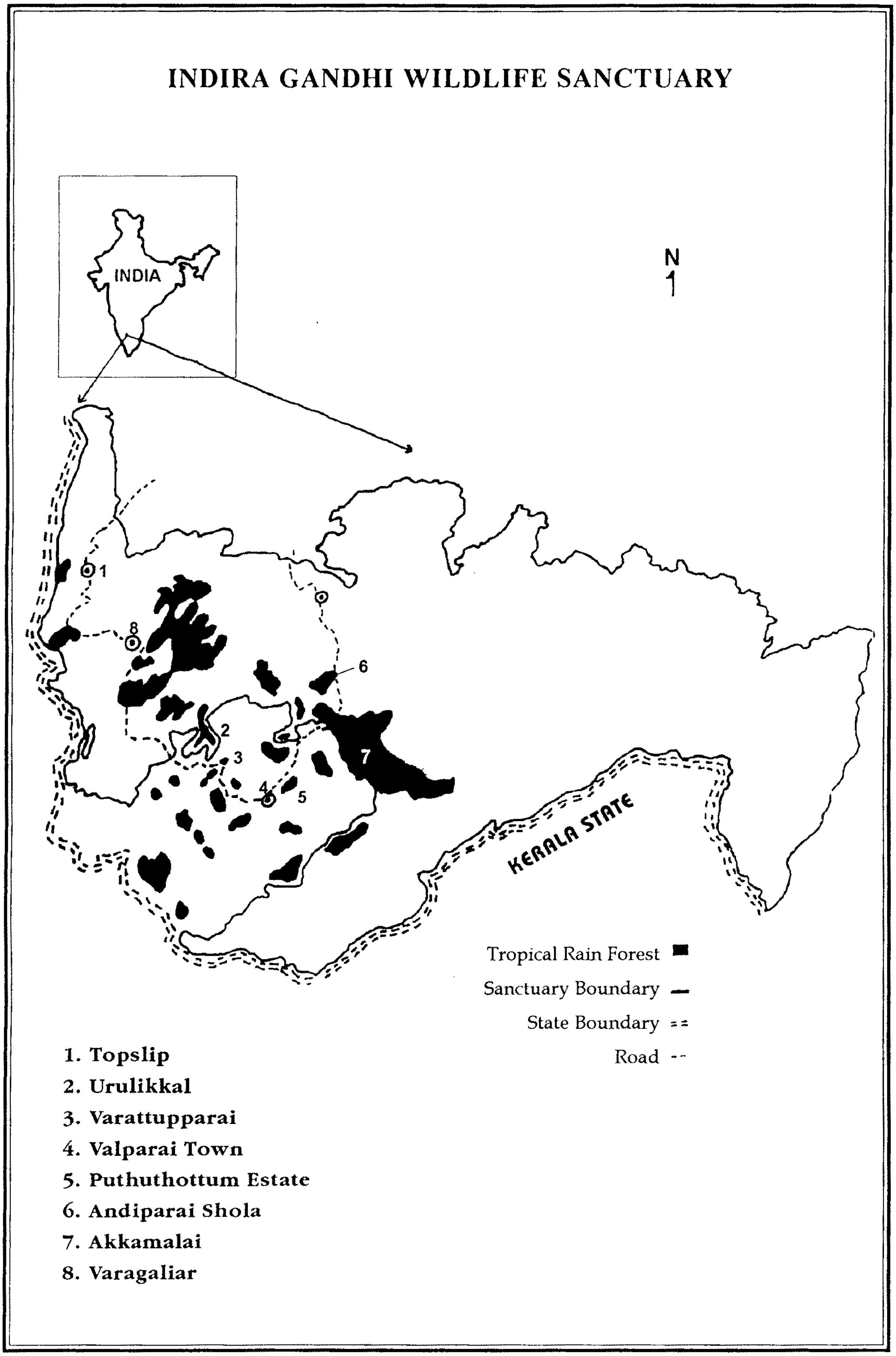

FIGURE 2.1 MAP OF THE INDIRA GANDHI WILDLIFE SANCTUARY SHOWING THE TROPICAL RAIN FOREST FRAGMENTS

Vazhachal Reserve Forest in the west and southwest, is an important large conservation area, especially for the large mammals. The area is drained by several rivers Chalakkudi, Sholayar and Idamalayar being the major west flowing rivers, and Aliyar being the major east flowing river.

In the central part of the Sanctuary there are about 130 km^2 of privately owned estates, mostly of tea, but also coffee, cardamom and cinchona. Most of the rain forest fragments in this area are privately owned. The Valparai township, catering to the estate workers, was at the centre of the study site. The field station was an accommodation provided by the Parambiculam-Aliyar Project of the Tamil Nadu Electricity Board. The rain forest fragments were within a 30 km radius of the town.

The altitude of the Sanctuary ranges from 220 m in the eastern forest hills to 2,513 m in the highest peak, Thanakkanmalai. Mountainous tracts of about 800 to 900 m cover nearly 90% of the Sanctuary. This mountainous tract and its orientation mediate the rainfall in the Sanctuary, from southwest and northeast monsoons. The eastern foothills are in the rain shadow area and receive only about 500 mm to 1,000 mm of rainfall. The mountains proper, especially the western parts, receive as much as 3,000 mm to 5,000 mm of rainfall. Most of the rainfall in the rain shadow area is received from the northeast or retreating monsoon, while the western mountainous tracts receive as much as 80% of the rainfall from the southwest monsoon.

Temperature regimes also show considerable variation within the Sanctuary. In the eastern slopes summer temperature could be as high as 40°C, and winter temperature as low as 15°C. In the mountainous tracts, summer temperature seldom exceeds 30°C, and winter temperature could be as low as 9°C. In the higher reaches above 2,000 m (e.g., Grass Hills) night time temperature in winter could be below freezing point. Taking temperature and rainfall regimes in the area, I divided the year in four seasons:

1. Summer (March-May): Relatively high temperature, and low rainfall, limited to a few summer showers.
2. Wet season I or Wet I (June -August): Period of very high rainfall, from the southwest monsoon.
3. Wet season II or Wet II (September-November): Period of high rainfall but often only in the afternoon, from northeast monsoon.
4. Winter (December-February): Relatively low temperature and rainfall.

2. 1. 2 Vegetation

Several vegetation types (Puri *et al.* 1983) occur in the study areas reflecting the altitudinal and rainfall regimes, excluding plantations. The dry thorn forests occur in the eastern parts and dry and moist deciduous forests occur at higher elevations. The predominant trees are *Terminalia paniculata, T.belerica, Tectona grandis, Lagerstroemia lanceolata, Grewia tiliaefolia* and *Cassia fistula*. The original vegetation above about 650 m was the luxuriant tropical rain forest. Predominant trees in this forest type are *Cullinea exarillata, Mesua ferrea, Pallaquim ellipticum, Listea insignis, Artocarpus heterophyllus, Ficus glomerata* and *Syzygium* spp. The high species richness and endemism typical of the Western Ghats, are in this vegetation type.

Above about 1,700 m, tropical montane forests occur interspersed with montane grass lands, forming the shola-grassland complex. In the study area, these extensive areas of the shola-grassland occur in the Grass Hills above 2,000 m; *Michelia nilagirica, Eugenia* spp. *Gordonia obtusa* and *Meliosma* spp. are predominant. These forests stand low with thick leaves; lianas, mosses, and epiphytes are scarce. This is contiguous with the shola-grasslands in the Eravikulam National Park. This unique formation of grassland with several montane rain forest fragments, is more than 20,000 years old, and not as a result of human activities (Sukumar *et al.* 1993). These forest fragments are therefore not considered in this study. The small mammals in this formation has been the subject of a recent study in the Nilgiri Hills (Shanker 1998). The major grass species are *Heteropogon, Cymbopogon* and *Abtuda.*

Extensive mono cultures are at present a major vegetation in the area. These include coffee, tea, cardamom, cinchona, eucalyptus and teak. These have been extensively cultivated since the early part of this century. These plantations were created after removal of an extensive area of deciduous forest and rain forest.

2. 1. 3 History of Human Disturbance in the Anaimalai Hills

The Anaimalai Hills was explored for the first time in the year 1864 by British planters. Congreve (1938) reported that he had seen many herds of gaur, deer, and Nilgiri tahr in the area, and that it was a mammoth task in the rain forests of the Anaimalais just to determine a direction in which to go. Virgin forests were given on lease in the 1860's to planters who soon began clearfelling the area for planting tea. Waverly Estate was the first to be established, where only a 180 ha forest patch was left, which is now called Andiparai shola. By the end of the 19th century, forest land had been leased to several planters, most of them Britishers. By 1930's nearly 130 km^2 had been planted with tea alone, with additional areas planted with coffee, eucalyptus and cardamom. Late 19th century also saw the plantation of teak for which also extensive areas were clearfelled. Teak plantations continued up to 1978, when all operations in rain forest were halted. In two decades from 1950's, some chains of reservoirs were built in the Anaimalai Hills, under the Parambiculam-Aliyar Project, leading to further loss of forests, and fragmentation of the remaining forests by extensive selective felling for timber.

2.1.4 Present Situation

The above-mentioned activities caused extensive loss and fragmentation of rain forest. Our initial surveys in the area were carried out between December 1993 and April 1994. These surveys revealed more than 25 rain forest fragments in a 30 km radius from Valparai town. Most of the small fragments are owned by private companies, such as Tata Tea, Peria Karamalai Tea company, NEPC and Parry Agro Ltd., and individuals. The larger fragments are owned by the forest department (Table 2.1). While the small fragments face a great deal of illegal tree felling and lopping, in the large and very large fragments these occur only in the edges. The vegetation around the fragments is coffee and tea, frequently interspersed with eucalyptus. Many of the fragments are underplanted with cardamom, and many also have labour settlements in close proximity.

2.1.5 Fauna

The study has a diverse assemblage of mammalian fauna. The Primate fauna includes the lion-tailed macaque (*Macaca silenus*), bonnet macaque (*M. radiata*), the Hanuman langur (*Semnopithecus entellus*), the Nilgiri langur (*Trachypithecus johnii*) and the slender loris (*Loris tardigradus*) are inhabitants in different altitudinal ranges (Singh *et al.* 1997). The study area has large population of the Asian elephant *(Elephas maximus*), gaur (*Bos gaurus*), sambar (*Cervus unicolor*) and the spotted deer (*Axis axis*). At higher elevations the endemic Nilgiri tahr (*Hemitragus hylocrius*) is found. Mammalian carnivores include the tiger (*Panthera tigris*), the leopard (*P. pardus*), the wild dog (*Cuon alpinus*) and sloth bear (*Melursus ursinus*). Small carnivores include the jungle cat (*Felis chaus*), the fishing cat (*F. viverrina*), the leopard cat (*F. bengalensis*), the rusty spotted cat (*F. rubiginosa*), the brown palm civet (*Paradoxurus jerdoni*), the common palm civet (P. *hermaphroditus*), the small Indian civet (*Viverricula indica*), the Nilgiri marten (*Martes gwatkinsi*), the brown mongoose (*Herpestes fuscus*), the stripe necked mongoose (*H. vitticollis*), the common mongoose (*H. edwardsii*) and ruddy mongoose (*H. smithi*).

The rodents include two species of flying squirrels, the endemic Travancore flying squirrel (*Petinomys fuscocapillus*) and the large brown flying squirrel (*Petaurista petaurista*), the grizzled giant squirrel (*Ratufa macroura*), the Malabar giant squirrel (*R. indica*), the dusky striped squirrel (*Funambulus sublineatus*), the jungle striped squirrel (*F. tristriatus*), the Layard's squirrel (*F. layardi*), one species of endemic dormouse (*Platacanthomys lasiurus*) and several species of murids. The lower vertebrates and invertebrates in the area have not been studied in recent years, even though species richness and endemism in this taxa are expected to be very high.

2.2 STUDY ANIMALS

2. 2. 1 Order Rodentia

Rodentia constitutes the single largest mammalian Order with by far the greatest number of species (Ellerman 1961; Anon. 1990). They are distributed throughout the world, with more than 1,700 species, and are naturally absent only from the Antarctic region, New Zealand, and a few oceanic islands (Anon. 1990). However, in New Zealand commensal rats (*Rattus rattus norvegicus*) have been introduced. Worldwide, there are nearly 1,250 species of rats and mice belonging to 280 genera in ten families, and 17 species of dormice and related species from nine genera (Anon. 1990). Rodents occupy niches from the arctic tundra to tropical rain forests, and have adapted to all sorts of habits and habitats (Dieterlen 1989; Anon. 1990). With 25% of the species, tropical rain forests are centers of rodent distribution (Dieterlen 1989). They live solitarily, in pairs or groups, and are crepuscular, diurnal or nocturnal. Their food habits range widely, from vegetable matter to insects and vertebrates, but are predominantly vegetarian (Anon. 1990).

The rodent fauna of the Indian region is highly impoverished (Dieterlen 1989) compared to other regions. In fact, Rodentia is only the second most speciose Order (with about 100 species) in India, next to Order Chiroptera (bats) with about 110 species. Rodents in India are represented by seven families, 39 genera and 99 species, excluding the little known Andaman and Nicobar Islands species. Faunal impoverishment is particularly true for the Western Ghats, with probably less than 30 species present there. In the south Indian region, 24 species belonging to four families have been reported (Ellerman 1961;

Sterndale 1982; Kumar *et al.* 1988) these include eight species of squirrels (Sciuridae), 14 species of rats and mice (Muridae), one species of dormouse (Muscardinidae) and one species of porcupine (Hystricidae) (see Appendix I). Our present knowledge of the taxonomy and distribution of Indian rodents are due to Ghose (1964), Chaturvedi (1966), Biswas and Tiwari (1966), Agrawal and Prakash (1992) and Prakash *et al.* (1995).

2. 2. 2 Order Insectivora

Insectivores are believed to be the most primitive living members of the Class Mammalia. Worldwide, the Order Insectivora comprises six families, with nearly 360 species (Anon. 1990). Four families have been reported from India: Talpidae (two species of moles from one genus), Soricidae (six genera with 22 species of ground shrews), Erinaceidae (one genus with six species of hedgehogs), and Tupaiidae (two species of tree shrews belonging to one genus) (Sterndale 1982; Kumar *et al.* 1998). They are solitary, and except for the tree shrews, nocturnal (Sterndale 1982; Anon. 1990). They live in diverse habitat types, ranging from arid regions to wet and dry forests, grass lands, farm lands and with species the grey musk shrew (*Suncus murinus*) is very common in human settlements (Anon. 1990). Five species of ground shrews from one genus (Soricidae) have been reported from south India (Sterndale 1982; Kumar *et al.* 1998), though little is known about their ecology and taxonomy. Shrews feed primarily on insects; sometimes they supplement this diet with plant matter (Anon. 1990). The distribution of individual species with their common name is listed (see Appendix I). The species of rodents and shrews that were trapped during the study are described below.

2. 2. 3 Accounts of Species Trapped in the Study

2.2.3.1 Family: Muridae

The White-bellied Rat: *Rattus rattus wroughtoni* (Hinton)

The common house rat or black rat (*Rattus rattus*) is the most common species in the order Rodentia having been introduced by man in most of its current range. Its coat colour and habits vary considerably depending upon the climate and habitat in which it lives (Ellerman 1961; Prater 1971). In peninsular India, the dingy bellied brown rat (*R. r. norvegicus*) is largely confined to houses (Ellerman 1961). It causes a great deal of damage to stored grains and standing crops, and is thus, considered a pest. A number of studies on this species have been carried out for pest management (Barnett and Prakash 1975; Prakash 1976; Sood and Guraya 1976; Doharay *et al.* 1980; Srivastava *et al.* 1980; Bhat and Sujatha 1986). Among the subspecies, the white-bellied rat (*R. rattus wroughtoni*) occurs in the wild and as a pest in crop field throughout southern India (Ellerman1961; Bhat and Sujatha 1986; Chandrasekar-Rao and Sunquist 1996; Anon. 1997; Shanker 1998). The dorsal fur is brownish black, the belly is white, and tail length is equal to its head and body length. It has been reported to breed throughout the year, with a peak in February-March and July-August (Bhat *et al.* 1987). This is perhaps the most dominant murid species in the natural forests of the Western Ghats (Chandrasekar-Rao and Sunquist 1996; Anon. 1997; Shanker 1998).

The White-tailed Wood Rat: *R. blanfordi* (=*Cremnomys*) *blanfordi* (Thomas, 1881)

The white-tailed wood rat is found throughout the peninsular India and Sri Lanka. In southern India it lives mostly in the wild, and is arboreal in the forest (Ellerman 1961; Prater 1971; Meena 1997; Shanker 1998). In open treeless country it has adapted to scrub and bush land habitats (Prater 1971). *R. blanfordi* can be identified by its distinctive long white tail. The tail is brownish black for 10 cm from the base and then white to the tip, which has long hairs. The dorsal fur is brownish in colour and the ventral fur is white. No study has been carried out on this rat in peninsular India either in the wild or in the laboratory.

The Indian Bush-rat: *Gollunda ellioti* (Gray, 1837)

It is common in Sri Lanka, peninsular India and the arid and semi-arid regions of central and western India (Prakash *et al.*1995). The Indian bush-rat lives mostly in the wild, but may venture into cultivated lands. However, its favourite habitat is bush and scrub jungle. Jain (1985) made a few observations on feeding, breeding and cannibalistic behaviour of this rat. Prakash *et al.* (1995) reported that in Aravalli range of Rajasthan this rat typically inhabits bushy undergrowth in order to avoid predation. It is a small rat whose distinctive features are short, rounded head and ears, and a rather hairy tail. The upper coat is yellowish brown speckled with black. The tail is shorter than the head and body length (Ellerman 1961; Prater 1971).

The Indian Long-tailed Tree Mouse: *Vandeleuria oleracea* (Bennett, 1832)

This is a very small mouse that is found throughout India (Ellerman 1961). The tail is much longer than the head and body. The fur is soft and the under part is white. It is completely arboreal in nature (Bhat and Sujatha 1986). The first and fifth toes on all four feet are partially opposable with flat nails instead of claws (Ellerman 1961). Feeding behaviour of this mouse had been studied by Sharma (1995). Jain (1985) reported that its range extends to Madhya Pradesh. This species has been reported even from the higher altitudes of the upper Nilgiris (Shanker 1998).

The House Mouse: *Mus musculus* (Linnaeus, 1758)

The house mouse is believed to be of Asiatic origin (Ellerman 1961) but has been introduced by man world wide. It closely resembles the common house rat, but is smaller in size. The dorsal coat colour varies from dark to light brown and is paler below. Tail length usually exceeds the head and body length.

The Common Indian Field-Mouse: *Mus booduga* (Gray, 1837)

This is one of the most common field mice known from India (Ellerman 1961). It is considerably smaller than *M.musculus*. The dorsal pelage varies from light to dark brown. The underpart is white or whitish grey, sometimes quite dull grey, but apparently both types occur together in many localities. It is common in the wild, in gardens, and sometimes even ventures into houses. An assessment has been made on this rodent's abundance for pest management by Karim (1994). It has been reported to breed through most of the year, with a peak in October-November (Chandrahas 1974). It has been capable of circulating Kysanore Forest Disease (Krishnaswami 1965).

Several other murids occur at the foothills of the Western Ghats, and probably in human settlements of the Western Ghats proper. These include the Indian gerbil, lesser bandicoot rat, the metad and the spiny field mouse (see Appendix I).

2.2.3.2 Family: Muscardinidae

The Malabar Spiny Dormouse: *Platacanthomys lasiurus* (Blyth, 1859)

The Malabar Spiny Dormouse is endemic to southern India, and inhabits the wooded rocky hills (Anon. 1990). It was thought that this dormouse was confined to an altitude below 900 m (Ellerman 1961; Jayson and Christopher 1995), but recently it has been reported from the higher altitudes (1,200-2,000 m) of Nilgiris, Kalakkad-Mundanthurai Tiger Reserve and Anaimalai hills (Shanker 1996; Anon 1997; Prabhakar 1997). It has flat spiny fur on the back extending to the head. The ears are large and prominent. The tail is fully haired and is bushy, as the squirrel's, but the hairs gradually increase in length towards the end of the tail. It often ventures into pepper and arecanut plantations where it causes considerable damage (Rajagopalan 1968; Jayson and Christopher 1995), giving it the name of `pepper rat' (Dieterlen 1989).

2.2.3.3 Family: Soricidae

The Nilgiri Wood Shrew: *Sorex niger* (=*Suncus*) *niger (Jerdon)*

This shrew is characterised by a long pointed snout, projecting considerably beyond the upperlip, small eyes, rounded ears, body covered with soft fur, and feet and a tail that is sparsely clad with hair (Sterndale 1982; Anon. 1990). The feet are five-toed, separate, and not webbed like in the moles (Sterndale 1982). The two front teeth are distinguishably different from the rest of the dentition (Sterndale 1982). Two sub species have been reported, *S. n. niger* from upper Nilgiris and *S. n. malabaricus* from lower altitudes.

In the upper Nilgiris, Shanker (1998) trapped another endemic ground shrew, *Suncus montanus*. The most widely distributed shrew, the grey musk shrew (*Suncus murinus*), has a pale or slightly brown coloured coat while *S. niger* has a blackish-brown coat, with a rufescent shade dorsally. The abdomen is greyish. The tail is equal to its head and body length (Sterndale 1982). Shrews are territorial with relatively small home ranges, which may vary with the mating season (Anon. 1990). They feed primarily on insects, such as cockroaches and crickets and rarely on vertebrates such as small amphibians and reptiles (Anon. 1990). In India, shrews have been reported to feed on toads (Tiwari 1994), geckos (Sharma 1995) and gerbils (Saini and Parshad 1996). Apart from this, four more species of shrews have been reported from the Western Ghats (Appendix I).

A note on taxonomic position of *Sorex* (=*Suncus*)

The old treatment of taxonomy, in terms of species nomenclature, used *Sorex* as generic name for all ground shrews in South India (Sterndale 1982). However, (Wilson and Reeder 1993), does not consider *Sorex* spp as existing in the Indian range, being replaced by *Suncus* spp. These authors name a subspecies of *Suncus montanus* as occurring in the southern range of the Western Ghats: *Suncus montanus niger*. Therefore, I update the species identification, making *Sorex niger* into *Suncus montanus niger*. As a footnote, *Sorex* is retained by one author (Hutterer 1993) who states that *Sorex planiceps* occurs in Kashmir and Pakistan. It was also retained due to some taxonomic and morphological problems.

Wilson and Reeder (1993) state that Linnaeus' (1766) taxonomic edition had been outdated due to nomenclatural reasons. Obviously, Sterndale (1982) followed old taxonomic treatment using Linnaeus (1766) work. Furthermore, the generic name *Sorex* has been used by Linnaeus for the following synonyms: *Pachyura, Podilik, Paradoxodon, Plerodus* and *Suncus*. Based on this species description *Suncus montanus* (Kelaart, 1850) is commonly included in *S. murinus* (Ellerman and Morrison-Scott 1966:66). The synonyms for this species are: *ferrugineus, kelaarti,*and *niger*. *Suncus montanus*, a much smaller and blackish species, inhabits primarily forest habitats. Corbet and Hill (1991:36) list this species as a separate species. However, as suggested by Hutterer (1993), the Indian populations may represent a valid subspecies (*niger*).

Based on the above recommendation I used the generic name *Suncus* instead of *Sorex*, since the modern literatures (*e.g.*, Corbet and Hill 1991; Hutterer 1993) describe the species in that way. Therefore, in order to distinguish the valid subspecies, I have chosen the subspecies name as *Suncus montanus niger* as it was mentioned in Hutterer's work (1993).

In conclusion, from the above taxonomic clarifications, wherever in this dissertation *Sorex niger* or *S. niger* are mentioned, hereafter, it can be referred to as *Suncus montanus niger* or *S. m. niger*. In addition, in the species data base, all the listed shrews generic name (*Sorex*) can be replaced with *Suncus*.

2. 3 METHODS

2.3.1 Surveys of Forest Fragments

As a part of the project, we conducted a survey of the Sanctuary and adjoining areas in order to identify rain forest fragments and to make an assessment of their status with reference to several landscape and habitat parameters. An assessment of the occurrence and abundance of three species of arboreal mammals (the lion-tailed macaque, the Nilgiri langur and the Malabar giant squirrel) was also made during the survey which was carried out in January to May 1994 (Kumar *et al.* 1995; Umapathy 1988). We identified 25 forest fragments which were originally of rain forest vegetation. The status of these fragments was assessed with reference to several landscape parameters (such as area, ownership, location of the landscape and distance to human settlement) and macrohabitat variables (such as tree density, basal area of trees >30 cm in GBH, canopy height, canopy cover and under growth). The macrohabitat parameters were estimated from 5 m radius circular plots, at an interval of about 100 strides, along existing trails in each fragment. In total, we sampled 350 quadrats. The number of quadrats in a fragment increased with increasing area of the fragment. The mean of quadrats in each fragment was estimated for macrohabitat parameters. The coefficient of variation among quadrats was also estimated for each fragment as an indicator of the spatial heterogeneity in each parameter.

2.3.2 Selection of Forest Fragments for the Study

Taking into consideration time and other constraints, I estimated that only seven forest fragments could be sampled in each of the four seasons for rodents and shrews using live trapping. Seven forest fragments were chosen so as to represent the variability in the landscape and habitat variables in the 25 forest fragments that we surveyed (Table 2.1). The selected fragments included Akkamalai shola, a very large fragment (>2,000 ha) that was relatively undisturbed as indicated by the higher values of macrohabitat parameters; Andiparai shola, a large fragment (>200 ha) that was relatively undisturbed; Korangumudi and Puthuthottum estates, both medium sized (50-100 ha), and moderately disturbed; Varattuparai-III and Varattuparai-IV both small (<10 ha) and highly disturbed; and Urulikkal shola, a small (30 ha) but relatively undisturbed fragment. A brief description of the seven fragments is given below.

Varattuparai - III & IV

These two fragments owned by Tata Tea company, situated at an altitude of 1,040 m, (10° 21'40N- 76° 55'70E) are less than 10 ha in area and are highly disturbed. The forest surrounding both fragments was clear felled during 1898 for tea cultivation (Congreve 1938). The Valparai-Chalakkudi road runs between the two fragments. Even though both fragments are highly disturbed, Varattuparai IV still has some good canopy cover. The topography consists of a gentle slope, with a sandy and rocky terrain. The undergrowth is full of climbers and exotic weeds such as lantana (*Lantana camara*) and eupatorium (*Eupatorium odorattum*). The major tree species are *Cullinea exarillata*, *Ficus* spp and

Table 2.1. The status of 25 rain forest fragments with reference to landscape and habitat parameters. The seven forest fragments selected for the study on small mammals are shown in bold.

Name of Fragment	Area (*ca.* ha)	Size@ class	Owner ship#	Tree density (/ha)	Canopy height (m)	Canopy cover (%)	Basal area (M^2)
Akkamalai	**2500**	**VL**	**F**	**879**	**19**	**67**	**97.54**
Varagaliar	2000	VL	F	455	31	57	49.81
Kuruvampalli	500	VL	F	410	30	55	58.73
Vellaimalai Top	200	L	F	323	13	33	33.71
Monompoly	200	L	F	347	23	52	55.46
Sankararankudi	180	L	F	184	21	33	37.48
Surlimalai	75	L	P	244	36	52	75.18
Iyerpadi	100	L	P	441	16	60	85.86
Andiparai Shola	**185**	**L**	**F**	**357**	**22**	**77**	**29.41**
Iyerpadi Church	50	M	F	403	17	59	43.90
Korangumudi	**35**	**M**	**P**	**161**	**17**	**23**	**37.82**
Puthutottum	**50**	**M**	**P**	**128**	**19**	**62**	**67.23**
Varattuparai-I	24	M	P	93	25	56	11.01
Cinnakallar	40	M	P	336	18	56	39.53
Sholayar P. House	10	S	P	244	13	27	31.41
Varattuparai-II	2	S	P	233	18	42	51.59
Varattuparai-III	**5**	**S**	**P**	**182**	**10**	**21**	**20.29**
Varattuparai-IV	**10**	**S**	**P**	**127**	**9**	**18**	**8.01**
Nirar Dam	8	S	F	164	8	75	6.54
Sholayar Dam	6	S	P	370	17	55	85.96
Monica Estate	2	S	P	127	20	51	8.48
Pannimedu-I	5	S	P	216	15	44	43.70
Pannimedu-II	10	S	P	306	14	48	42.82
Urulikkal Nursery	5	S	P	178	16	36	61.87
Urulikkal	**30**	**S**	**P**	**111**	**9**	**35**	**19.28**

@- size of fragments: S= Small <10 ha, M= Medium 11-70 ha, L= Large 71-200 ha, VL= Very Large >201 ha. #- ownership of fragments: F= Forest Department, P= Private Estate.

Mesua ferrea. The nearest forest to these patches is about 4 km away. These fragments harbour a small number of Malabar giant squirrel, two species of flying squirrels, and occasional visits by a group of Nilgiri langur. The fragments also act as day time refuges to the Indian gaur, sambar, barking deer, civets and mongooses.

Urulikkal

This fragment is located at 10° 21'40N-76° 55'70E, at an elevation of 1,030 m. The surrounding area was felled in 1927 leaving an area of less than 30 ha (Congreve 1938). The Western slope is bordered by a coffee plantation, the eastern side by eucalyptus and coffee plantations, and the southern side by the Manomboli Power House road. Even though occasional illegal tree felling and lopping were seen, this forest patch is relatively undisturbed. The fragment has a bottleneck connectivity in the northeast with the mainland forest about 2 km away. The topography is steep and rocky, hence there is sparse but good quality undergrowth. The common trees are *Mesua ferrea, Listea* spp, *Ficus* spp and *Cullinea exarillata*. The major mammals present include Nilgiri langurs, Malabar giant squirrels, flying squirrels, small carnivores such as civets, lesser cats and mongooses, gaur, sambar and barking deer.

Puthuthottum Estate

Puthuthottum Estate is situated at 10° 20'10N, 76° 57'80E, at an elevation of about 1,110 m. This was one of the earliest fragment to be formed when the surrounding area was clear felled in 1906 for a tea plantation, leaving 65 ha of forest (Congreve1938). The major tree species are *Symploccos cochinchinensis*, *Mesua ferrea* and *Cullinea exarillata*. Presently, this forest is owned by the Korangumudi Estate group. The Valparai to Pollachi state highway borders the fragment for about 2 km. Akkamalai, the nearest large fragment is about 4 km away. The undergrowth has been partially replaced with cardamom and coffee. This forest has been selectively felled a few times, the last being in 1992. Puthuthottum also faces a great deal of disturbance from tree felling and fuel wood collection. A small population (<50 animals) of lion-tailed macaques exists in this fragment. This population has been drastically affected by these changes (Kumar 1987; Menon 1993; Kumar *et al.* 1995). This fragment serves as a refuge for tiger, leopard, small carnivores such as civets, mongooses and lesser cats, as well as sambar, barking deer and is a migratory refuge for elephants, and gaur.

Korangumudi Estate

This fragment is located at 10° 18' 80N-76° 54' 60E at an elevation of 1,110 m. The area surrounding Korangumudi was also clear felled and planted with tea and coffee during 1906, leaving 35 ha of forest (Congreve 1938). The ownership of the fragment is with the Korangumudi Estate Group. In recent years, the undergrowth of the patch is being removed for cardamom and coffee plantations. The western side of the fragment is bordered by the Upper Sholayar Reservoir. The distance between the nearest mainland forest and the patch is about 7 km. Intensive selective felling, increasing illegal tree felling and firewood collection have left the patch in a highly degraded state. Even though highly degraded, it still harbours a group of lion tailed-macaque, four groups of the Nilgiri langur, Malabar giant squirrel, two species of flying squirrel, and small carnivores. In addition, elephant-man conflict is quite frequent in the area.

Andiparai Shola

Andiparai Shola is situated at 10° 24' N-76° 59'5E at an elevation of about 1,280 m. This was the first site to be explored and clear felled by British planters, creating the Waterfalls and Waverly tea estates (Congreve 1938). This left 180 ha of forest now called the Andiparai Shola. Later, the Pollachi-Valparai road and selective logging heightened its degree of disturbance. The southern and western sides of the patch are surrounded by tea plantations and the northern side is a clear felled barren valley, with thick secondary scrub vegetation. Even though the patch comes within the Sanctuary it is rapidly being degraded due to illegal tree felling and fuel wood collection by the surrounding estate people. However, a rich floral and faunal assemblage could still be seen. The major tree species are *Mesua ferrea, Listea* spp, *Myristica beddomii, Ficus* spp, *Symploccos cochinchinensis* and *Cullinea exarillata*. The major mammals are a group of about 20 lion-tailed macaques, many groups of Nilgiri langur, flying squirrels, giant squirrel, gaur, sambar, barking deer, leopard, tiger and small carnivores. In addition, it is migratory refuge for elephants during the summer.

Akkamalai Shola

This is one of the two large fragments in the Indira Gandhi Wildlife Sanctuary (the other being Varagaliar Shola), covering an area of >2,000 ha. It is located at 10° 19'80N-77°

01'30E and has an elevation range of 1,100 to 1,700 m, reaching up to the foothills of Grass Hills. The Forest Department has underplanted a small part (<10 ha) with cardamom. Floral and faunal diversity is very high. The major wildlife includes a few groups of lion tailed-macaque, many groups of Nilgiri langur, flying squirrels, giant squirrel, gaur, sambar, barking deer, leopard, tiger, wild dogs, and small carnivores. The dominant trees are *Mesua ferrea, Hopea parviflora, Pallaquium ellipticum*, *Cullinea exarillata*, *Artocarpus heterophyllus*, *Vateria indica* and many species of *Syzygium* spp and *Ficus* spp. Even though the northern side is bordered by a tea plantation and a settlement and as such is subject to degradation, the degree of disturbance in Akkamalai especially in its interior is very low.

2.3.3 Study Period

The study lasted for 27 months from December 1993 to February 1996. After an initial period of standardisation of survey methods, between January and March of 1994 we conducted a survey of the Indira Gandhi Wildlife Sanctuary to identify rain forest fragments, and to characterise them with reference to several landscape and macrohabitat parameters (see Section 2.3.1). Trapping of rodents began in the winter of 1994 in four privately owned fragments. In three Forest Department owned fragments, the trapping began later due to a delay in obtaining the trapping permission from the State Government. However, the trapping was continued through summer, wet I and wet II in 1995 and ended with winter season data in 1996 for all fragments.

2.3.4 Estimation of Rodent Abundance

2.3.4.1 Selection of Traps

Almost all the studies on rodent abundance have involved captures using various kinds of traps, such as Sherman traps of various sizes (Malcolm 1991; Kaufman *et al.* 1993; Gore 1988; Woodman *et al.* 1995; Shanker 1998), snap traps (Kemper and Bell 1985; Canova and Fasola 1991; Prakash *et al.*1995), pitfall traps (Gore 1988; Canova and Fasola 1991) and wire-mesh cage traps (Kemper and Bell 1985; Walker and Rabinowitz 1992; Laurance 1994). Methods other than Sherman traps were not suitable for our study. Making pitfall traps is very labour intensive and often impossible in rain forest areas where there is low soil depth and high rock and root density. Snap traps kill the animals, and those killed are often other than rodents and shrews. Therefore, we decided to use Sherman live traps. We selected medium sized traps (22 x 7 x 9 cm: Prakash Foundry and Engineering Works, Jodhpur, India) as they were suitable for animals of the desired size range, and have been used widely.

2. 3.4.2 Selection of Trapping Grid Configuration

Studies on rodents and shrews elsewhere have used trapping grids of various sizes and configuration (Adler and Seamon 1991; Canova and Fasola 1991; Walker and Rabinowitz 1992; Laurance 1994; Prakash *et al.* 1995). Some of these have been run over three to five days (Peltonen and Hanski 1991; Adler and Seamon 1991; Canova and Fasola 1991; Gore 1988), and some for more than five consecutive days (Kemper and

Bell 1985; Malcolm 1991; Walker and Rabinowitz 1992; Woodman *et al.*1995). Most of the studies have used only one trap at each trap station (Adler and Seamon 1991; Peltonen and Hanski 1991; Walker and Rabinowitz 1992; Laurance 1994), while some have used two traps of the same trap type (Smith and Vrieze 1979) or different traps (Gore 1988; Malcolm 1991; Woodman *et al.* 1995). Some studies have used paired trap stations, one on the ground and another on the tree at a 2 m height, for arboreal rodents and shrews (Malcolm 1991; Laurance 1994).

Taking into consideration the paucity of traps, the labour involved, constraints of time, and comparability of results with most other studies, we selected trap stations with single traps on the ground, and an inter trap station distance of 10 m. Each trapping session lasted five consecutive nights. Since the selected fragments were far apart, simultaneous trapping sessions in different fragments were not possible. Traps were baited with peanut butter, and checked once in the morning. Unsprung traps were rebaited. Sprung traps were replaced, and traps with rodents were removed from the capture site. Species, age, sex, morphometric measurements and reproductive condition were recorded. While most animals were released elsewhere, some were collected as voucher specimens and for analysis of stomach contents.

The sampling effort, in terms of the number of trap nights, was increased with increasing fragment area by changing the grid size and the number of grids used per trapping session of five days. Thus, in the very small fragments (<10 ha), one grid of 5x10 at 10 m intervals, consisting of 50 traps were run for five days per season, giving a total of 250 trap nights. In medium and large fragments (25-200 ha) one grid of 10x10, at 10 m intervals consisting of 100 traps was run for five days in each of the four seasons giving a total of 500 trap nights. In the very large fragment (>2,000 ha), two grids of the above dimensions were used giving 1,000 trap nights per season (see also Table 3.1).

2. 3. 5 Habitat Parameters

The macrohabitat parameters for each fragment were measured during the preliminary survey (Table 2.1). The parameters included fragment area, proximity to human settlement, tree density, basal area, canopy height, canopy cover and under growth. Most studies on rodents and shrews have measured several microhabitat parameters in order to examine species differences in microhabitat selection (see for example Gore 1988; Canova and Fasola 1991; Laurance 1994; Shanker 1988). These microhabitat parameters reflect differences in terrain, ground cover, undergrowth, canopy vegetation and structural complexity. These often relate to food abundance and predator protection. For the present study, I recorded 17 microhabitat parameters in $1m^2$ and $5m^2$ quadrats centred around the trap stations (Table 2.2). These were recorded at the end of the session for all sprung trap stations, and at every alternate unsprung stations.

2. 3. 6 Data Analysis

Many parameters were visually estimated, were of unknown distribution or were derived from other variables. For these reasons, wherever possible non-parametric statistical tests have been used (Siegel and Castellan 1988). Thus, Friedman test has been used for two way analysis of variance, Spearman Rank Correlation (r_s) coefficient has been used for

Table 2.2. Microhabitat parameters measured from $1m^2$ and $5m^2$ plots centred on trap station with and without captures.

Name	Definition
NTrees	Number of trees (stems >30cm girth at breast height (GBH)
NLianas	Number of lianas (counted within the plots)
NCLRattan	Number of climbing rattan (counted within plots)
NShrubs	Number of shrubs (no.of contacts <160cm height)
NFlogs	Number of fallen logs (>30 cm diameter)
NButtress	Number of buttress trees (stems >30cm gbh)
DNtree	Distance to the nearest tree from trap station
Ntree gbh	Nearest tree diameter (if within the plot)
CCover	Percent of canopy cover measured at the centre of the plot, by holding a rear view mirror divided into 20 grids
HCover	Grass and herb cover (visually estimated in %)
LCover	Fallen leaves and twigs cover on ground (visually estimated in %)
SCover	Exposed soil (visually estimated in %)
RCover	Rock cover (visually estimated in %)
LDepth	Litter depth: Mean number of leaves poked from 4 sites in a plot
PNtree	Vegetative and reproductive phases quantified visually in percentage from the tree nearest to the trap station
DWater}	Measured in meters from grid site to the nearest
DEdge }	water and edge

testing association between two variables, Kendall's coefficient of concordance has been used to test agreement between k-related groups, and Kruskal-Wallis (K-W) one way analysis of variance for k-independent groups. Chi-square test has been used for contingency table analysis, further partitioning the table, if the overall χ^2 value was significant (Siegel and Castellan 1988).

Since the number of fragments sampled were only seven, and the number of independent variables many, I attempted to reduce the latter into few, independent variables using Principal Component Analysis (PCA). Since some of the relationships appeared non-linear, I have attempted to fit linear and quadratic models to the data, choosing the most parsimonious fit. For all tests, a probability of less than 0.05 was chosen *a priori*, as the criterion for significance. However, probabilities between 0.10 and 0.05 considered marginally significant. Further details of the parameters and analysis are given in the appropriate chapters. All tests were done using SPSS (Norusis 1990), initially on a UNIX plat form, and later on Window's 95 (SPSS 1996 version 7.5).

3. COMMUNITY STRUCTURE

3.1. INTRODUCTION

Fragmentation of forest causes changes in species richness and relative abundance of terrestrial small mammals (Malcolm 1991; Bierregaard *et al.* 1992; Andrén 1994; Laurance 1994; Bright and Morris 1996; Shanker 1998). Important determinants of such changes include fragment area, changes in micro and macro habitats, extent of isolation in time and space, invasion of exotic species and vegetation surrounding the fragment. For example, the declining range and abundance of the dormouse (*Muscardinus avellanarius*) in northern counties of Britain were due to a complex interplay of factors that included fragmentation, and deterioration and loss of specialised habitats (Bright and Morris 1996). Bierregaard *et al.* (1992) found that communities of Brazilian primates responded to the fragment area, while the small mammals and lepidopterans were hypersensitive to the matrix surrounding the fragments. Densities of the better adapted small mammals increased in the fragments, while that of the primates declined (Malcolm 1991; Bierregaard *et al.* 1992).

The vegetation surrounding the fragments plays a vital role in determining small mammal communities in the fragments. In a fragmented landscape in Australia, edge or disturbance tolerant species increased, while in the contiguous forest the assemblages of ecologically similar or forest dependant communities were more structured (Laurance 1994). Species which are not habitat specialists often survive in the fragments by traversing or making use surrounding areas; for instance, rodents (Bierregaard *et al.* 1992;

Andrén 1994; Laurance 1994), elephant shrews (FitzGibbon 1994), red-backed voles (Mills 1995), and birds (Stouffer and Bierregaard 1994; Kattan *et al.* 1994). In Massachusetts Islands, out of eight species of small mammals trapped, the densities of four habitat specialists depended on the island area and that of the remaining four species depended on isolation (Adler and Wilson 1985).

Many factors that influence the occurrence and abundance of small mammals operate at macrohabitat level. These factors include area, canopy cover, tree density, primary productivity, weather, predator abundance, and successional stage of the forest stand (Kemper and Bell 1985; Bowers and Flanagan 1988; Corn 1988; Gore 1988; Nordyke and Buskirk 1988; Raphael 1988; Shanker 1998), and habitat heterogeneity (Bowers and Flanagan 1988; Pahl *et al.* 1988; Kerley 1992; Soulé *et al.* 1992). Changes at the macro habitat level, therefore, would affect the community structure of mammals.

Habitat alteration leads a species either adapt to the new environment or to perish in due course (Laurance 1990; Newmark 1991; Soulé *et al.*1992; Mills 1995). Introduced or invading species often succeed in the altered habitats (Lovejoy *et al.* 1986; Raphael 1988; Bennett 1990; Soulé *et al.* 1992; Wirminghaus and Perrin 1993; Probert and Litvaitis 1996). The immediate response to habitat fragmentation is related to the life history pattern and ecology of the animals concerned. For instance, large bodied animals often have a large home range; similarly forest dependent species, arboreal species and animals dependent on unpredictable resources are more prone to extinction (Terborgh 1986; Noss 1987; Laurance 1990; Saunders *et al.* 1993). Soulé *et al.* (1992) reported that large bodied birds in a chapparal habitat persisted longer in fragments than small bodied birds.

In this Chapter, I examine the variation among rain forest fragments in the community structure of small mammals and attempt to identify the macro habitat parameters with which such changes are associated. The community structure is analysed with reference to species richness and relative abundance. The specific objectives in this chapter are:

a) to evaluate the differences in species richness and relative abundance of rodents and shrews among rain forest fragments; and

b) to examine the influence of macrohabitat parameters on the species richness and abundances of rodents and shrews in fragments.

3.2 METHODS

3.2.1 Estimation of the Abundance of Small Mammals

A preliminary survey was conducted in Indira Gandhi Wildlife Sanctuary and adjoining areas, during January - May 1994. In total, 25 fragments were identified, which were then intensively surveyed to characterise them with reference to several habitat features (see Chapter 2). Out of these, seven fragments were chosen to conduct studies on terrestrial rodents and shrews. These fragments varied in area (<10 ha to >2,000 ha), disturbance level, ownership and matrix surrounding the fragments (Table 2.1).

The species richness and abundance of rodents and shrews were estimated through live trapping using Sherman traps. Animals were trapped through summer, southwest monsoon and northeast monsoon in 1995 and ended with the winter season in 1996 (see Section 2.3.3). The sampling effort, in terms of the number of trap nights, was increased with increasing fragment area by changing the grid size and the number of grids used per trapping session of five days. In the very large fragment (>2,000 ha) 200 traps were used in each session, medium and large sized fragments had 100 traps, and the small fragments (<10 ha) had 50 traps each. Upon capture, species, age/sex, morphometric measurements and reproductive condition of the animals were recorded. Further details are given in Chapter 4 (see also Table 3.1).

The abundance or capture rate of rodents and shrews was estimated as the number of animals/100 trap nights, from:

$$\text{Capture rate} = \frac{\text{no. of animals trapped}}{\text{total trap nights}} \times 100$$

No effort was made to estimate density, since it required reasonably high sample sizes, which were not obtained for most species.

3.2.2 Measures of Community Structure

Community structure is analysed with reference to species richness and relative abundance in each fragment, and species overlap and species composition overlap among fragments. In species richness, both overall species richness and seasonal species richness are compared among fragments. The capture rate for each species (no. of animals /100 trap nights) is used as an index of relative abundance. Species overlap or similarity between two fragments is estimated using Sorensen's index (Sorensen 1948):

$$QS = \frac{2c}{(a \times b)}$$

where,

a = the number of species in fragment 'a',

b = the number of species in fragment 'b',

c = the number of species in common between 'a' and 'b',

The index is estimated for each pair of fragments, thus yielding 21 estimates of overlap for seven fragments. This overlap index was examined with reference to the similarity between fragment pairs in macrohabitat parameters. The latter was estimated as the absolute difference between fragment pairs in the mean values of each parameter.

The species composition overlap or similarity between fragment pairs is estimated using Morisita index (Horn 1966).

$$\hat{C}_{\lambda} = \frac{2\sum_{i=1}^{s} x_i y_i}{\sum_{i=1}^{s} x_i^2 + \sum_{i=1}^{s} y_i^2}$$

where,

S = total species in both fragments, x and y, together,

x_i = proportion of species i in fragment x

y_i = proportion of species in fragment y

For both indices of overlap, the estimates are based on data pooled over all four seasons. While the Sorensen's index considers only the occurrence of species, Morisita's index considers their relative abundance also.

3.2.3 Evaluation of Habitat Parameters

An estimation of the macrohabitat parameters was done for all 25 forest fragments in the study area during the preliminary survey in January-May 1994 (see Chapter 2 and Table 2.1). This data is used in the analysis. The macrohabitat parameters included fragment area, tree density, basal area, canopy height, canopy cover, and undergrowth. Coefficient of variation (CV) between vegetation plots of 5m radius was also estimated for each habitat parameter as a measure of spatial habitat heterogeneity in each fragment. I also recorded 17 microhabitat parameters in $1m^2$ and $5m^2$ quadrats centred around all sprung trap station (see Table 2.2), and at every alternate unsprung station. This data was however, used only for examining microhabitat selection (see Chapter 5).

3.2.4 Data Analysis

Scatter plots showed that the association of the parameters of community structure with various habitat parameters might be non-linear in many cases, showing an initial increase and then a decline. Therefore, I fitted both linear and quadratic curves to the data. A quadratic model was selected when there was a substantial increase in the R^2 value and a reduction in the significance level, in spite of a reduction in the degrees of freedom from five in the linear model to four in the quadratic model due to the estimation of one more coefficient. Otherwise, a linear model was chosen so that the model selected was the most parsimonious.

Since capture rates were low, data pooled over the study period was used in analysis. The abundance of individual species was analysed only for *R. r. wroughtoni, S. niger, M. booduga* and *M. musculus* for which there were sufficient captures. Since there were only seven fragments, but several independent variables (such as area, tree density and canopy cover *etc.*) I attempted to reduce the latter to a few independent variables that captured most of the variance, through a principal component analysis (PCA). The variables included in the analysis were mean values of the habitat parameters and the CVs of these parameters. Most of the analysis was then based on these few principal component factors.

3. 3 RESULTS

3. 3. 1 Trapping Effort

In one and a half years of live trapping in seven fragments, I had 10,595 trap nights, excluding sprung traps with no capture (Table 3.1), I caught 572 animals. The overall trapping success or capture rate was 5.40 animals/100 trap nights. Trapping effort was highest in winter (3,267 trap nights) and summer (3,243), followed by wet II (2,265). Only 1,820 trap nights were run in Wet I season due to heavy rain when two fragments were excluded from sampling. One fragment was excluded in wet II season. Trapping effort was proportional to the fragment area and was highest in the largest fragment, Akkamalai (2,695 trap nights).

Table 3.1. The trapping effort (number of trap grids and trap nights) in seven forest fragments in four seasons, (excluding sprung traps with no capture).

Fragment name	Area (ha)	Grid Configuration	No.of Grids	Number of trap nights Winter	Summer	Wet I	Wet II	Total
Varattuparai-III	5	5x10	1	232	210	227	218	887
Varattuparai-IV	10	5x10	1	231	222	225	219	897
Urulikkal	30	10x10	1	480	496	nd	nd	976
Puthuthottum	50	10x10	1	478	476	478	446	1878
Korangumudi	35	10x10	1	459	474	nd	472	1405
Andiparai	185	10x10	1	464	478	469	446	1857
Akkamalai	>2000	10x10	2	923	887	421	464	2695
Total				3267	3243	1820	2265	10595

nd =no data.

3. 3. 2 Species Richness

Seven species of rodents and one species of shrew were obtained over the study period. Six species of rodents were of the Family Muridae; the white-bellied wood rat (*Rattus rattus wroughtoni)*, the white-tailed wood rat (*R. blanfordi*), the Indian field mouse *(Mus booduga)*, the house mouse (*M. musculus)*, the long-tailed tree mouse *(Vandeleuria oleracea)*, and the Indian bush rat (*Golunda ellioti)*. One species of dormouse of the Family Muscardinidae, the Malabar spiny dormouse (*Platacanthomys lasiurus)* was also trapped. The ground shrew trapped was the Nilgiri wood shrew (*Sorex niger*) of the Family Soricidae. Two species of ground squirrels, the three-striped palm squirrel (*Funambulus tristriatus*) and the dusky-striped squirrel (*F. sublineatus*) were often seen in the study area. The former was infrequently trapped in the disturbed forest patches while the latter was never trapped. These two diurnal species are not considered further in the analysis.

The number of species trapped from a fragment varied from four to seven. *P. lasiurus* and *V. oleracea* were trapped only in Akkamalai while *R. blanfordi* was trapped only in Varattuparai-III and IV, and Akkamalai. *Golunda ellioti* was trapped only in Korangumudi and Puthuthottum. All fragments had *R. r. wroughtoni*, *S. niger*, *M. booduga* and *M. musculus,* except Varattuparai-III where *M. musculus* was not trapped. The maximum number of species (seven) was trapped in Akkamalai (see Table 3.2), *G. ellioti* being the only species that was not trapped here.

Table 3.2. Rodent and shrew species trapped during the study period in the rain forest fragments.

Name of	Species trapped								
Fragment	RRAT	SNIG	MBOD	MMUS	RBLAN	VOLEA	GELIO	PLASI	NSPP
Varattuparai-III	*	*	*		*				4
Varattuparai-IV	*	*	*	*	*				5
Urulikkal	*	*	*	*					4
Puthuthottum	*	*	*	*			*		5
Korangumudi	*	*	*	*			*		5
Andiparai	*	*	*	*					4
Akkamalai	*	*	*	*	*	*		*	7

Abbreviation: RRAT=*Rattus rattus wroughtoni*, SNIG=*Sorex niger*, MBOD=*Mus booduga*, MMUS=*M.musculus*, RBLAN=*R.blanfordi*, VOLEA=*Vandeleuria oleracea*, GELIO=*Golunda ellioti*, PLASI=*Platacanthomys lasiurus* , NSPP= Number of species.

All the eight species were trapped only in wet I season. Six species were trapped in summer, five in wet II and four in winter. *R. r. wroughtoni*, *S. niger* and *M. booduga* were trapped in all seasons, and *P. lasiurus* only in wet I, while the others were trapped in two or three seasons (Table 3.3).

The number of species in a fragment varied with the season, from two to five (Table 3.4). The highest number of species (five) was in Akkamalai in wet I, and in Puthuthottum in summer and wet I. In the other fragments, the number of species trapped in a season varied from two to four. The mean of seasonal species richness was highest in Puthuthottum (4.0), followed by Korangumudi (3.7), and lowest in Varattuparai-III (2.5). The mean of seasonal species richness was highest in summer (4.0).

Table 3.3. Rodent and shrew species trapped in seven rain forest fragments in four seasons.

	Species trapped								
Season	RRAT	SNIG	MBOD	MMUS	RBLAN	VOLEA	GELIO	PLASI	NSPP
Winter	*	*	*		*				4
Summer	*	*	*	*	*		*		6
Wet I	*	*	*	*	*	*	*	*	8
Wet II	*	*	*			*	*		5

Abbreviation: RRAT=*Rattus rattus wroughtoni*, SNIG=*Sorex niger*, MBOD=*Mus booduga*, MMUS=*M.musculus*, RBLAN=*R.blanfordi*, VOLEA=*Vandeleuria oleracea*, GELIO=*Golunda ellioti*, PLASI=*Platacanthomys lasiurus* and NSPP=Number of species.

Table 3.4. The seasonal variation in the number of species trapped in seven rain forest fragments.

Name of fragment	Season				Mean Seasonal Spp.richness
	Winter	Summer	Wet I	Wet II	
Varattuparai-III	2	4	2	2	2.50
Varattuparai-IV	3	4	3	3	3.25
Urulikkal	3	4	nd	nd	3.50
Puthuthottum	3	5	5	3	4.00
Korangumudi	3	4	nd	4	3.67
Andiparai	3	4	2	3	3.00
Akkamalai	3	3	5	3	3.50
x̄ number of species	2.86	4.00	3.40	3.00	

nd= no data

3. 3. 3 Correlates of Species Richness

The number of species captured in the fragments over the entire study period (overall species richness) was correlated with area of the fragment, basal area, tree density, canopy height and canopy cover (Figure 3.1 a-e). The overall species richness was positively correlated with all habitat parameters, correlation being highest and significant with basal area (R^2=0.858; *df*=5; P=0.020), followed by fragment area (R^2=0.797; *df*=5; P=0.041) and nearly significant for tree density (R^2 =0.748; *df*=5; P=0.064). However, these significant correlations, were due to the greater species richness in the Akkamalai. When Akkamalai was excluded, there was no correlation between overall species richness in a fragment and various habitat parameters, the R^2 varying from 0.02 to 0.23.

Overall species richness in fragments was also compared with spatial heterogeneity of the habitat parameters as measured by coefficient of variation (CV). The R^2 varied from 0.295 to 0.595, none being significant. Here also, the largest fragment, Akkamalai, stood apart from other fragments showing low spatial heterogeneity (low CV), but high species richness. Among the other fragments the habitat heterogeneity varied considerably, but number of species varied only from four to five. There was thus no indication that the overall species richness in the fragments was influenced by the macrohabitat parameters that I measured, including spatial heterogeneity.

Since the overall species richness was influenced by rare captures of species, I examined mean seasonal species richness in fragments in relation to the habitat parameters and spatial heterogeneity. The mean seasonal species richness showed only a weak linear

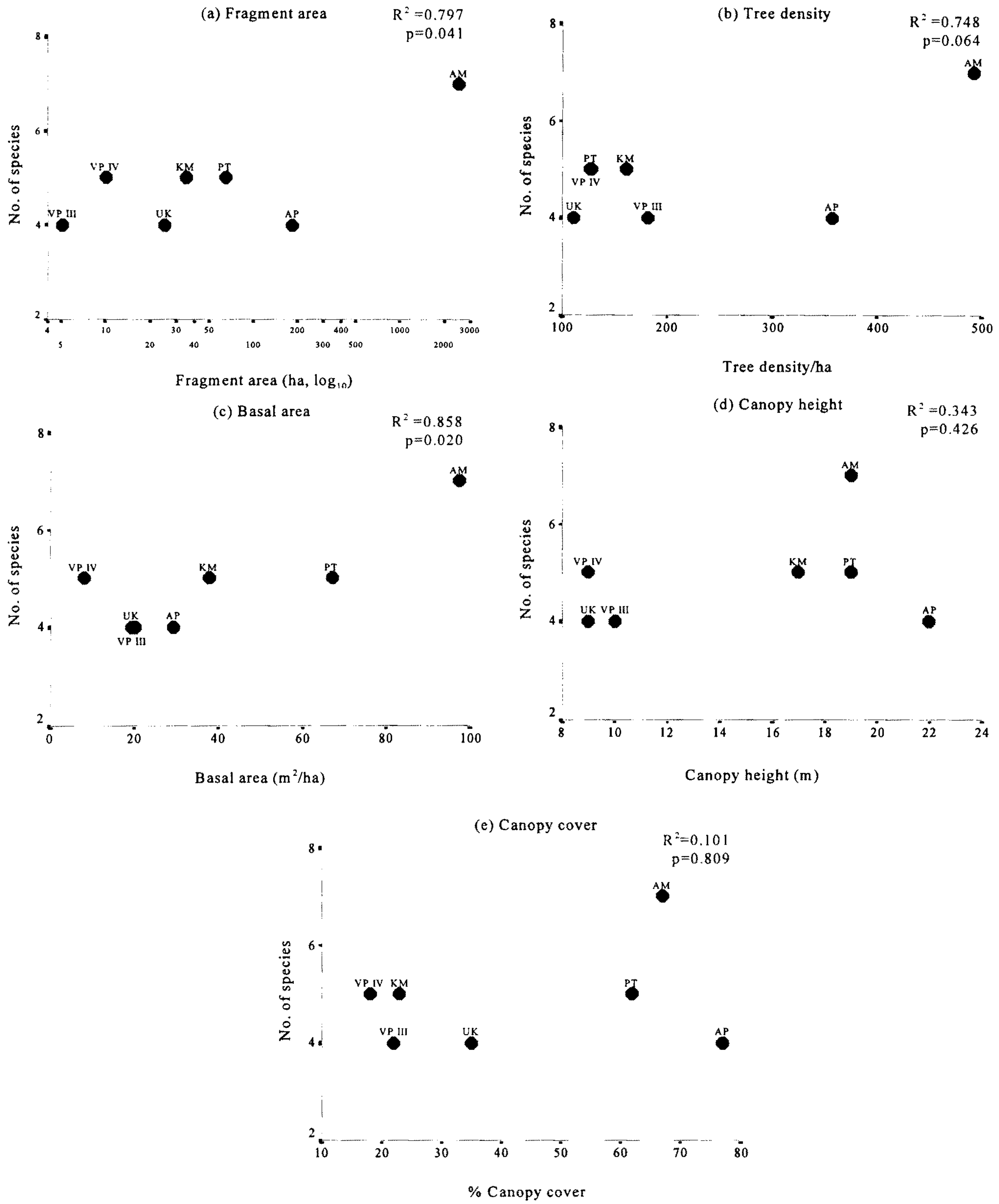

Figure 3.1. The overall species richness in fragments in relation to habitat parameters (case labels: VP III=Varattuparai-III, VP IV=Varattuparai-IV, UK=Urulikkal, PT= Puthuthottum, KM=Korangumudi, AP=Andiparai, AM=Akkamalai).

correlation with habitat parameters, the R^2 varying from 0.02 to 0.24 in the case of mean values of habitat parameters, none being significant. It showed better linear correlation with the CVs of habitat parameters, being significantly and positively correlated with CV of canopy height (R^2=0.591; *df*=5; P=0.043), and marginally so with tree density (R^2=0.436; *df*=5; P=0.107).

An examination of the scatter plot revealed the relationship between seasonal species richness and fragment area might be non-linear, a quadratic model giving a better fit both in R^2 (0.420) and significance level (P=0.336), than linear fit (R^2=0.130; P=0.427). Seasonal species richness increased rapidly with area initially, but the two larger fragments (Andiparai and Akkamalai) had lower seasonal species richness than medium sized fragments. A quadratic model gave a better fit also with canopy cover (R^2=0.374; P=0.392), compared with linear model (R^2=0.050; P=0.629). For the other habitat parameters, quadratic model did not substantially increase the R^2, and moreover the increase was associated with an increase in P values. Thus seasonal mean species richness did not seem to be related, either linearly or non-linearly, with the habitat parameters, even though quadratic function gave a substantially better fit with fragment area and canopy cover.

On the other hand, linear models gave better fits with CVs of habitat parameters. The seasonal species richness increased with increasing CV of tree density and canopy height (Figure 3.2 a-d). Seasonal species richness was the lowest in the fragments with low CV of tree density and canopy height, which included both the large fragments (Akkamalai and Andiparai) as well as the small fragments. The seasonal species richness was highest

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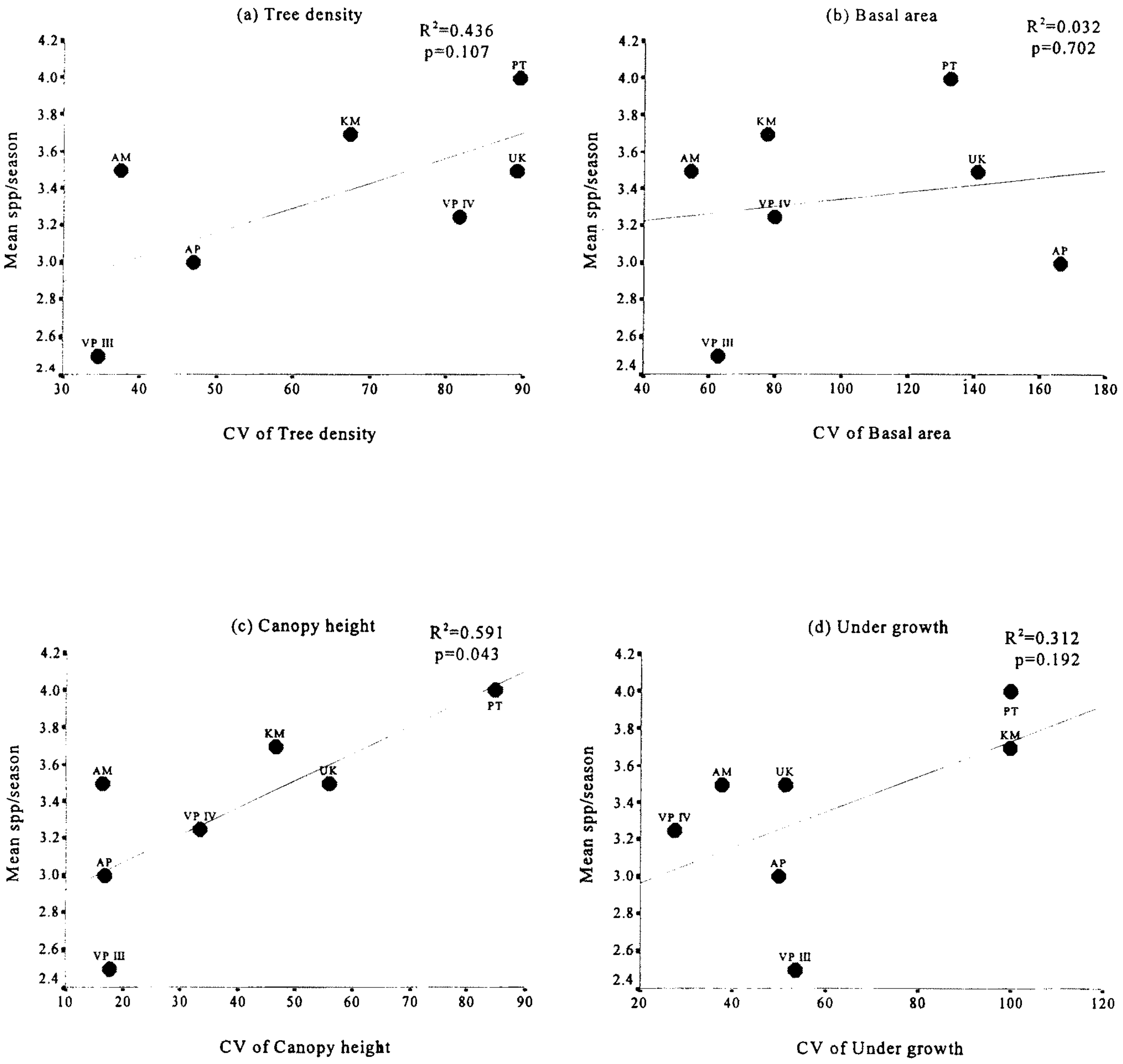


Figure 3.2. Mean seasonal species richness of rodents and shrews in relation to CVs of habitat parameters (case labels as in Figure 3.1).

in the fragments with a high CV in tree density and canopy height which were the medium sized and moderately disturbed fragments. In the case of basal area however, the highest seasonal species richness was in the fragments with moderately high CVs, and lower in fragments with low (the largest and smallest fragments) and high CV (*e.g.*, Andiparai shola). An examination of the R^2 values and significance levels showed that for CV of tree density, linear relation ($R^2=0.436$; *df*=5; P=0.107) gave a better fit than quadratic ($R^2=0.450$; *df*=4; P=0.307). For canopy height, linear ($R^2=0.591$; *df*=5; P=0.043) was better than quadratic ($R^2=0.595$; *df*=4; P=0.164). For basal area a quadratic fit ($R^2=0.314$; *df*=4; P=0.470) was better than linear ($R^2=0.032$).

Thus, seasonal species richness showed a non-linear relationship with some of the habitat parameters (fragment area and canopy cover), and a linear relationship with habitat heterogeneity as indicated by the CVs of habitat parameters, for example tree density and canopy height. For mean values of most parameters, the seasonal species richness was highest in fragments with medium values, which indicated medium sized fragments that were moderately disturbed. Both small fragments which were heavily disturbed and large fragments which were least disturbed had fewer species in a season.

3.3.4 Principal Component Analysis

A principal component analysis (PCA) of mean values of habitat parameters and their CVs, identified two factors that together accounted for nearly 77% of the total variance in the data. The first factor (Factor 1), which accounted for 51% of the variance, represented the vegetation status of the fragment, as indicated by the mean values of the habitat parameters, especially tree density (Table 3.5). This represented the general trend among fragments for the tree density, basal area, canopy cover, and canopy height to increase with increasing area. The second factor (Factor 2) represented the spatial variation in the habitat parameters, and thus the habitat heterogeneity. This factor accounted for 26% of the total variance in the data set. Factor 1 was linearly correlated with area of the fragment ($R^2=0.85$; P=0.003). Thus the vegetation status of the habitat, as measured by tree density, basal area, canopy height and canopy cover, improved as fragment area increased (Figure 3.3). In contrast, Factor 2 had a quadratic relation with fragment area ($R^2=0.73$; P=0.073), the probability being marginally significant.

Table 3.5. Factor loadings of habitat parameters, obtained in a principal component analysis.

Habitat parameters	Factor 1	Factor 2	Factor 3
Log area	.920	.113	.140
Tree density	.938	-.331	.010
Basal area	.741	.229	.565
Canopy cover	.869	.377	-.377
Canopy height	.832	.430	.096
CV basal area	.076	.645	-.748
CV canopy height	-.394	.869	.214
CV litter cover	-.947	-1.23	.263
CV tree density	-.584	.663	-.062
CV under growth	-.110	.738	.440

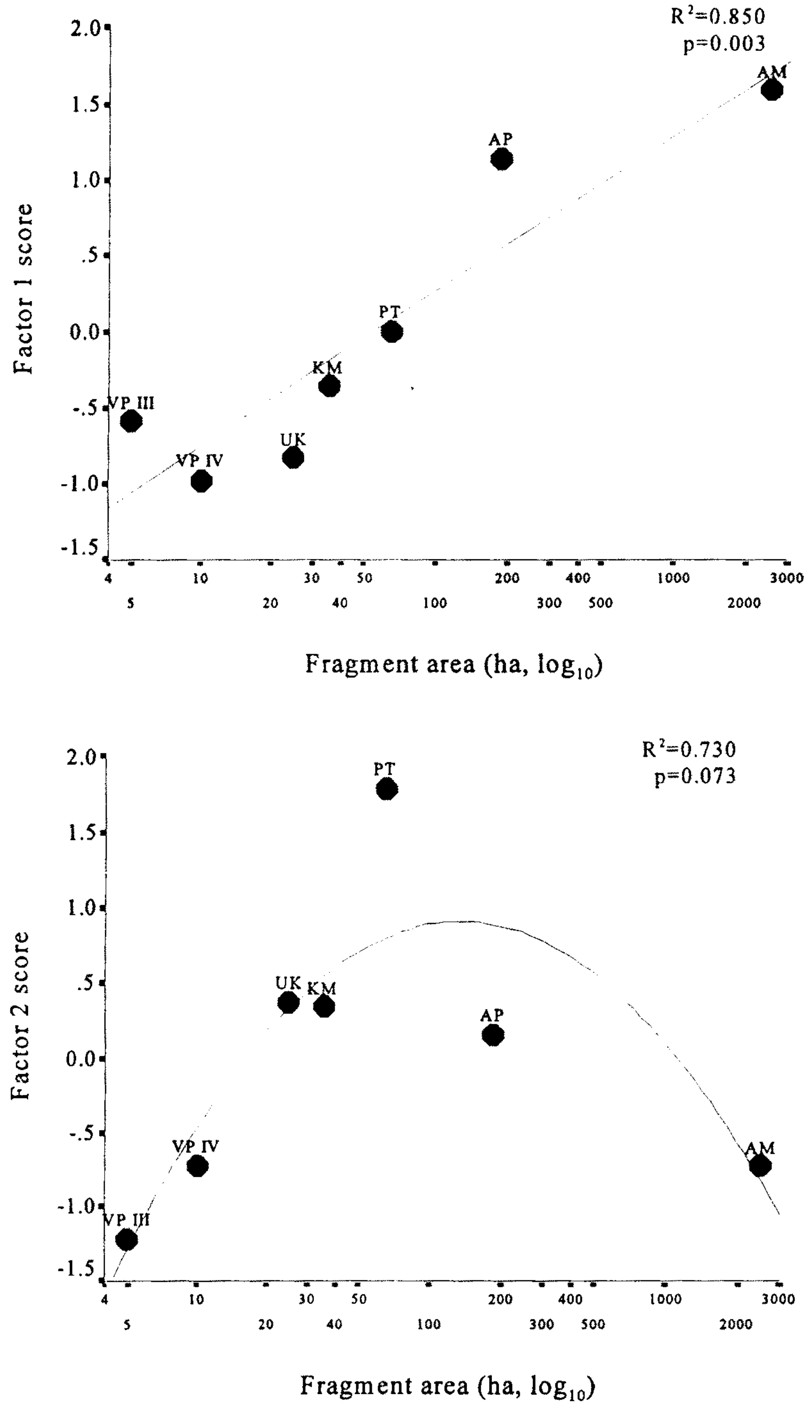


Figure 3.3. The relationship of the factors obtained in a PCA of habitat parameters and their CVs. Factor 1 (above), represented the status of the fragments in terms of mean values of habitat parameters. Factor 2 (below), represented spatial heterogeneity as measured by CVs of the habitat parameters.

The linear correlation was considerably lower (R^2=0.013; P=0.809) (Figure 3.3). Thus, spatial heterogeneity (as measured by CV of the habitat parameters) initially increased, reaching a peak at medium sized fragments, before decreasing. The smaller and larger fragments differed considerably in vegetation status (as measured by the mean values of habitat parameters), but both had similar spatial variation or heterogeneity. The medium sized fragments had medium mean values for habitat parameters, being moderately disturbed, but had the greatest spatial heterogeneity.

3.3.5 Total Abundance

The abundance of animals was estimated as the number of animals captured/100 trap nights. The abundance in a fragment over the study period varied from 4.38 to 8.26 with an overall mean of 5.51 animals/100 trap nights (Table 3.6). The seasonal abundance, for all fragments together, varied from 3.87 to 6.59 animals/100 trap nights. The seasonal abundance within fragments showed greater variation, from 1.05 in summer to as high as 13.29 animals/100 trap nights in Korangumudi in winter. Out of the 25 trapping sessions, ten had an abundance of 4.0 to 5.9, seven had 1 to 3.9, five had 6.0 to 9.9, and only three had an abundance of greater than 10 animals per 100 trap nights.

The within season variation among the fragments in abundance seemed fairly consistent, the coefficient of variation ranging from 37.43% to 64.27%. The within fragment variation among seasons had a larger range, from 16.85% to 77.38%. The seasonal variation was low in some fragments (Varattuparai III, Puthuthottum and Andiparai shola), and high in some others (Korangumudi and Akkamalai). When seasonal variations were controlled for, there was no significant difference among fragments in overall rodent and shrew abundance (Friedman's two way analysis of variance, χ^2=6.56; P>0.1610). Similarly, when variations among fragments were controlled for, there was no significant difference among seasons (Friedman's test, χ^2=2.04; P>0.5641). However, as indicated by the χ^2 and P values, differences among fragments were greater than seasonal differences.

Table 3.6. The capture rates (number of animals/100 trap nights) of all rodents and shrews together, in seven fragments and four seasons.

Name of fragment	No.of trap nights	No.of animals/100 trap nights Winter	Summer	Wet I	Wet II	Mean	CV(%)
Varattuparai-III	887	6.03	4.28	4.40	4.59	4.85	16.85
Varattuparai-IV	897	9.09	4.50	1.33	6.85	5.46	61.02
Urulikkal	976	7.10	3.43	nd	nd	5.22	49.29
Puthuthottum	1878	5.23	5.67	5.02	8.29	6.02	25.05
Korangumudi	1405	13.29	1.05	nd	10.59	8.26	77.38
Andiparai	1857	3.45	3.56	4.90	5.83	4.41	25.70
Akkamalai	2695	1.95	4.62	10.69	3.02	4.38	77.00
Mean	10,595	6.59	3.87	5.27	6.53	5.51	
CV (%)		57.09	37.43	64.27	41.25	24.37	

nd= no data

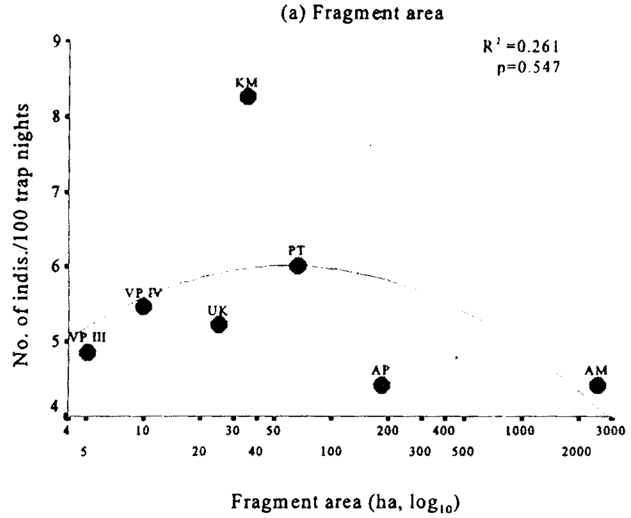
3.3.6 Correlates of Total Abundance

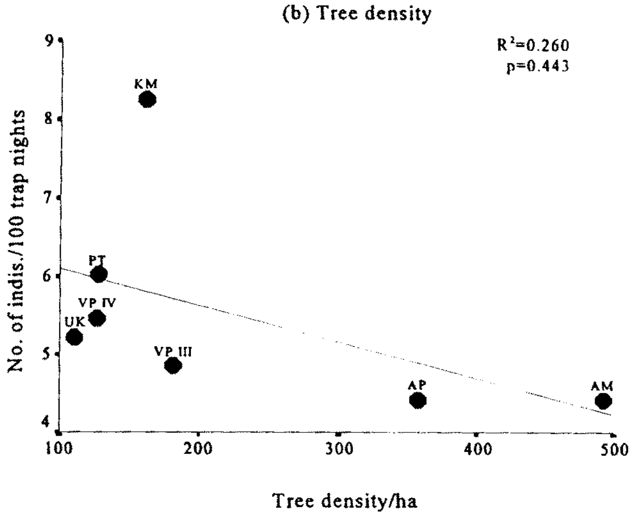
The total abundance of rodents and shrews in fragments was compared with fragment area, and mean values of tree density, basal area, canopy height, canopy cover and their CVs (Figure 3.4 a-e). The linear correlations of total abundance with habitat parameters were mostly low, the R^2 varying from 0.00 to 0.26, none being significant (Table 3.7). Quadratic models gave a greater R^2 for all parameters, however, there was a substantial increase in R^2 with a reduction in P values, only for basal area, (R^2 from 0.008 to 0.29, and P value, from 0.849 to 0.503) and canopy height (R^2 from 0.000 to 0.456, and P value from 0.966 to 0.296). A visual examination of the scatter plot showed that the total abundance increased rapidly with an initial increase in area up to about 70 ha, with a drastic decline in the two large fragments, Andiparai and Akkamalai. A similar trend was perhaps discernible with tree density, even though a linear model was a better fit (Table 3.7). For canopy cover, a linear decrease was obtained in total abundance. None of the relationships were statistically significant, however. Quadratic curves gave better fit (with greater R^2 and lower P values) with CVs, in the case of tree density, basal area, canopy height and canopy cover, the R^2 varying from 0.437 to 0.488, and P values from 0.262 to 0.317 (Figure 3.5 a-d). In contrast, in linear models the R^2 varied from 0.024 to 0.248, and P values from 0.255 to 0.740. Only CV of undergrowth was linearly and positively correlated and it was statistically significant (R^2=0.569; *df*=5; P=0.050). The quadratic curve gave a higher R^2 (0.712), but a lower P value (0.083).

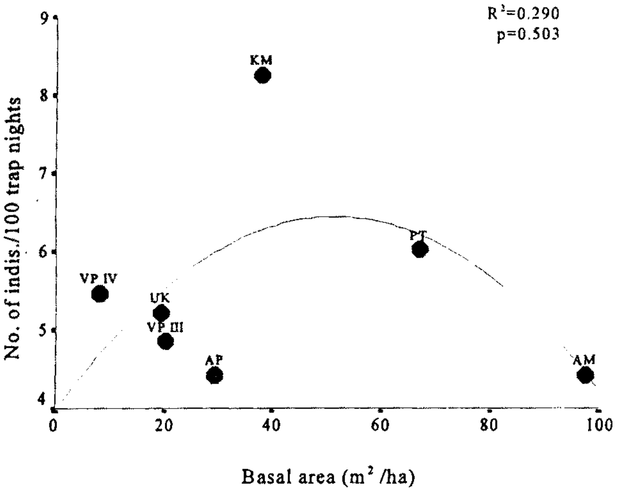
Table 3.7. The R^2 and P values of linear and quadratic fits of the abundance (animals/100 trap nights) of rodents and shrews with habitat parameters.

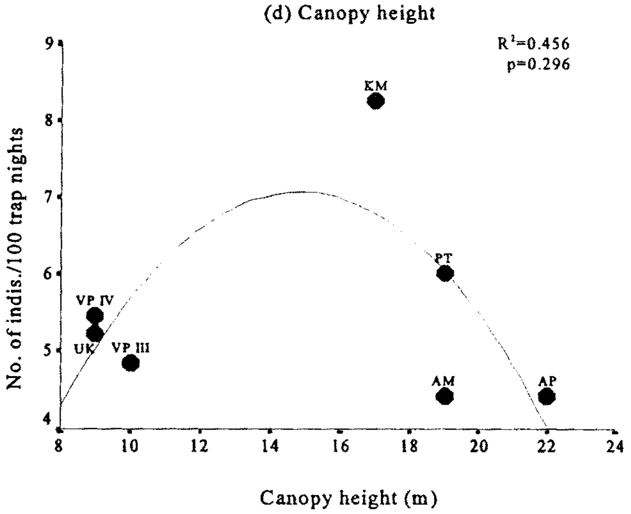
Parameters	Model	Total Abundance		RRAT		SNIG		MBOD		MMUS	
		R^2	P	R^2	P	R^2	P	R^2	P	R^2	P
Area	Linear (L)	0.082	0.533	0.000	0.974	0.253	0.250	0.038	0.667	0.017	0.778
	Quadratic (Q)	0.261	0.547	0.215	0.616	0.265	0.540	0.512	0.239	0.448	0.262
Tree density	L	0.260	0.443	0.057	0.607	0.159	0.376	0.288	0.214	0.049	0.633
	Q	0.261	0.887	0.058	0.887	0.363	0.406	0.298	0.442	0.107	0.797
Basal area	L	0.008	0.849	0.074	0.554	0.158	0.378	0.013	0.807	0.057	0.605
	Q	0.290	0.503	0.570	0.185	0.339	0.437	0.571	0.184	0.170	0.689
Canopy cover	L	0.214	0.296	0.153	0.385	0.407	0.123	0.008	0.853	0.201	0.314
	Q	0.238	0.582	0.521	0.230	0.455	0.297	0.284	0.513	0.272	0.530
Canopy height	L	0.000	0.966	0.086	0.523	0.021	0.758	0.350	0.687	0.136	0.416
	Q	0.456	0.296	0.100	0.810	0.245	0.570	0.136	0.416	0.148	0.727
Tree density CV	L	0.177	0.348	0.073	0.557	0.003	0.912	0.409	0.122	0.498	0.076
	Q	0.486	0.264	0.295	0.497	0.355	0.416	0.447	0.306	0.499	0.251
Basal area CV	L	0.024	0.740	0.265	0.237	0.094	0.504	0.065	0.581	0.354	0.159
	Q	0.437	0.317	0.311	0.474	0.175	0.681	0.543	0.209	0.533	0.218
Canopy height CV	L	0.248	0.255	0.454	0.097	0.000	0.966	0.737	0.013 ***	0.571	0.049 *
	Q	0.488	0.262	0.724	0.076	0.277	0.522	0.763	0.056 *	0.658	0.117
Under growth CV	L	0.569*	0.050	0.430	0.110	0.235	0.270	0.835	0.004 ***	0.138	0.412
	Q	0.712	0.083	0.623	0.142	0.237	0.583	0.859	0.020 **	0.280	0.519
Factor 1	L	0.170	0.358	0.002	0.933	0.190	0.329	0.084	0.928	0.004	0.897
	Q	0.392	0.369	0.536	0.215	0.330	0.449	0.554	0.199	0.130	0.757
Factor 2	L	0.160	0.374	0.565	0.051 *	0.010	0.835	0.689	0.021 **	0.730	0.014 ***
	Q	0.194	0.649	0.711	0.083	0.051	0.901	0.688	0.097	0.741	0.067

RRAT=*R. r. wroughtoni*, SNIG=*S. niger*, MBOD=*M. booduga* & MMUS=*M. musculus*
*=$P<0.05$; **=$P<0.02$ & ***=$P<0.01$.


(a) Fragment area


(b) Tree density


(c) Basal area


(d) Canopy height

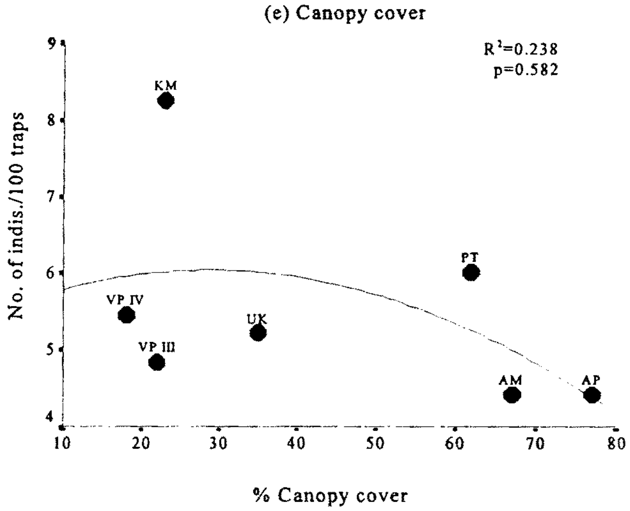

(e) Canopy cover

Figure 3.4. The total abundance of rodents and shrews (no. of animals/100 trap nights) in relation to macrohabitat parameters, with the best fit curves.

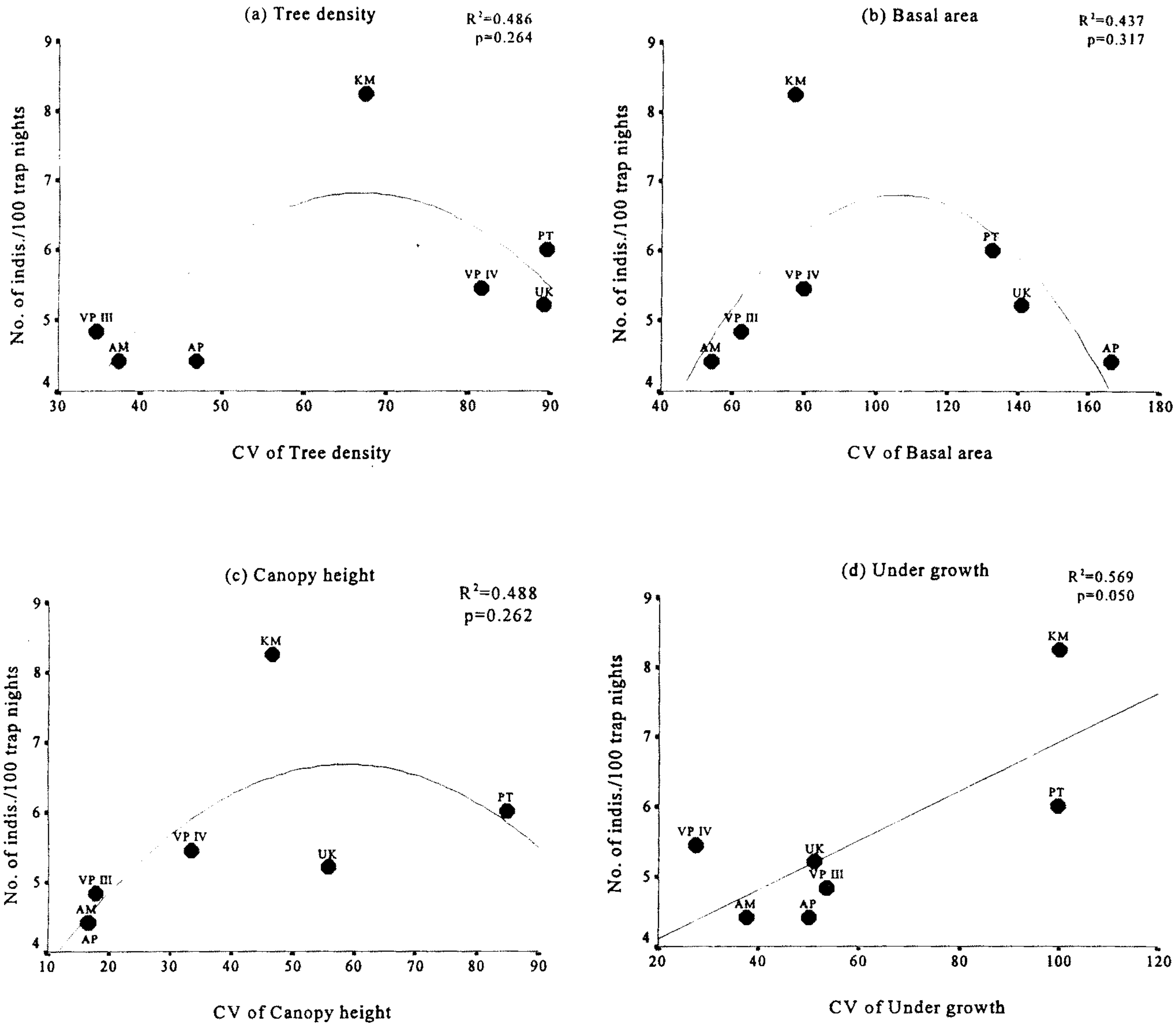


Figure 3.5. The total abundance(no. of animals/100 trap nights) in relation to spatial heterogeneity (CVs) of macrohabitat parameters, with the best fit curves.

Thus, the total abundance had a non-linear relationship with most of the habitat parameters and their CVs with an initial increase with increasing values of parameters, followed by a decline. The absence of significance is due to the small sample size of only seven fragments, and the loss of degrees of freedom, from five to four in a quadratic fit.

An examination of the total abundance with the two Factors did not indicate a strong correlation, with both linear and quadratic models (Table 3.7). A quadratic fit with Factor 1 or vegetation status gave the highest R^2 (0.392; P=0.369). Thus the total abundance of rodents and shrews did not show a strong correlation with any macrohabitat parameters. The most discernible trend was for it to be highest in fragments with moderate vegetation status, *i.e.,* medium sized fragments with moderate mean values of habitat parameters (Figure 3.6).

3.3.7 Abundance of Species

A multiple regression revealed that 58.9% of the variance in total abundance was due to *M. booduga*, 29% due to *S. niger*, and 10.2% due to *R. r. wroughtoni*. These three species together thus accounted for 98.1% of the variance in total rodent abundance among fragments over the study period.

R. r. wroughtoni was by far the most abundant species in all the seasons (2.25 to 2.91 animals/100 trap nights) and overall (2.62 animals/100 trap nights) (Table 3.8). The next most abundant species was *S. niger*, with the seasonal abundance ranging from 0.65 to 2.47, and an overall abundance of 1.52, the seasonal variation being greater than in *R. r.*

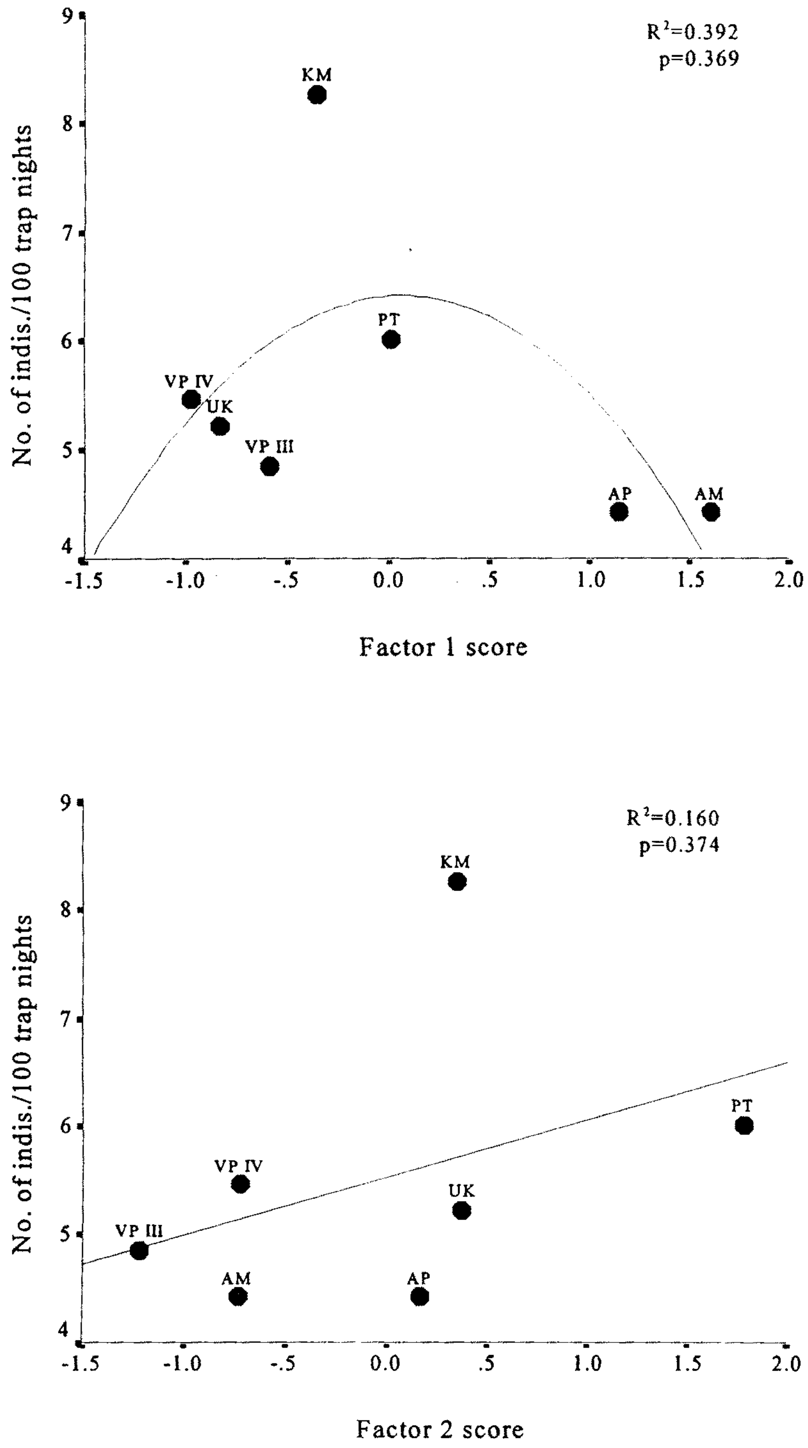


Figure 3.6. The total abundance of rodents and shrews in relation to Factor 1 (habitat parameters) and Factor 2 (habitat heterogeneity).

wroughtoni. The third most abundant species was *M. booduga*, with an overall abundance of 0.95, and a seasonal range of 0.44 to 1.44. The abundance of these three species was greater than other species in all seasons.

Table 3.8. Capture rates of rodents and shrews (no. of animals/100 trap nights) in four seasons, for all fragments together.

		no. of animal/100 traps								
Season	Ntrap	RRAT	SNIG	MBOD	MMUS	RBLAN	VOLEA	GELIO	PLASI	Total
Winter	3267	2.75	1.53	1.44		0.06				5.78
Summer	3243	2.25	0.65	0.58	0.28	0.06		0.06		3.08
Wet I	1820	2.69	1.87	0.44	0.11	0.05	0.16	0.11	0.33	5.77
Wet II	2265	2.91	2.47	1.19			0.09	0.04		6.71
Overall	10595	2.62	1.52	0.95	0.10	0.05	0.05	0.05	0.06	5.40
CV		10.7	46.6	52.5	135.6	67.58	124.2	87.11	200	

Abbreviation: RRAT=*Rattus rattus wroughtoni*, SNIG=*Sorex niger*, MBOD=*Mus booduga*, MMUS=*M.musculus*, RBLAN=*R.blanfordi*, VOLEA=*Vandeleuria oleracea*, GELIO=*Golunda ellioti*, PLASI=*Platacanthomys lasiurus*, Ntrap = number of trap nights.

All seasons together, *R. r. wroughtoni* was the most abundant in five fragments, and the second most abundant in two fragments. *S. niger* was the most abundant in Korangumudi, and second most abundant in five fragments. *M. booduga* was the most abundant in Puthuthottum and the third most abundant in five fragments (Table 3.9). *M. musculus* occurred in six fragments, but in low abundance (0.07 to 0.21). The other four species occurred in low abundance or were absent in many fragments. When variation among fragments was controlled for, the abundances of the most common four species were significantly different (Friedman test, $\chi^2=15.04$; *df*=3; $P<0.0018$), *R. r. wroughtoni,* being the most common, followed by *S. niger*, *M. booduga* and *M. musculus*.

The relative abundance of species varied among fragments for the four most abundant species (*R. r. wroughtoni, S. niger*, *M. booduga* and *M. musculus*). Over all the fragments, *R. r. wroughtoni* comprised 50% of the captures, followed by *S. niger* (30%), *M. booduga* (15%), and *M. musculus* (1.76%). The remaining four species together formed less than 4% of the captures. The relative abundance of the four dominant species varied considerably among the fragments. *R. r. wroughtoni* varied from 64.71% of the captures in Akkamalai to 32.72% in Korangumudi. Similarly *S. niger* varied from 41.86% of the captures to 20.27%. *M. booduga* showed greater variation from 40.03% to 0.90%, and *M. musculus*, from 3.49% to 0.85%. The relative abundance of each of the dominant species was examined in relation to fragment area and the two Factors.

3.3.8 Correlates of the Abundance of Species

There was sufficient data on only four species (*R. r. wroughtoni, S. niger, M. booduga* and *M. musculus*) to examine the relationship between the abundance and the macrohabitat variables. As in the case of total abundance, I examined linear and quadratic relationships between the abundance of individual species and the habitat parameters. The most parsimonious model with the highest R^2 and lowest significant value was selected

Table 3.9. The abundance of rodents and shrews (number of animals/100 trap nights) in seven rain forest fragments.

Name of Fragment	no.of animals/100 trap nights								
	RRAT	SNIG	MBOD	MMUS	GELIO	RBLAN	VOLEA	PLASI	Total
Varattuparai-III	2.59	2.03	0.11			0.11			4.84
Varattuparai-IV	3.23	1.56	0.22	0.11		0.33			5.45
Urulikkal	2.46	1.33	1.33	0.10					5.22
Puthuthottum	1.97	1.22	2.41	0.21	0.21				6.02
Korangumudi	2.77	2.92	2.42	0.07	0.07				8.25
Andiparai	2.64	1.40	0.27	0.11					4.42
Akkamalai	2.86	0.96	0.04	0.07		0.04	0.18	0.22	4.37

Abbreviation: RRAT=*Rattus rattus wroughtoni*, SNIG=*Sorex niger*, MBOD=*Mus booduga*, MMUS=*M.musculus*, RBLAN=*R.blanfordi*, VOLEA=*Vandeleuria oleracea,* GELIO=*Golunda ellioti*,PLASI=*Platacanthomys lasiurus.*

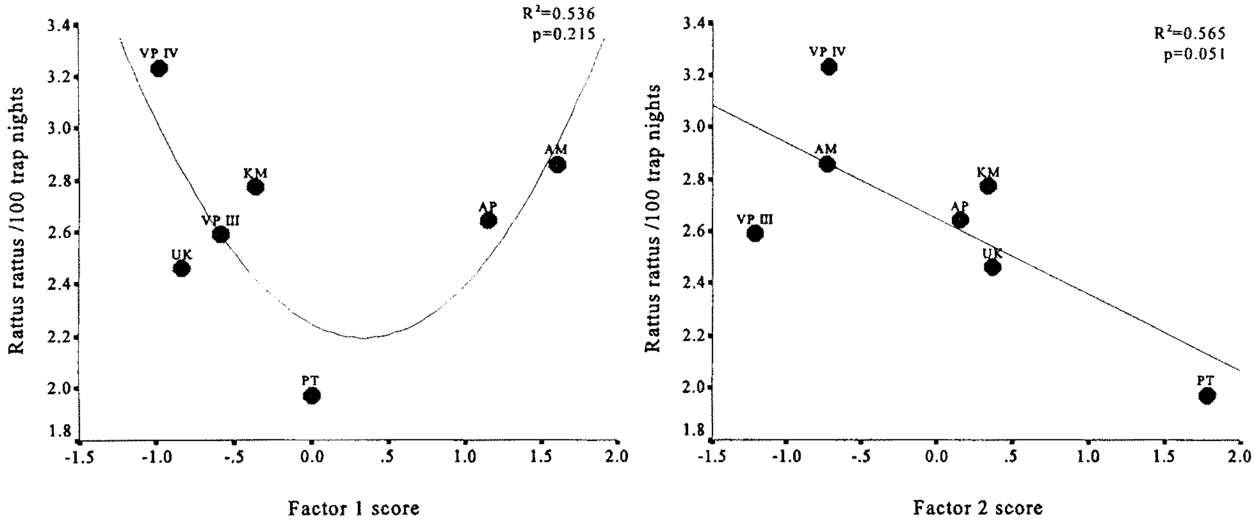
for each species (see Table 3.7). I also examined the relationship with Factors 1 and 2 which accounted for about 77% of the total variance in the habitat parameters (Section 3.4.4). The abundance of *R. r. wroughtoni* showed only weak linear and quadratic relationship with each of the macrohabitat parameters, the R^2 varying from 0.00 to 0.57, with lowest significant level being 0.185. Quadratic fit with basal area gave the highest R^2 (0.57), which was however not significant (0.185). It was better correlated with CVs of the macrohabitat parameters, the best correlation being with CV of canopy height, a quadratic model giving an R^2 of 0.724 which was marginally significant (P=0.076), and lower than linear relationship. The quadratic function also gave a better correlation with CV of undergrowth (R^2 = 0.623) compared to linear function (R^2 =0.430), with only a marginal increase in value (from 0.110 to 0.142).

Among the two Factors, the best fit was given by Factor 2 which measured habitat heterogeneity, with the linear function giving an R^2 of 0.565, very close to significant level (P=0.051). The quadratic function increased the R^2 to 0.711, but with an increase in P value to 0.083. Thus, the abundance of *R. r. wroughtoni* was linearly and negatively related to habitat heterogeneity as measured by spatial variation in the macrohabitat parameters, rather than to the mean values of these parameters (Figure 3.7)

The abundance of *S. niger* was only weakly correlated with all macrohabitat parameters (with R^2 ranging from 0.003 to 0.455, and P values from 0.123 to 0.912). The highest correlation was with canopy cover, the linear function giving an R^2 of 0.407 (P=0.123). Correlations with CVs of habitat parameters were also consistently low. Among the two factors, Factor 1 gave a better correlation, the quadratic function giving a higher R^2 (0.330), than linear (0.190), but with an increase in P value from 0.329 to 0.449. Thus, the abundance of *S. niger* was not significantly correlated with any of the macrohabitat parameters or their CVs, either individually or collectively (Figure 3.7).

The abundance of *M. booduga* showed much clearer relationships than *R. r. wroughtoni* and *S. niger* with macrohabitat parameters. Among the mean values of macrohabitat parameters, the highest R^2 values were obtained for quadratic relationships with basal

a) *R. r. wroughtoni*



b) *S. niger*

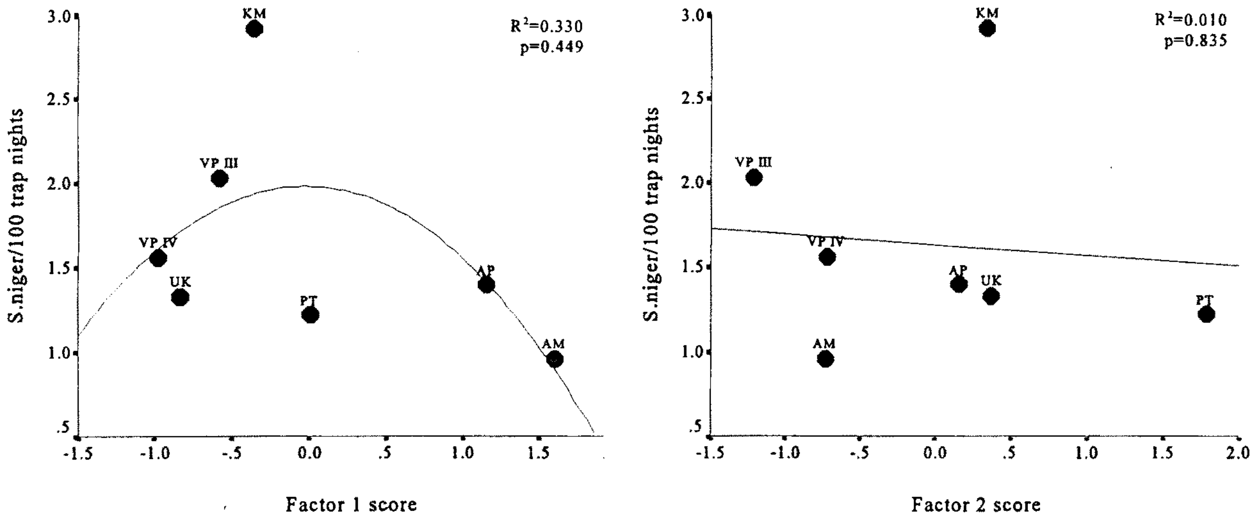


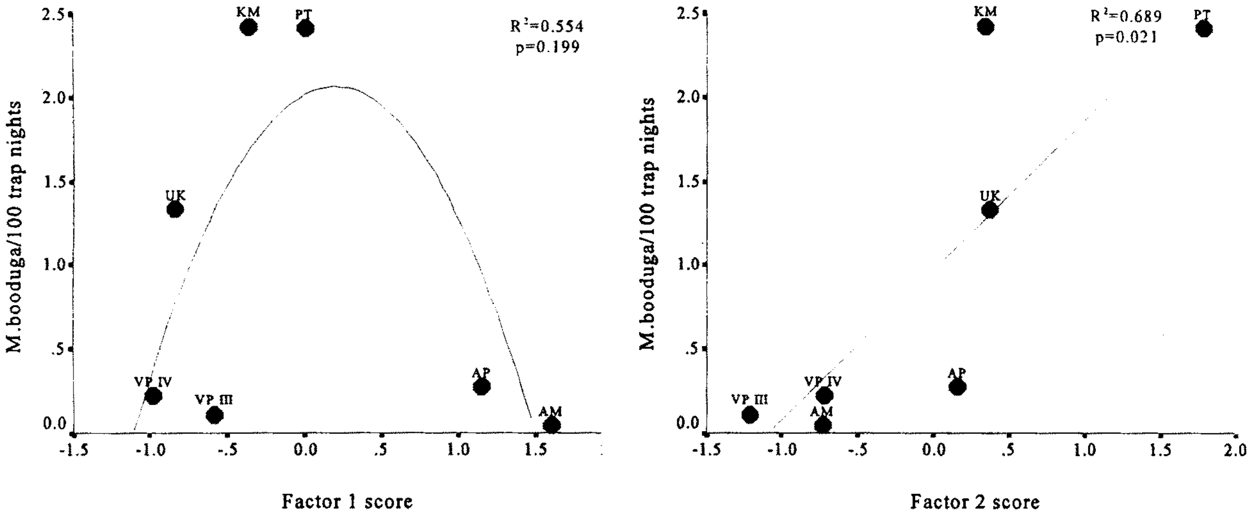
Figure 3.7. The abundance (no. of animals/100 trap nights) of *Rattus rattus* and *S. niger* in relation to Factor 1 and Factor 2 scores, with the best fit curves.

area (R^2=0.571; P=0.184) and fragment area (R^2= 0.512; P=0.239), even though both were not significant. Correlations were generally greater and linear with CVs of macrohabitat parameters, a linear increase with CV of undergrowth giving the highest R^2 (0.835; P=0.004), and a similar increase with CV of canopy height (R^2=0.737; P=0.013) both being statistically significant. Among the two factors, the linear increase with habitat heterogeneity (Factor 2) gave the highest R^2 which was significant (R^2=0.689; P=0.021). A quadratic relation with Factor 1 also gave a high R^2 (0.554), even though not statistically significant (P=0.199) (Figure 3.8).

The abundance of *M. musculus*, was only poorly correlated with the mean values of macrohabitat parameters (Table 3.7), a quadratic relationship with fragment area giving the best fit (R^2=0.448; P=0.262). Correlations with CVs were generally greater, a linear increase with CV of canopy height being significant (R^2=0.571; P=0.049), and a similar increase with CV of tree density being nearly significant (R^2=0.498; P=0.076). Among the two Factors, abundance of *M. musculus* increased linearly with Factor 2 which represented the habitat heterogeneity (R^2=0.730; P=0.014) (Figure 3.8).

Thus, the relationship of the abundance of individual species with mean values of habitat parameters, their CVs, and the Factors that represent vegetation status and habitat heterogeneity brings out striking differences among the species. Of the four species that were analysed, *S. niger* did not seem to be responding to any of the independent variables. *R. r. wroughtoni* decreased linearly with Factor 2 (habitat heterogeneity), while both *M. booduga* and *M. musculus* increased. *R. r. wroughtoni* and *M. booduga* had a quadratic relation with Factor 1 (vegetation status), the former reaching the lowest abundance in

a) *M.budooga*



b) *M. musculus*

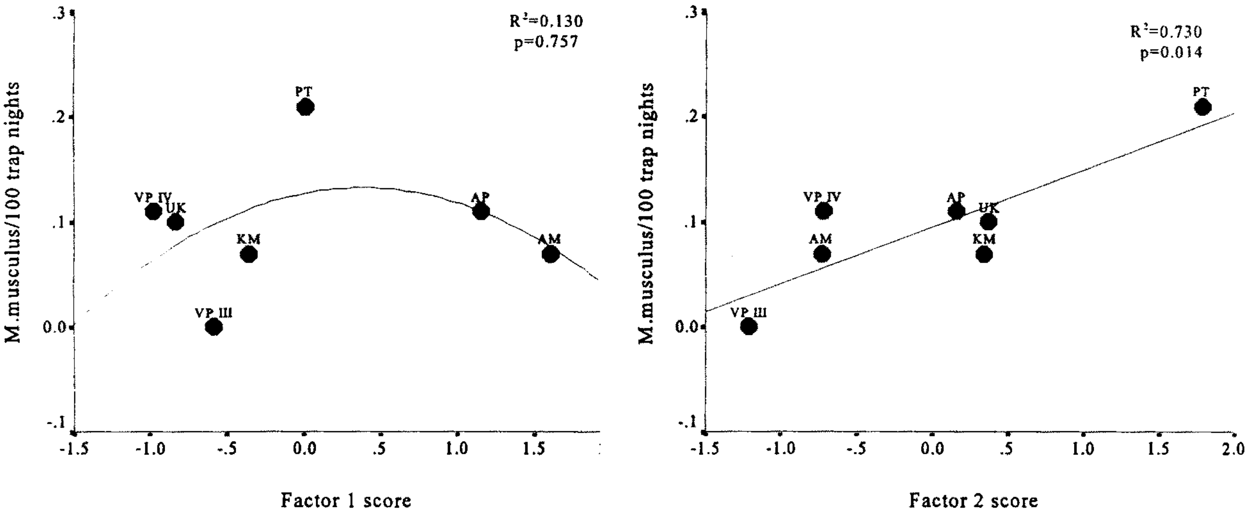


Figure 3.8.The abundance (no. of animals/100 trap nights) of *M. booduga* and *M. musculus* in relation with Factor 1 and Factor 2, with the best fit curves.

moderate values of Factor 1, while *M. booduga* had the highest abundance. *M. musculus*, was not sensitive to Factor 1. Similar relationship was observed for these three species with individual habitat parameters. All four species had quadratic relationship with the area of the fragment. *R. r. wroughtoni* and *M. booduga* also had similar but opposing relation with the mean values of many of the habitat parameters, while *M. musculus* and *S. niger* did not. The relationships with CVs of parameters were mostly linear, the abundance of *R. r. wroughtoni* decreasing with an increase in CV, while *M. booduga* and *M. musculus* increased.

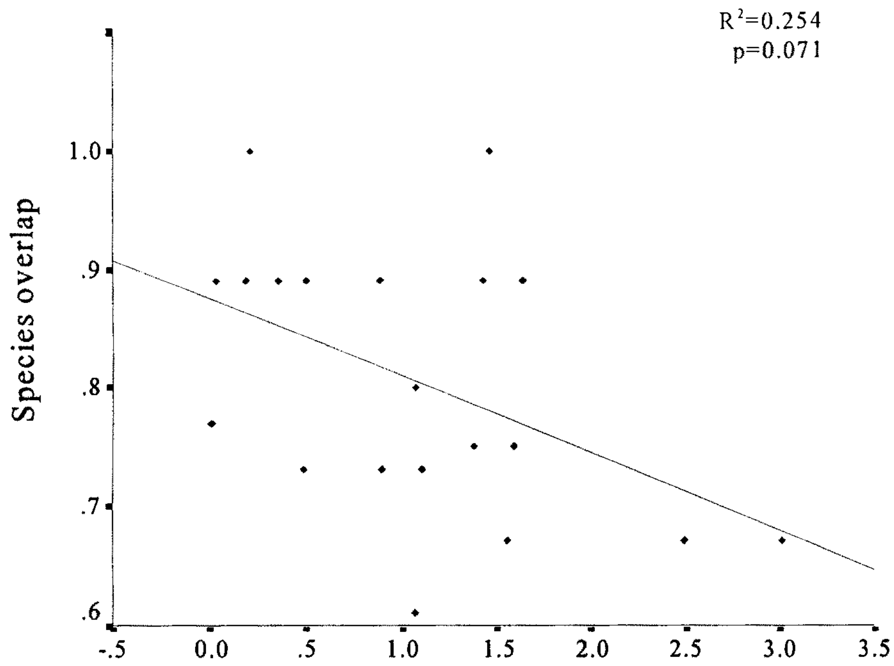
3.3.9 Species Overlap

I estimated the overlap or similarity in the occurrence of species between all pairs of fragments using Sorensen's index (1948), with seven fragments there were 21 fragment pairs. Since there were only eight species out of which five species occurred in almost all fragments overlap in the occurrence of species was generally high, the index varying only from 0.8 to 1.0 (the maximum possible overlap). This overlap was tested for correlation with the absolute difference between fragment pairs in the mean values of the macrohabitat parameters, their CVs and the two Factors. Because fragments generally had a high overlap in species occurrence, there were no significant correlations between the species overlap and differences in the macrohabitat parameters, the R^2 being close to zero in most cases. However, the largest fragment showed the lowest overlap with other fragments, and thus species overlap decreased as difference in the area increased ($R^2=0.338$; $P=0.024$). This was due to the presence of *P. lasiurus* and *V. oleracea* in the largest fragment, but not in the others. Once the largest fragment was removed, there was no such relationship. Similarly, difference in Factor 2 (habitat heterogeneity), also had a weak, nearly significant negative relationship with species overlap (R^2=0.254; P=0.071), due to two outlier fragment pairs (Puthuthottum-Akkamalai, and Varattuparai III - Puthuthottum). With these excluded the R^2 was close to near zero (Figure 3.9). The presence of human settlements nearby seemed a major factor, in deciding species overlap. Fragment pairs both of which were close to or away from human settlements had a greater species overlap (0.82)than if one was close to and the other away from settlements (0.79).

3.3.10 Similarity in Species Composition

The overlap in the relative abundance of species was estimated using Morisita index of similarity, for all possible 21 pairs of fragments. This index was tested for correlation with the absolute differences between fragment pairs in the mean values of macrohabitat parameters, their CVs, and the two Factor scores. The species composition overlap between fragment pairs was generally high, the index varying only between 0.60 and 0.900. This reflected the dominance of *R. r. wroughtoni* in all fragments, followed by *S. niger* and *M. booduga*. The variation in the index was mostly because of the remaining rare species.

The index was most correlated and negatively with the absolute difference between fragment pairs in Factor 2 score, with an R^2 of 0.290, which was significant (P=0.012) (Figure 3.10). Thus, fragments with similar habitat heterogeneity tended to have similar species composition. The other correlations were weaker, with R^2 ranging from 0.000 to only 0.093, and P values ranging from 0.179 to 0.932. This lack of correlation was most



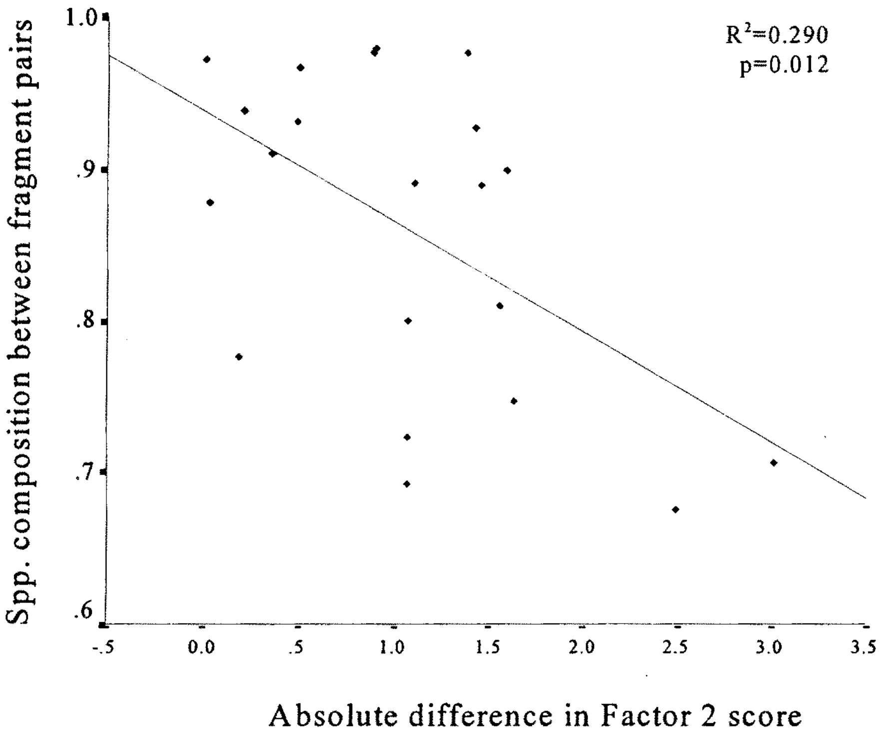


Figure 3.9. Species overlap (Sorensen's index) and species composition overlap (Morisita index) between fragment pairs in relation to absolute difference between them in Factor 2 score.

probably due to the presence of species that were caught rarely from few fragments, for example, *P. lasiurus* and V. *oleracea* only from Akkamalai, *G. ellioti* only from Puthuthottum and Korangumudi and *R. blanfordi* from only three fragments. As in the case of species occurrence overlap, the significant correlation of species composition overlap with the difference in Factor 2 was due to the presence of two outliers, which when removed, it reduced the R^2 close to zero.

3.4 DISCUSSION

3.4.1 Species Richness

Over one and a half years of trapping, I obtained only eight species, and only two endemics, *P. lasiurus* and *S. niger*. The largest fragment had all species, with *P. lasiurus* being limited only to it. However, seasonal species richness was greater in the medium sized fragments, as also was total abundance. More than the mean values of macrohabitat parameters, it is to habitat heterogeneity, measured by CVs of habitat parameters, that species seemed to respond more. Thus, *R. r. wroughtoni* decreased as habitat heterogeneity increased, while *M. booduga* and *M. musculus* increased, and *S. niger* showed no clear relationship. As a result, these species were more responsive to Factor 2, which measured habitat heterogeneity than Factor 1, that measured mean values of habitat parameters. Similarity in species composition between fragment pairs which was generally high was also correlated to differences in habitat heterogeneity between them, than to the mean values of habitat parameters.

Only seven species of rodents and one species of ground shrew were caught during this study in the Anaimalais. This is a reflection of the general faunal impoverishment among rodents and shrews in India in general, Western Ghats in particular. Even though Rodentia is by far the most species rich mammalian Order of the world (1,750 species), in India with only about 100 species, it is second to bats (110 species) in species richness. Species richness in Rodentia is particularly low in the Western Ghats. Out of 77 species in the Family Muridae reported from India, 61 species (79%) occur in north and northeast India, but only 16 (21%) in the Western Ghats. This faunal impoverishment is also true for ground shrews. In the Family Insectivora, out of 360 species reported worldwide (Anon. 1990), only 29 species occur in India (Kumar *et al.* 1998), of which only five have been reported from the Western Ghats (Kumar *et al.*1998). Studies elsewhere in the Western Ghats have also reported only few species of rodents and shrews. In the contiguous rain forests in the Kalakkad-Mundanthurai Tiger Reserve (KMTR), in the southern end of the Western Ghats, only five species were captured even after one year of intensive live trapping, covering 9, 613 trap nights (Anon. 1997). These included, two species of murids (*R. r. wroughtoni* and *Mus* spp.), the spiny dormouse, and one species of shrew (*Suncus* spp.) and *Funambulus tristriatus*. In the deciduous forests in the Mudumalai Wildlife Sanctuary, in the Nilgiri Hills, only four species were trapped in the dry season (Meena 1997). In the montane forest fragments and surrounding grasslands in the upper Nilgiris, nine species were trapped after three years of intensive trapping effort, covering about 20,000 trap nights (Shanker 1998). One species, the metad (*Millardia meltada*), was trapped only from the grasslands. Some species were common with the present study (*R. r. wroughtoni, R. blanfordi, M. booduga* and *P. lasiurus*), some were not captured in the present study (*Suncus montanus, S. dayi, Mus famulus, M.*

platythrix, and *Millardia meltada*), while some captured in this study were not captured in the Nilgiri's (*S. niger* and *M. musculus*). These studies as well as other studies in the Western Ghats show the general impoverishment of the rodent and shrew fauna in this area. Some species are common to all study areas. These included *R. r. wroughtoni, P. lasiurus, M. booduga* and *R. blanfordi* and thus probably widely distributed in the Western Ghats. Those that are not common include, *M. musculus, M. famulus, S. niger, S. dayi* and *Suncus* spp. (in KMTR). This might reflect real differences in the distribution of species or taxonomic uncertainties that beset this taxa in India.

It is interesting, however, that studies in southeast Asia which have considerably more number of species, have also reported few species from capture studies. For instance, five species in China (Wu *et al.* 1996); seven species in Thailand (Walker and Rabinowitz 1992). In contrast 10-17 species have been reported from temperate areas (Brooks and Healy 1988; Raphael 1988; Gore 1988; Sieg 1988), neo-tropics (Fonseca and Kierulff 1988; Stallings 1988; Iyawe 1989; Malcolm 1991), and Africa (Kerley 1992; Wirminghaus and Perrin 1993; Mares and Ernest 1995; Woodman *et al.* 1995; Kelt 1996; Monadjem 1997). Two species of insectivores and 12 rodents have been reported from the semi arid Aravalli Range in north India (Prakash *et al.* 1995). This might reflect a greater species richness in these areas, or differential trappability or methods.

It is very likely that even among the eight species trapped in the Anaimalais only a few are native to these forests, and the others have been introduced by man. Among the former is *P. lasiurus*, an endemic to the Western Ghats. This was caught only from the largest and least disturbed fragment (Akkamalai shola), that too only in the first wet season. It is more widely distributed than previously thought (Shanker 1996; Prabhakar 1997). It was infrequently caught in the Upper Nilgiris (Shanker 1998), as in this study. In contrast, in the contiguous and undisturbed rain forest of Kalakkad and Mundanthurai Tiger Reserve, it was the second most abundant species. Thus, habitat fragmentation may be a threat to the dormouse. *P. lasiurus,* however, has been reported to be a pest in the pepper and arecanut orchards along the foothills in Kerala, where the species is known as pepper rat (Rajagopalan 1968).

Even though five species of shrews have been reported from the Western Ghats, at least three of them being endemic, their field identification and taxonomy are far from clear. The species reported from the upper Nilgiris (above >1,500 m) is *S. montanus* (Shanker 1998). No shrews were caught in the Anaimalais by Chandrasekar-Rao and Sunquist (1996), while the species caught in KMTR remain to be identified. The species that I caught was identified as *S. niger* following Ellerman (1961) and Sterndale (1982). The distribution of *S. dayi* is not known. The fifth species of shrew (*S. murinus*), is widely distributed, which might also occur within the study area in human settlements.

R. r. wroughtoni is endemic to Western Ghats at the subspecies level, but the species is widely distributed in India. This was the most dominant species in all fragments, in almost all seasons, forming over 30% of the animals trapped in almost every session. This subspecies was even more dominant forming 80% of the captures in KMTR (Anon. 1997). In the Upper Nilgiris it constituted >75% of the captures (Shanker 1988) and in another part of the Anaimalais 57% of the captures (Chandrasekar-Rao and Sunquist 1996).

Among the other five species, two were caught frequently (*M. booduga* and *M. musculus*), while three were caught infrequently (*V. oleracea, R. blanfordi* and *G. ellioti*). All these species have also been caught in the Upper Nilgiris, and the Anaimalais before. However, these have not been caught in the undisturbed and contiguous forests in KMTR (Anon. 1997), where one unidentified *Mus* spp has been caught. It is likely that the occurrence of these species in the rain forest fragments (and other habitats) is largely due to human settlements nearby and associated habitat disturbance. These are thus likely to be invaders. The invasion of forest fragments by several generalists or human introduced species, is a major consequence of habitat fragmentation (Raphael 1988; Bennett 1990; Soulé *et al.* 1992; Wirminghaus and Perrin 1993; Kattan *et al.* 1994; Probert and Litvaitis 1996). *M. musculus* and *R. rattus* are in fact the most common invaders into disturbed forests elsewhere also (Bennett 1990; Soulé *et al.* 1992).

Seasonal species richness was linearly related to habitat heterogeneity (as indicated by the CVs) which was highest in the medium sized fragments. Thus mean seasonal species richness was highest in the medium sized fragments. Habitat heterogeneity (both vertical and horizontal) is a major factor that governs species richness in birds (MacArthur and Wilson 1967). In a tropical rain forest, one of the factors that could increase habitat heterogeneity in terms of tree density, basal area and canopy cover is moderate disturbance. The heterogeneity could be expected to be greatest in the medium sized fragments. In the large undisturbed forest as well as the heavily disturbed small fragments, heterogeneity is low. The former is more or less spatially uniform with high tree density, basal area, canopy height and canopy cover, while the latter have low values. In the moderate sized fragments, spatial variation is high. This was clear from the quadratic relation between fragment area and Factor 2 that measured habitat heterogeneity. Factor 1 which measured mean values of habitat parameters was linearly related to fragment area.

The greater species richness in the moderate sized fragments with high habitat heterogeneity, was due to the invasion of several commensal species, and is facilitated by the presence of human settlements near fragments. An increase in the number of species due to moderate disturbance has been reported else where also (Bierregaard *et al.* 1992; Russel-Smith and Bowman 1992; Andrén 1994). In Western Ghats itself, bird species richness is greater in moderately disturbed rain forest (Vijayan *et al.* 1998). It was clear from this study that this invasion is not a function of area, but habitat heterogeneity, resulting from disturbance.

3.4.2 Abundance

The abundance values reported for rodents and insectivores based on capture studies are as follows: in the upper Nilgiris 10.6% (Shanker 1998), in KMTR 5.6% (Anon. 1997) and in the northern part of Anaimalais 5.1% (excluding *Funambulus tristriatus*) (Chandrasekar-Rao and Sunquist 1996). In this study, the total abundance of rodents was greatest in the medium sized fragments. This is because of the contrasting relationship between the abundance of the four most common species and habitat parameters. The abundance of *R. r. wroughtoni* decreased with the increasing habitat heterogeneity, while that of *M. booduga* and *M. musculus* increased, and *S. niger* showed no relationship. The other species which were trapped infrequently (*V. oleracea, R. blanfordi* and *G. ellioti*) are also commensal species, these are expected to show a pattern similar to that of *M. musculus* and *M. booduga,* increasing with habitat heterogeneity, which was greatest in the medium sized fragments. Therefore, the greater seasonal species richness and greater abundance were observed in the medium sized fragments. However, total abundance showed weaker relationships with habitat parameters, their CVs and the two Factors, compared to that shown by individual species that I examined. This was probably due to the inclusion of several species that were trapped only infrequently.

Many previous studies have examined the relationship between habitat parameters and small mammal abundances, and many of these have used derived variables (principal components) (Canova and Fasola 1991; Laurance 1994; Shanker 1998). However, very few have included spatial variation in the analysis, even though the importance of habitat heterogeneity in small mammal abundance has been well known at landscape and lower levels (*e.g.*, Kozakiewicz 1983, 1993; Soulé *et al.* 1992).

As this study shows, horizontal habitat heterogeneity could be an important factor that governs the abundance of small mammals. Unlike the murids, the shrew *S. niger*, did not seem related to any of the parameters that I examined, including the two Factors. The abundance of the species was considerably greater (1.52/100 trap nights) compared to the *Suncus* spp. in KMTR (0.07/100 trap nights) (Anon. 1997), whereas the abundance of *S. montanus* in upper Nilgiris was greater (2.5/100 trap nights) (Shanker 1998). Since shrews are primarily insectivores, it is likely that they respond to a different set of macrohabitat parameters compared to rodents. For example, an increase in ground insect abundance with a decline in the canopy insects occur in the forest fragments, compared to contiguous forest (Malcolm 1991). This can cause an increase in the density of terrestrial insectivorous shrews.

As in this study, an increase in the abundance of edge species has been reported in forest fragments elsewhere. The loss of species from fragments, has also been associated with isolation mediated habitat changes, rather than to fragment area or isolation *per se* (see VanDorp and Opdam 1987; Martin *et al.* 1995). The absence of *P. lasiurus*, from the smaller fragments, is also likely to be due to habitat changes rather than area *per se*. In KMTR, *P. lasiurus* was the second most abundant species (0.21 animal/100 trap nights), in contrast to only 0.057 animals/100 trap nights in the forest fragments. All the six animals were caught from the largest fragment. They have been caught from sites with greater number of lianas and buttress trees (Anon. 1997). It is very likely that the disappearance or low abundance of this species from forest fragments is due to the loss of their microhabitats from logging (which removes buttress trees and lianas), rather than to a reduction in the area *per se*. It is noteworthy that in England, the disappearance of the dormouse (*Muscardinus avellanarius*) has been associated with the loss of specific microhabitats, rather than a reduction in the area (Bright and Morris 1996).

3. 4. 3 Species Composition and Overlap

A comparison of the relative abundance of *R. r. wroughtoni* in the forest fragments with that reported from KMTR and upper Nilgiris show the consistent reduction in the percentage of this species in the small mammal community. In KMTR, this species formed as much as 80% of total captures (Anon. 1997); whereas in the upper Nilgiris

(Shanker 1998) it formed more than 75% of the captures. In the man made forest fragments, the relative abundance of this species was greatest in the largest fragment 54% (Akkamalai shola), and showed a steady decline with declining fragment area and increasing habitat heterogeneity. This is counter balanced by an increase in the abundance of *M. booduga* and *M. musculus* from 0.7% in the largest fragment to 9% in the moderately disturbed medium sized fragments. *S. niger* on the other hand has retained the same relative abundance in all fragments.

Whether the changes in species composition that follow habitat disturbance or fragmentation are due to changes in macro and microhabitats, or competition has been a matter of considerable debate (Canova and Fasola 1991; Adler 1994). The loss of specific microhabitats due to disturbance and habitat fragmentation, has been reported in the case of dormouse (*Muscardinus avellanarius*) in England (Bright and Morris 1996). On the other hand, the invasion by species that are better adapted to the altered habitat in fragments, can competitively displace resident species (Bierregaard *et al.* 1992; Russel-Smith and Bowman 1992). The differences among species in microhabitat selection are discussed in detail in Chapter 5.

Invasion by predators to which the resident species are ill-adapted, and invaders that are better adapted, can also cause changes in the species composition (Johns 1985). For example, the abundance of fallen logs in fragments seemed to provide better cover for *M. booduga* than perhaps for *R. r. wroughtoni*, from predators such as feral cats and raptors. Invasion by parasites and pathogens to which resident species are not adapted can also alter species composition.

Another major factor that facilitates the changes in species composition in fragments is the presence of human settlements, which might serve as a source for the dispersal of commensal species (such as *M. booduga* and *M. musculus*) into the forest fragments. This is indicated by the greater species overlap among fragments that were close to the human settlement and among fragments that were away from them..

Since there were only eight species, many of which occurred in all fragments, the similarity between fragment pairs in the species occurrence was generally high and not related to differences in any habitat parameters. The similarity in the relative abundance of species was also high because *R. r. wroughtoni* (50% of the capture), *S. niger* (29.27%) and *M. booduga* (15.45%) were the most dominant species in most of the fragments. The species caught only from a few fragments were also caught infrequently. *R. r. wroughtoni* seems to be the most dominant species in the forests of Western Ghats, in KMTR where it formed 80% of the captures (Anon. 1997), in the northern part of Anaimalais (57.54%) (Chandrasekar-Rao and Sunquist 1996), and in the upper Nilgiris (>75%) (Shanker 1998). The second dominant species was *S. niger* in this study, the upper Nilgiris had a *Suncus* spp. (23.66%) (*op.cit.*), but in the undisturbed rain forest in KMTR, the second dominant species was *P. lasiurus* (10%), *Suncus* spp. forming the third dominant species with only about 4% of the captures. Studies elsewhere have reported the dominance of the rodent and shrew community by one species (often a murid) that forms nearly (30%) of the captured individuals (Gore 1988). In many areas, the second or third species is a shrew (Brooks and Healy 1988; Raphael 1988). In many sites, however, the dominance is shared by a number of species (Malcolm 1991; Laurance 1994).

3.5 SUMMARY

1. In this chapter I examined the differences among forest fragments in the species richness and abundance of rodents and shrews. I attempted to identify the macrohabitat parameters that govern the differences among fragments.

2. The data come from nearly 10,595 trap nights in seven fragments of varying area and disturbance levels. Several micro and macrohabitat parameters also measured in these fragments. A PCA identified two factors that accounted for 77% of the total variance in macrohabitat parameters and their CVs Factor 1 represented the status of the fragment in terms of area, tree density *etc.*, while Factor 2 represented spatial heterogeneity. Factor 1 was linearly related to area of the fragment while Factor 2 had a quadratic relation.

3. Seven rodent and one shrew species were trapped. The maximum number of species (seven) were caught in the very large fragment (Akkamalai). One endemic species was also caught only in that fragment . *R. r. wroughtoni*, *S. niger*, *M. booduga* were caught in all fragments and *M. musculus* caught in all fragments, except in Varattuparai III. *R. blanfordi* was caught in three fragments and *G. ellioti* in two. The most abundant species *R. r. wroughtoni* comprised 50% of the total captures, followed by *S. niger* (30%), *M. booduga* (15%) and *M. musculus* (1.76%). The other species together formed less than 4% of the overall captures.

4. Seasonal species richness showed a non-linear relationship with some of the habitat parameters and Factor 1, and a linear relationship with Factor 2. The mean seasonal species richness was highest in medium sized fragments, which were moderately disturbed and had the highest spatial heterogeneity. This is due to the greater abundance of invaders which are commensal to man.

5. The total abundance of rodents and shrews did not show any clear relationship with habitat parameters, though, individual species did. Thus, the abundance of *R. r. wroughtoni* increased with Factor 1 and decreased with Factor 2, while *M. booduga* and *M. musculus* showed an opposite relation. *S. niger* showed no relationship.

6. Species overlap between fragment pairs was generally high and was not related to similarity in habitat parameters. The proximity of human settlements seemed to affect similarity in species occurrence among fragments. The species composition overlap between fragment pairs was also high, and was correlated negatively with the absolute differences between fragment pairs in Factor 2 score. Thus, fragments with similar spatial heterogeneity had similar species composition.

7. It is concluded that among the species trapped, the endemic *P. lasiurus* may be the most affected by habitat fragmentation. Another endemic subspecies *R.r. wroughtoni* also is adversely affected, however, this species widely distributed, occurring even in coconut and arecanut plantations. An important factor that govern species occurrence and abundance is habitat heterogeneity, to which species respond differently.

4. DEMOGRAPHY

4.1 INTRODUCTION

Impacts of forest fragmentation on fauna have been mostly assessed in terms of species richness and diversity, and community structure, which were discussed in Chapter 3. Changes in species richness and composition are ultimately linked to the demographic processes that take place in forest fragments and the landscape around them. The important demographic processes in this context are changes in age structure, adult sex ratios, reproductive and survival rates, and dispersal. Surprisingly, very few studies have addressed the demographic consequences of habitat fragmentation on small mammals(*e.g.*, Gliwicz 1980; Bujalka 1986; Adler and Seamon 1991; Kozakiewicz 1993; Adler and Levins 1994; Adler 1996; Shanker 1998). These studies have, however, reported differences among species that are consistent with the ultimate fate of the species in fragmented landscape. The matrixes that surround the forest fragments often function as a filter that allows some species to disperse and not others (Kozakiewicz 1993). Many rodents have been reported to be capable of using and dispersing over long distances across a mosaic of habitats in a fragmented landscape (Kozakiewicz and Jurasińska 1989; Jurgiel 1992; Kozakiewicz 1993). These species have a greater chance of persistence as a metapopulation in a fragmented landscape (Bierregaard *et al.* 1992). On the other hand, isolated populations of some species seldom leave their habitat fragments, and are thus more susceptible to environmental fluctuation (Kozakiewicz 1993). Fluctuations in demography occur in accordance with changing environmental conditions even among populations that live in habitat fragments. In the low rate or absence of dispersal, population fluctuations in small fragments often track variation in resource abundance at a local level, and not at a regional scale as has been reported to occur in unfragmented population (Steen *et al.* 1996). Habitat fragmentation would thus promote asynchrony among populations (Adler 1994, 1996: Shanker 1998).

In addition, small fragments show greater environmental stochasticity which might further promote demographic asynchrony. Another factor that might promote asynchrony is demographic stochasticity to which small population are highly susceptible (Gilpin and Soulé 1987). Predation is another factor which affects population fluctuation (Soulé *et al.* 1992; Smith and Quin 1996; Shanker 1998). The lack or low rate of dispersal, the asynchronous variation in resource abundance among fragments, greater environmental and demographic stochasticity, and differences in predation levels, would also affect demographic parameters such as breeding seasonality, age structure and adult sex ratios.

Body weight has been reported to increase in island population due to lack of predations (Foster 1964; Angerbjorn 1985), and competition (Angerbjorn 1985). Very few studies have examined the effect of habitat fragmentation on body weight, which may not be comparable to that in islands. Shanker (1998) found that body weight of *R. r. wroughtoni* was greater in forest fragments and old wattle plantations compared to young plantations.

Adler (1994) reported that on islands in Panama, the spiny rat bred regardless of seasons and that the sex ratio fluctuated out of phase among populations. This was attributed to the difference in the floral compositions among fragments. In Poland, Kozakiewicz and Jurasińska (1989) trapped a higher percentage of sexually active males with fewer reproducing females, and underweight bank voles in wooded fragments than from continuous forest. In the upper Nilgiris in India, the proportion of adults and the mean body weight in *Rattus rattus* were negatively correlated with fragment size (Shanker 1998). Adler (1994) concluded that the fragmented tropical rodent populations have more complex dynamics and show less synchrony than temperate populations.

A lower reproductive rate, and a greater proportion of adults have been reported among rodents in habitat fragments (Gliwicz 1980). Changes in sex ratios could also occur due to a greater predation of males when they disperse from fragments (Bujalka 1986; Jurgiel 1992). When compared with mainland populations, differences were observed in island populations in population dynamics, breeding biology and other natural histories of rodents (Gliwicz, 1980). In this chapter I examine the variation in some demographic parameters in rodents and shrews in forest fragments, and attempt to identify the macrohabitat parameters that influence this variation.

The objectives in this chapter are:

a) to examine the seasonal differences in the age/sex composition in the three most abundant species of small mammals (*R. r. wroughtoni, S. niger* and *M. booduga*),

b) to assess the differences in the age/sex composition of these species among the fragments in relation to macrohabitat parameters; and

c) to examine differences in body weight of these species among seasons and fragments.

4.2 METHODS

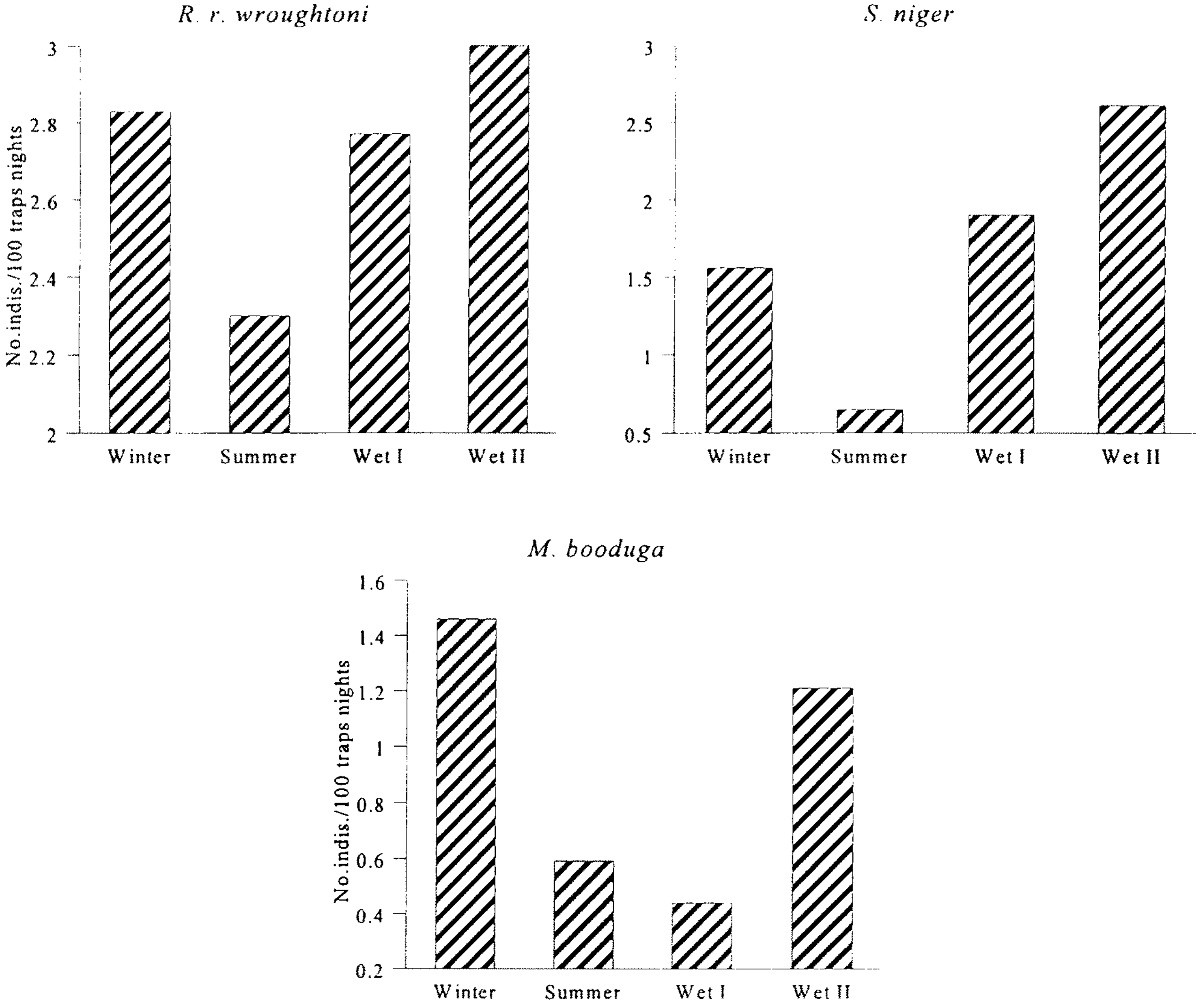
Animals trapped were anaesthetized with chloroform. Later they were weighed using standard weighing scale. Morphometric measurements such as head, body, ear, hind foot and tail lengths were measured. I visually assessed the sex and reproductive condition, since the adult males have conspicuously descended testes and females have perforated vagina and prominent nipples. Animals with prominent nipples or descended testes were considered as 'adult', and others grouped as 'sub adults'. Sexing was sometimes difficult in *S. niger*, however, a gentle press in the lower abdomen region extruded the penis. In addition, age was also assessed based on body weight and colouration of the coat.

4.3 RESULTS

4.3.1 Synchrony in Abundance

The factors affecting variation among the fragments in the total abundance of rodents and shrews and that of individual species were discussed in Chapter 3. In this section I discuss seasonal variation and synchrony among fragments in total abundance and the abundances of three most common species (*R. r. wroughtoni, S. niger* and *M. booduga*) on which there was sufficient data for analysis.

All fragments together, the abundance of *R.r.wroughtoni* was lowest in summer, and highest in wet II (Figure 4.1). For *S. niger*, abundance was highest in wet II, followed by wet I, and for *M. booduga*, highest in winter, and lowest in wet I. *M. musculus* was



R. r. wroughtoni

S. niger

M. booduga

Figure 4.1 Seasonal variation in capture rates of three species of small mammals.

caught in only two seasons, summer and wet I, the former having the highest abundance. Other species were caught in only few seasons. All fragments together, seasonal difference was significant only for *S. niger* (χ^2=7.84; *df*=3; P=0.05), and not for *R. r. wroughtoni* (χ^2=2.42; *df*=3; P=0.441) and *M. booduga* (χ^2=1.55; *df*=3; P=0.67).

Each fragment showed considerable variation in the seasonal abundance of each species (Table 3.8 & 3.9). The mean of CV of seasonal abundance in a fragment for *R. r. wroughtoni* was 56.28% (range 25.82% - 85.8%), 65.48% for *S. niger* (22.07% - 99.40%), and 127.10% for *M. booduga* (27.18% - 200%). Thus, even though there was no significant seasonal difference in the abundance when fragments were pooled (except for marginal significance in *S. niger*) each fragment showed considerable seasonal difference. When a χ^2 test was done for each fragment separately, seasonal difference was significant in Varattuparai IV (χ^2=14.24; *df*=3; P<0.01), Puthuthottum (χ^2=8.13; *df*=3; P<0.05), Korangumudi and Andiparai shola (χ^2=28.10; *df*=2, χ^2=22.64; *df*=3 respectively, P<0.001) for *R. r. wroughtoni*. In *S. niger* a marginal significant seasonal difference was observed in Varattuparai III (χ^2=7.58; *df*=3; P<0.05), where as it was highly significant in Korangumudi (χ^2=19.63; *df*=2; P<0.001) and Akkamalai (χ^2=29.16; *df*=3; P<0.001). For *M. booduga*, seasonal comparisons were possible only for Puthuthottum, Korangumudi and Andiparai shola, because there were not enough captures in some seasons in other fragments. Of these fragments, Korangumudi showed a significant difference (χ^2= 17.13; *df*=2; P<0.001).

The coefficient of variation (CV) of seasonal abundance in a fragment was not related to Factor 1 or Factor 2, in the case of *R. r. wroughtoni* and *S. niger*. In *M. booduga* the seasonal variation seemed to have quadratic relation with Factor 1 (R^2= 0.70; *df*=4; P=0.089), with seasonal variation being lowest in the medium sized fragments. It also decreased as Factor 2 or habitat heterogeneity increased (R^2=0.390; *df*=4; P=0.373). Thus, seasonal fluctuation in population was not related to habitat parameters in *R. r. wroughtoni* and *S. niger*, while in *M. booduga* it was lowest in the medium sized fragments, and also decreased with habitat heterogeneity.

The lack of overall seasonal difference in spite of many fragments showing such a difference was due to the lack of synchrony among fragments in seasonal variation. Thus for *R. r. wroughtoni*, the highest abundance was in winter in three fragments, in wet II in three fragments, and in wet I in one fragment. The season with the lowest abundance also varied among the fragments. There was thus no synchrony among fragments in the seasonal abundance of *R. r. wroughtoni* (Kendall's coefficient of concordance W=0.155; χ^2=3.26; P=0.353). In *S. niger*, the lowest abundance was in summer in three fragments, and in winter and wet I, in two fragments each. The highest abundance was in wet II in four fragments, and in winter in two fragments. There was thus better synchrony among fragments in seasonal abundance of *S. niger*, the Kendall's coefficient of concordance being significant (W=0.387; χ^2=8.13; P=0.043) (Figure 4.2). Thus, when data on this species was pooled among fragments, the seasonal difference was nearly significant.

M. booduga was not caught in some seasons in many fragments. The season of highest abundance was winter for three fragments, and wet II and summer for two fragments each, there being no synchrony among the fragments (W=0.257; χ^2=5.40; P=0.145) (Figure 4.2).

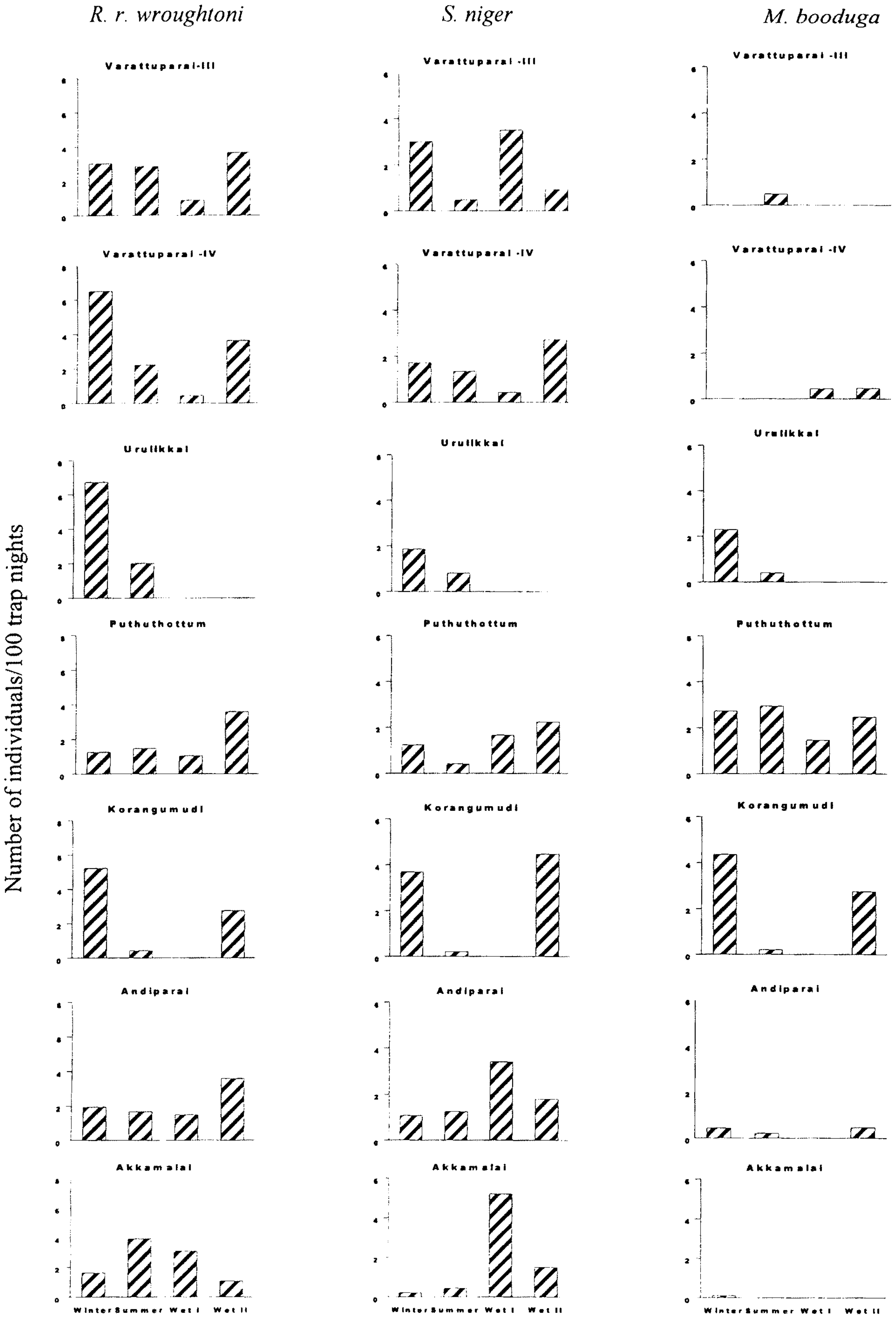


Figure 4.2 Seasonal abundance of three small mammals in seven forest fragments.

There was thus no synchrony among the fragments in the seasonal abundance of *R. r. wroughtoni* and *M. booduga*, and only a marginal synchrony in *S. niger*. This was in spite of considerable variation in seasonal abundance within each fragment. In the case of *M. booduga* the fragments with high habitat heterogeneity showed the least seasonal variation.

4.3.2 Age Structure

All fragment together, the ratio of subadults to adults was highest in wet Season I in *R. r. wroughtoni* (0.88), *S.niger* (0.42) and *M.booduga* (0.93), but the seasons with the lowest ratio were different, in summer for *R.r. wroughtoni* (0.21), and in winter for *S.niger* (0.04) and *M. booduga* (0.21). Thus, all three species had greater proportions of subadults in the wet season I (Figure 4.3), followed by wet season II. Proportion of adults and subadults differed significantly among the seasons in *R. r. wroughtoni* (χ^2=14.10; *df*=3; P<0.01). This was largely due to the greater proportion of subadults in wet season I, compared with winter and summer months (partitioned χ^2 test=11.1; *df*=1; P<0.01), there being no difference between the latter two seasons (partitioned χ^2 =1.15; *df*=1; P<0.05). Thus, in *R. r. wroughtoni* the proportion of subadults was significantly greater in wet season I compared to the other seasons.

The seasonal difference in the ratio of subadults to adults was significant also in *S.niger* (χ^2=12.50; *df*=3; P<0.01), that of wet season I being significantly greater than winter and summer (partitioned χ^2 test=12.48; *df*=1; P<0.001). Moreover, the ratio in wet season II was significantly greater than winter and summer (partitioned χ^2 test =9.4; *df*=1; P<0.01).

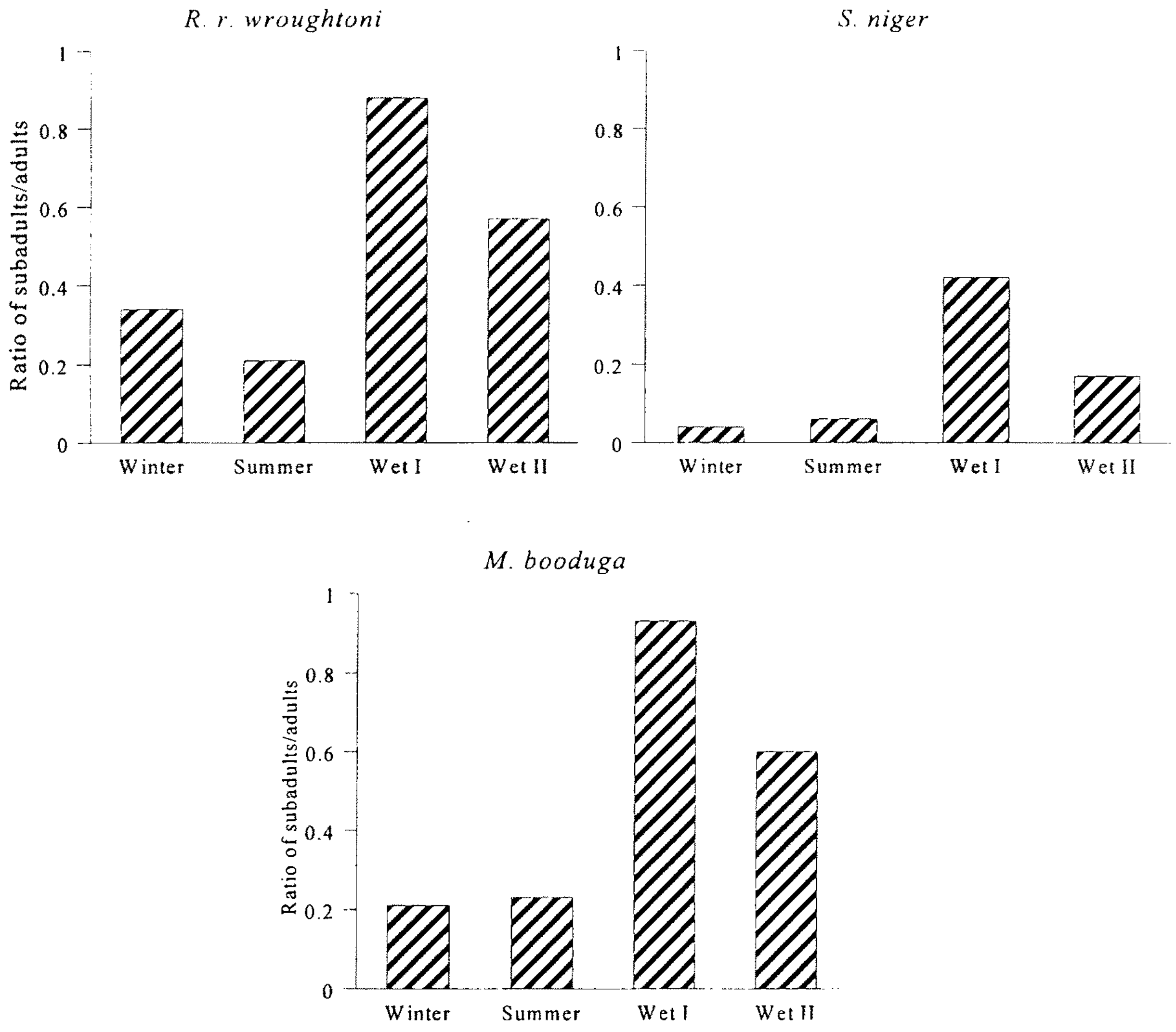


Figure 4.3 Seasonal differences in subadults/adults ratio in *R. r. wroughtoni*, *S. niger* and *M. booduga* in the study area.

Thus, the ratio of subadults was significantly greater in the wet season I and wet season II. The ratio differed significantly among the seasons also in the case of *M.booduga* (χ^2=8.90; *df*=3; P<0.05); again the ratio in the wet season I was significantly greater, compared to winter and summer (χ^2 partitioned test=8.5; *df*=1; P<0.01).

A comparison of the age ratio with the fragment area was done for the three species. Since captures were few, for comparison the fragments grouped into four area classes (small, medium, large and very large), and data was pooled for all seasons. In *R. r. wroughtoni*, there was no difference among fragments of different sizes. In *S. niger*, the highest ratio was in small and lowest in large, whereas in *M. booduga* the highest in medium and lowest in small. Animals from the very large fragment were not sexed (Figure 4.4).

4.3.3 Adult Sex ratio

All fragments together, the sex ratio (adult females/adult male) in *R. r. wroughtoni* was highest in winter (1.70), and lowest in wet II (0.81). In *S.niger*, it was highest in summer (2.30) and lowest in wet I (1.06), and in *M. booduga*, it was highest in the wet seasons (1.70) and lowest in winter (0.50) (Figure 4.5). In general, the ratio decreased in *R. r. wroughtoni* from winter through summer to wet season I. The sex ratio of *R. r. wroughtoni* and *S. niger* did not vary significantly among the seasons (χ^2= 6.2; P=0.10 and χ^2= 2.9; P=0.30; *df*=3, respectively). In *M. booduga*, the sex ratio was lowest in winter and through summer, wet I and wet season II, the difference being significant (χ^2

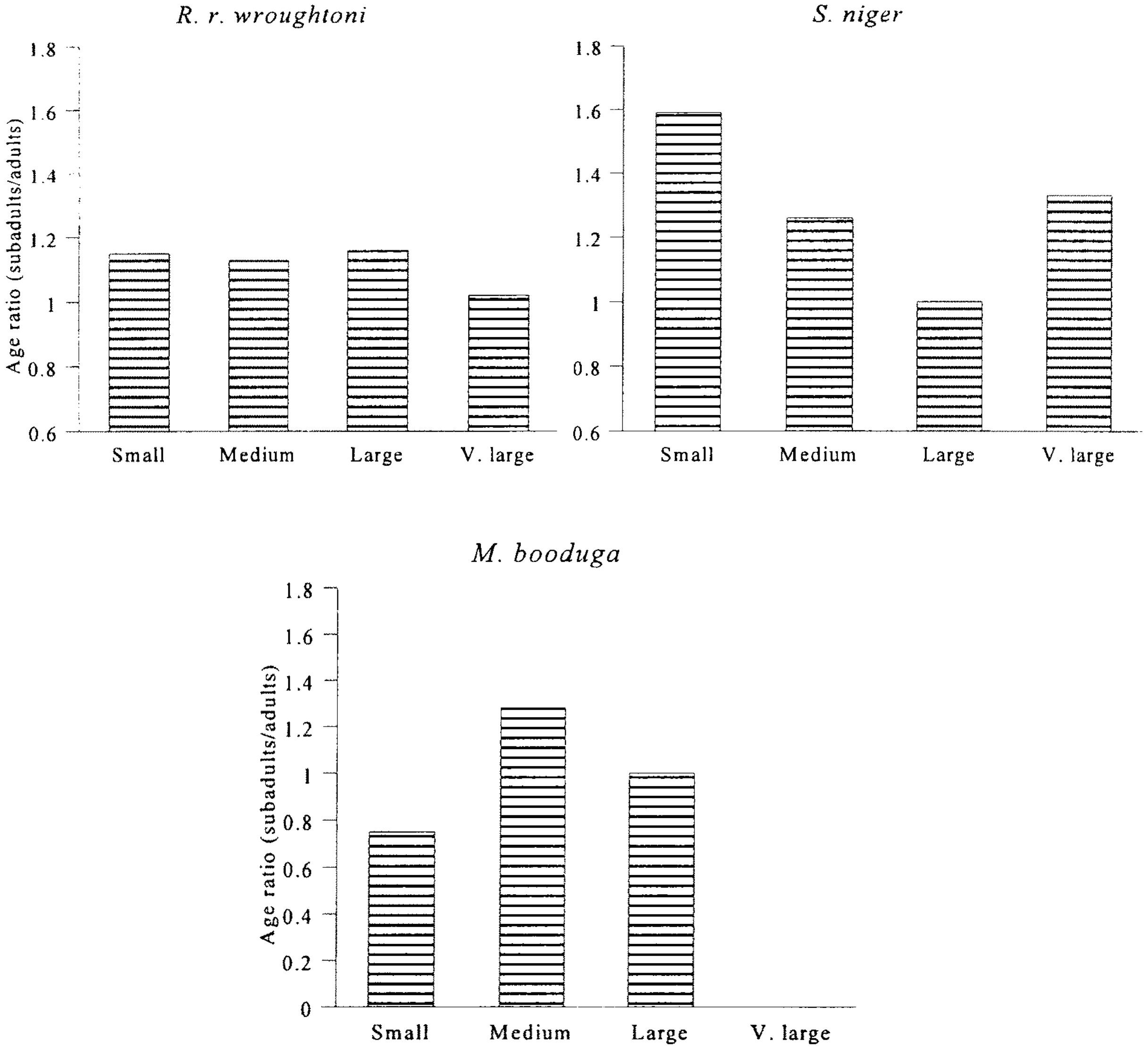


Figure 4.4 The age ratios of subadults/adults in *R. r. wroughtoni*, *S. niger* and *M. booduga* among fragments of four size classes.

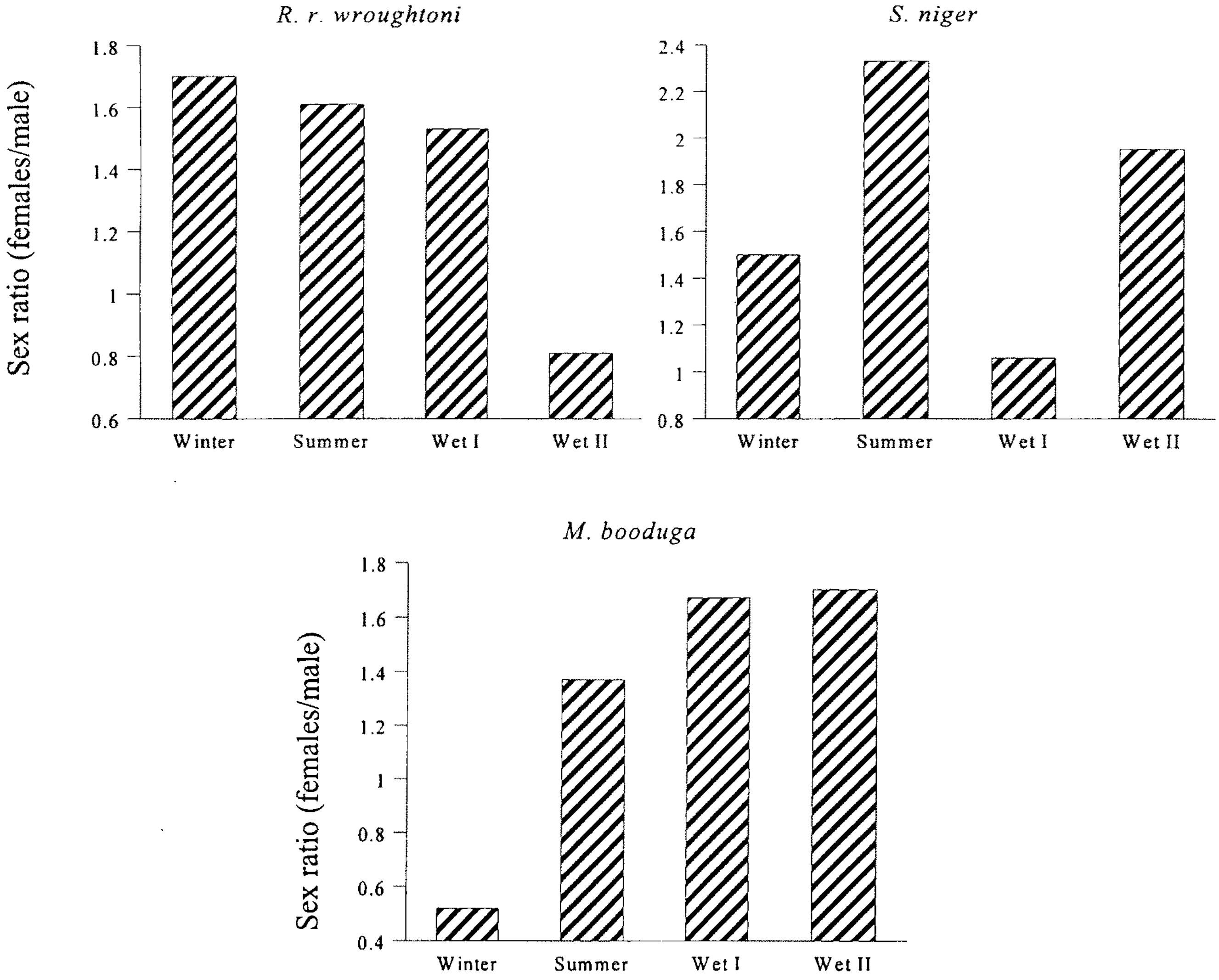


Figure 4.5 Seasonal differences in adult sex ratio (females/male) of *R. r. wroughtoni*, *S. niger* and *M. booduga*.

=7.5; *df*=3; $P<0.05$). *R. r. wroughtoni* and *M. booduga* had high sex ratios in three seasons (between 1.4 and 1.7), but had a very low ratio in one season, wet II in *R. r. wroughtoni* (0.8), and winter in *M. booduga* (0.50).

An examination of the adult sex ratio with reference to fragment area (in four classes) did not show any pattern for *R. r. wroughtoni* and *M. booduga*. In *M. booduga*, the sex ratio decreased with fragments area (Figure 4.6).

4.3.4 Seasonal Synchrony in Age/Sex Ratios

The number of animals caught from each fragment in a season was low for all species. Therefore, it was not possible to examine seasonal synchrony among fragments in the ratio of subadults to adults, and adult sex ratios. Fragments were grouped into small, medium, large and very large to examine whether fragments had seasonal synchrony in age and sex ratios.

There was no seasonal synchrony in adult sex ratios of *R. r. wroughtoni* among fragments (Figure 4.7) ($W=0.083$; $\chi^2=1.00$; $P=0.801$). In the case of the large and very large fragments an opposite trend was apparent. There was a greater synchrony in *S. niger*, among small, medium and large fragments with a decline in the ratio from winter to summer, an increase from summer to wet I, and then a decline to wet II season, the synchrony being nearly significant ($W=0.500$; $\chi^2=6.00$; $P=0.112$).

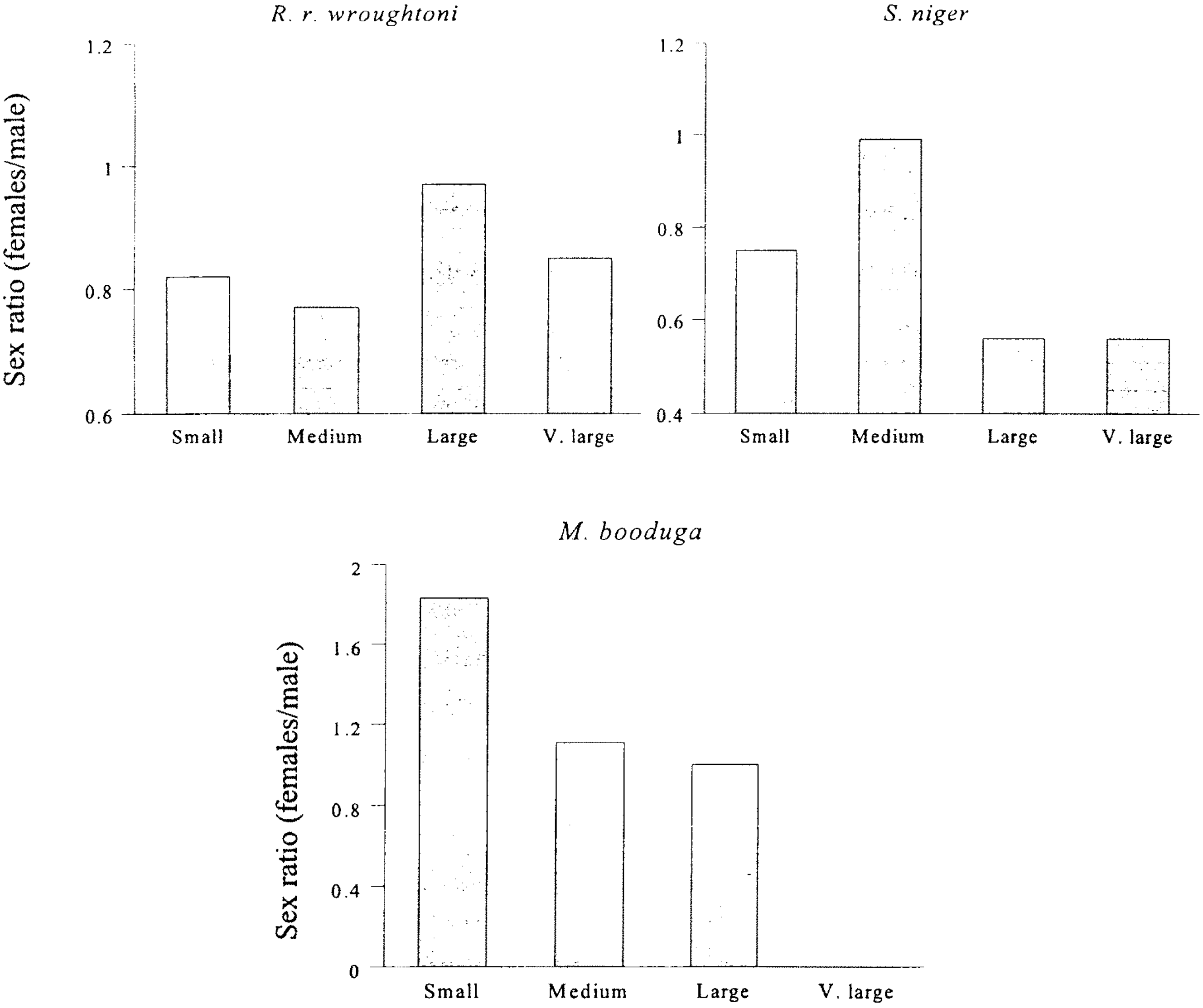


Figure 4.6 The adult sex ratio of (females/male) in *R. r. wroughtoni*, *S. niger* and *M. booduga* among fragments of four size classes.

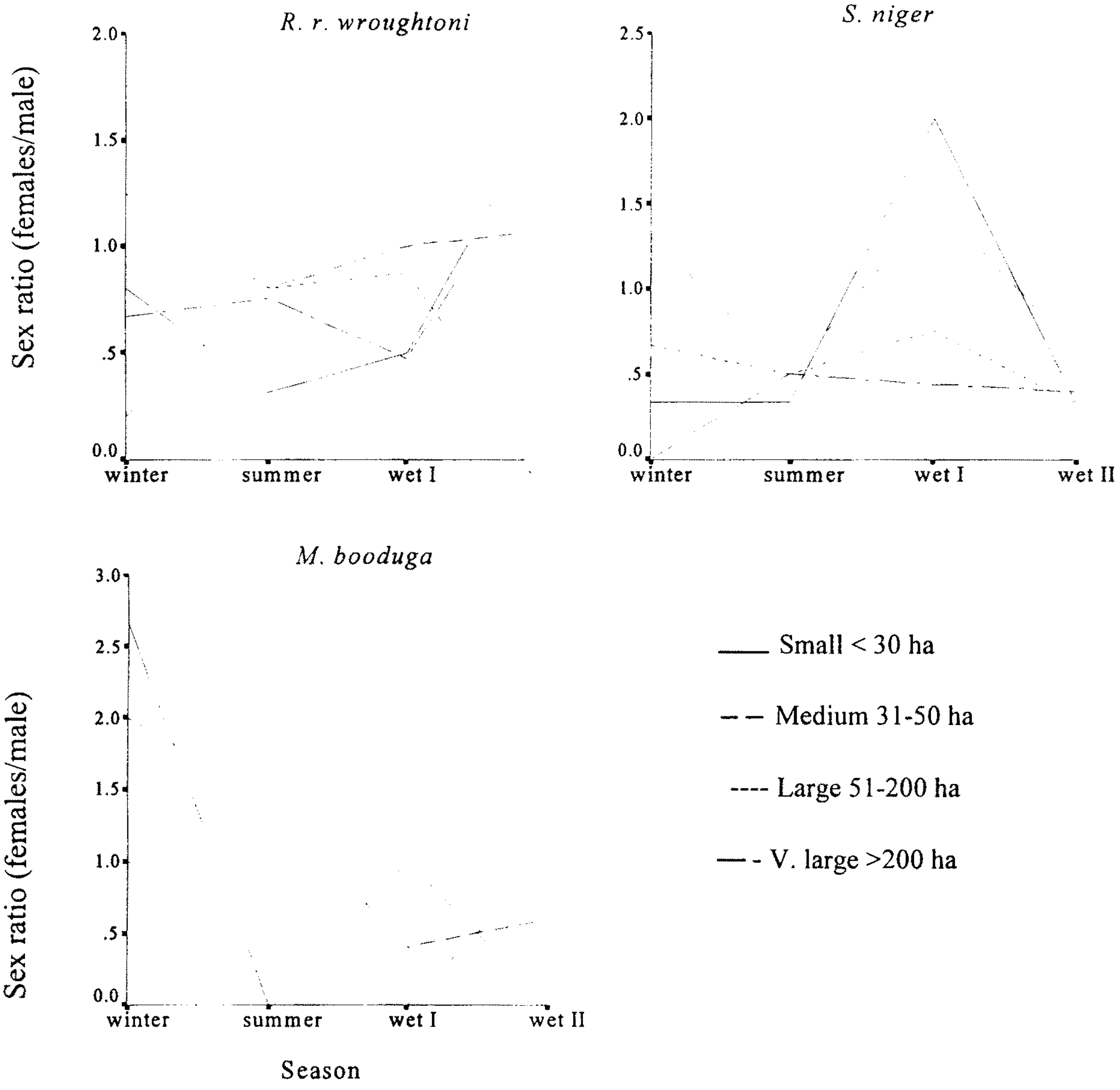


Figure 4.7. Seasonal synchrony in sex ratio (adult females/male) amomg fragments of four size classes, in *R. r. wroughtoni, S. niger* and *M. booduga*.

There was no seasonal synchrony among fragments in age ratio, when fragments where grouped into four area classes (Figure 4.8). The synchrony was highest in *S.niger* (W=0.332; χ^2=5.13; P=0.162), with the lowest ratio in winter and the highest ratio in summer. For *R. r. wroughtoni* (W=.009; χ^2=0.130; P=0.988) and *M. booduga* (W=0.076; χ^2=1.13; P=0.769), there was no synchrony. There was thus no indication of seasonal synchrony among fragments in subadult to adult ratio and adult sex ratio, in *R. r. wroughtoni, M. booduga*, while *S. niger* showed some degree of synchrony.

4.3.5 Body Weight

4.3.5.1 Synchrony in Body Weight

Among the species trapped, *R. blanfordi* was the largest species with an average body weight of 137.80 g (s.e.=9.60, n=5), and head and body length of 15.80 cm (s.e.=0.20, n=5) (Table 4.1). *R. r. wroughtoni* was the second largest, with a body weight of 102.58 g (s.e.=1.59, n=259) and heads and body length of 14.38 cm (s.e.=0.12, n=259). This was followed by *G. ellioti* (body weight = 65.25 g, s.e.=11.61, n=4), *P. lasiurus* (59.17 g, s.e.=4.03, n=6), *S. niger* (25.41 g, s.e.=0.47, n=155), *M. musculus* (24.73 g, s.e.=1.69, n=11), *V. oleracea* (21.00 g, s.e.=2.05, n=5) and *M. booduga* (13.23 g, s.e.=0.43, n=101). Among the species with a large sample size, the males were heavier than the females by about 10% in the case of *R. r. wroughtoni*, by about 20% in the case of *M. musculus* and about 4% in *S. niger*, where as the female was heavier than the males by about 12% in *M. booduga* (Table 4.1). The body weight was significantly positively correlated with head and body length in the three species (*R. r. wroughtoni*, *S. niger* and *M. booduga*) on

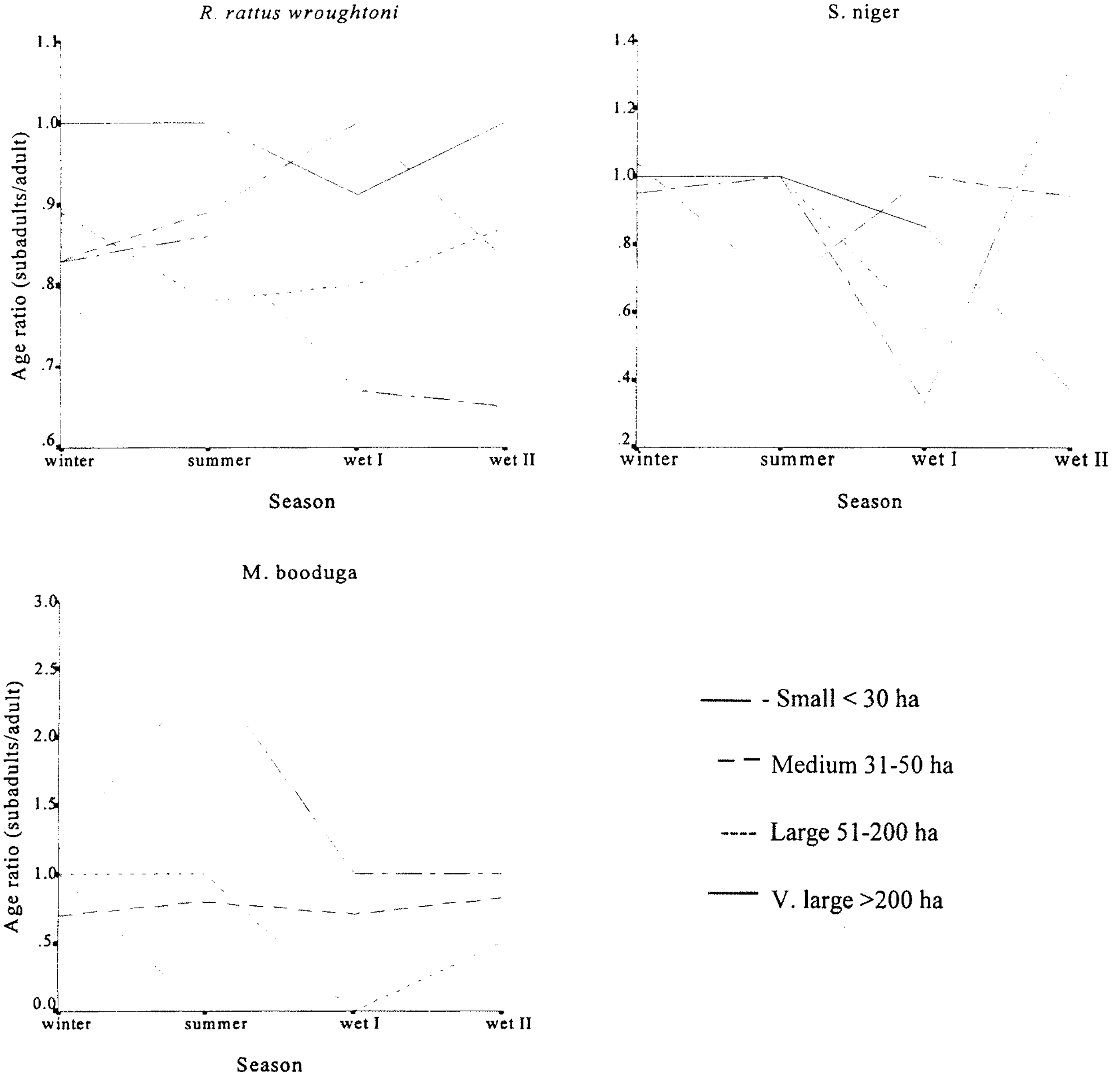


Figure 4.8. Seasonal synchrony in age ratio (subadults/adult) among fragments of four size classes in *R. r. wroughtoni, S. niger* and *M. booduga* in the study area.

Table 4.1 Differences in mean body weight and head and body length of adult male and female of small mammals in the study area.

Species	Mean body weight (se) (g)		Mean head & body length (se) (cm)	
	Male	Female	Male	Female
R.r. wroughtoni	107.61(*2.66*)	98.50(*1.86*)	14.64(*2.66*)	14.16(*0.14*)
n=	116	143	116	143
S.niger	26.07(*0.83*)	25.02(*0.60*)	9.81(*0.20*)	9.64(*0.15*)
n=	57	98	57	98
M.booduga	12.43(*0.50*)	13.96(*0.68*)	6.69(*0.11*)	7.10(*0.11*)
n=	49	52	49	52
M.musculus	27.20(*2.65*)	22.67(*1.98*)	9.30(*0.44*)	8.98(*0.31*)
n=	5	6	5	6
R.blanfordi	122.00	141.75(*11.30*)	15.00	16.00(*.00*)
n=	1	4	1	4
V.oleracea	16.00	22.25(*2.10*)	8.20	6.12(*0.73*)
n=	1	4	1	4
*G.elliot*i	85.00	58.67(*13.53*)	12.50	11.93(*1.03*)
n=	1	3	1	3
P.lasiurus	61.00(*6.11*)	57.33(*6.36*)	10.27	10.67(*0.17*)
n=	3	3	3	3

which there were sufficient data, the R^2 ranging from 0.195 to 0.446 (Figure 4.9). For all fragments together, the mean body weight of *R. r. wroughtoni* was greatest in wet I (111.96 g) and lowest in wet II (95.23 g). For *S. niger* it was greatest in wet I (31.23 g) and lowest in winter (22.47 g), so also for in *M. booduga* with the highest body weight in wet I (17.50 g) and lowest in winter (11.60 g). There was no synchrony in seasonal variation of body weight either in *R. r. wroughtoni* (Kendall's W=0.20; χ^2=0.286; *df*=2; P=0.867), or *S. niger* (Kendall's W=0.25; χ^2=1.00; *df*=1; P=0.317). When the seasons

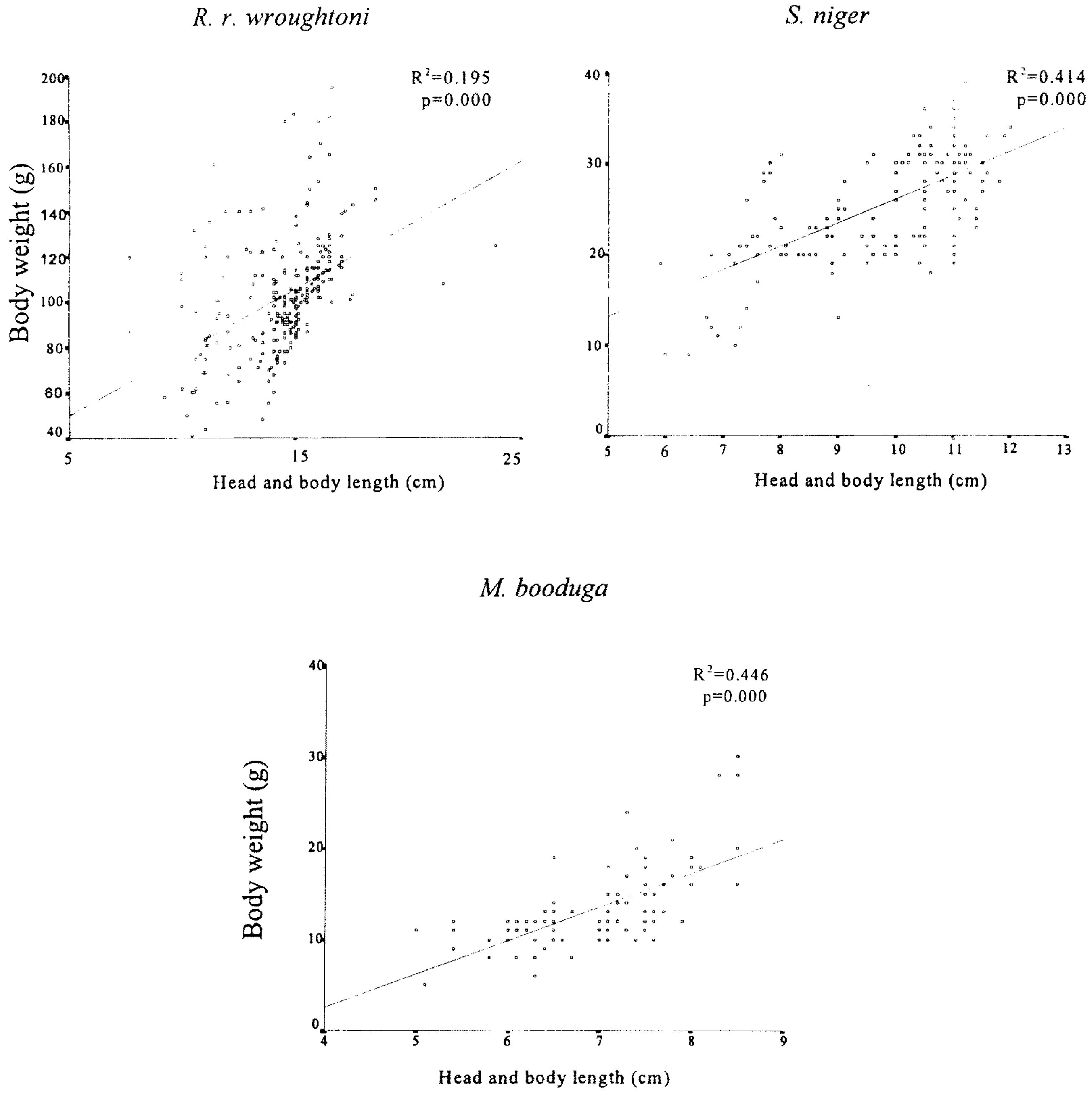


Figure 4.9. Body weight in relation to head and body length in *R. r. wroughtoni*, *S. niger* and *M. booduga*.

were pooled into dry (winter and summer) and wet (wet I and wet II) seasons, the body weight of *R. r. wroughtoni* was greater in the small fragments in the wet seasons, but in the dry season as fragments become greater (Figure 4.10). There was thus some synchrony in the body weight among the small fragments that was different from the synchrony in the large fragments. More over, the difference in body weight between the dry and wet seasons was far greater in the smaller fragments than in the larger fragments.

In *S. niger*, the body weight was greater in the wet season in five out of the six fragments in which the species was caught in both dry and wet seasons (Figure 4.10). In this species, therefore, there was greater synchrony in body weight across all sizes of fragments, unlike in *R. r. wroughtoni* in which the small fragments and the large fragments had different synchronies. *M. booduga* was caught only from three fragments in both dry season and wet seasons, with body weight being greater in the wet seasons in two fragments and in the dry season in one fragment (Figure 4.10). For the other species, captures were not sufficient to make similar comparisons. There was no correlation between the abundance and body weight of any three species, the R^2 varying from 0.000 to 0.086.

4.3.5.2 Correlates of Body weight

The body weight of *R. r. wroughtoni* varied considerably among fragments, from a lowest mean of 88.54 g to a highest mean of 111.95 g. For *S.niger,* the lowest mean body weight was in a small fragment (21.75 g), while the highest was in the large fragment (29.92 g). In *M. booduga*, the lowest body weight was in the large fragment (10.80 g), while the highest was from a small fragment (18.00 g) (Table 4.2).

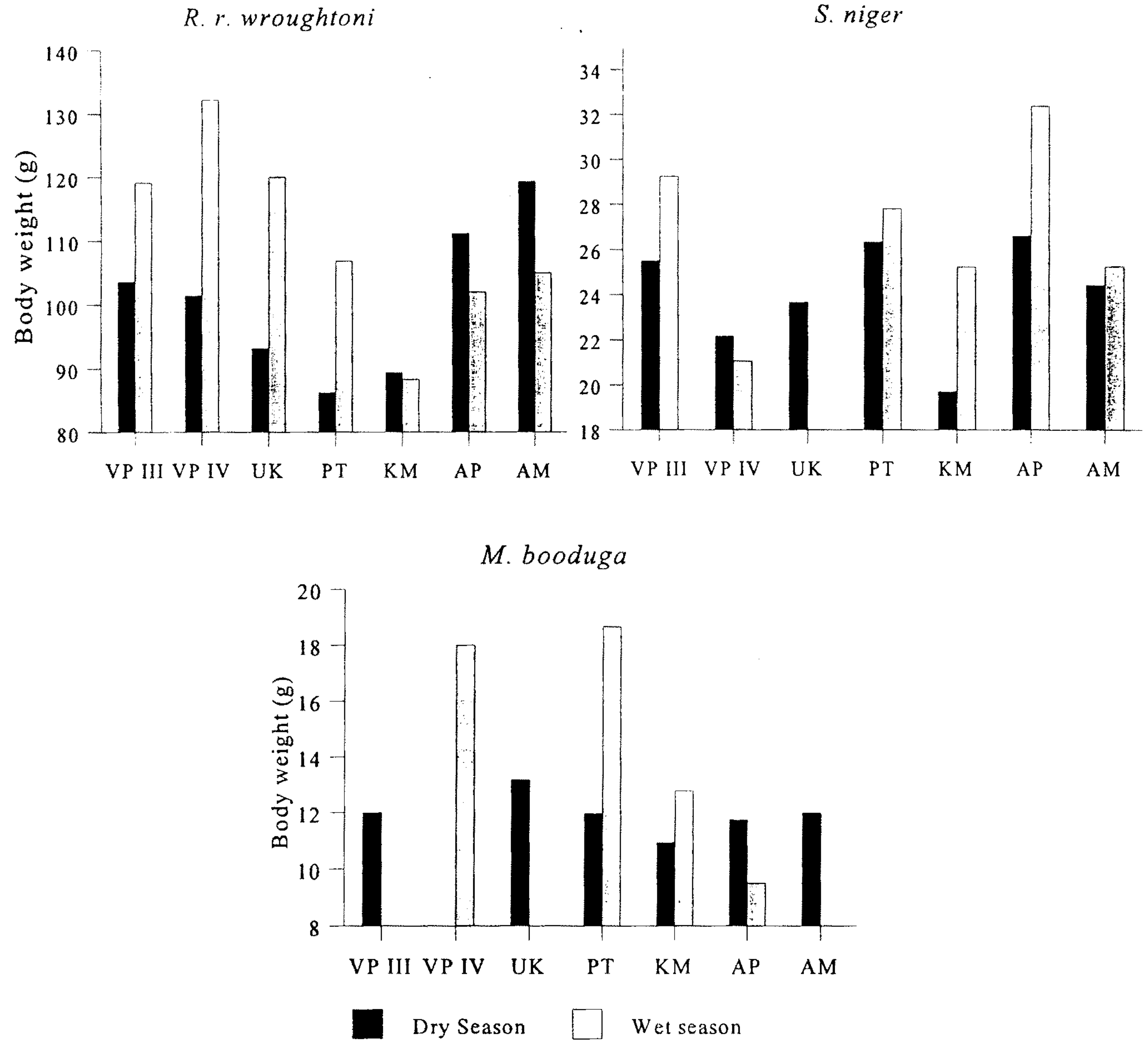


Figure 4.10. Body weight of *R. r. wroughtoni, S. niger* and *M. booduga* in forest fragments in the dry (winter and summer) and wet (wet I and wet II) seasons (abbreviations: VP III= Varattuparai-III, VP IV=Varattuparai-IV, UK=Urulikkal, PT=Puthuthottum, KM=Korangumudi, AP=Andiparai shola, AM=Akkamalai).

Table 4.2 The mean body weight of rodents and shrews trapped in forest fragments during the study period.

Name of Fragment	mean body weight (g)							
	RRAT	SNIG	MBOD	MMUS	RBLAN	VOLEA	GELIO	PLASI
Varattuparai-III	106.42	27.67	12.00		122.00			
Varattuparai-IV	101.34	21.75	18.00	30.00	139.00			
Urulikkal	88.54	23.00	12.77	20.00				
Puthuthottum	97.20	26.57	14.76	25.00			65.25	
Korangumudi	91.06	22.82	11.69	20.00				
Andiparai	106.57	29.92	10.80	27.50				
Akkamalai	111.95	25.77	12.00	23.50	150.00	21.00		59.17

Abbreviation: RRAT=*Rattus rattus wroughtoni*, SNIG=*Sorex niger*, MBOD=*Mus booduga*, MMUS=*M.musculus*, RBLAN=*R.blanfordi*, VOLEA=*Vandeleuria oleracea*, GELIO=*Golunda ellioti*, PLASI=*Platacanthomys lasiurus*.

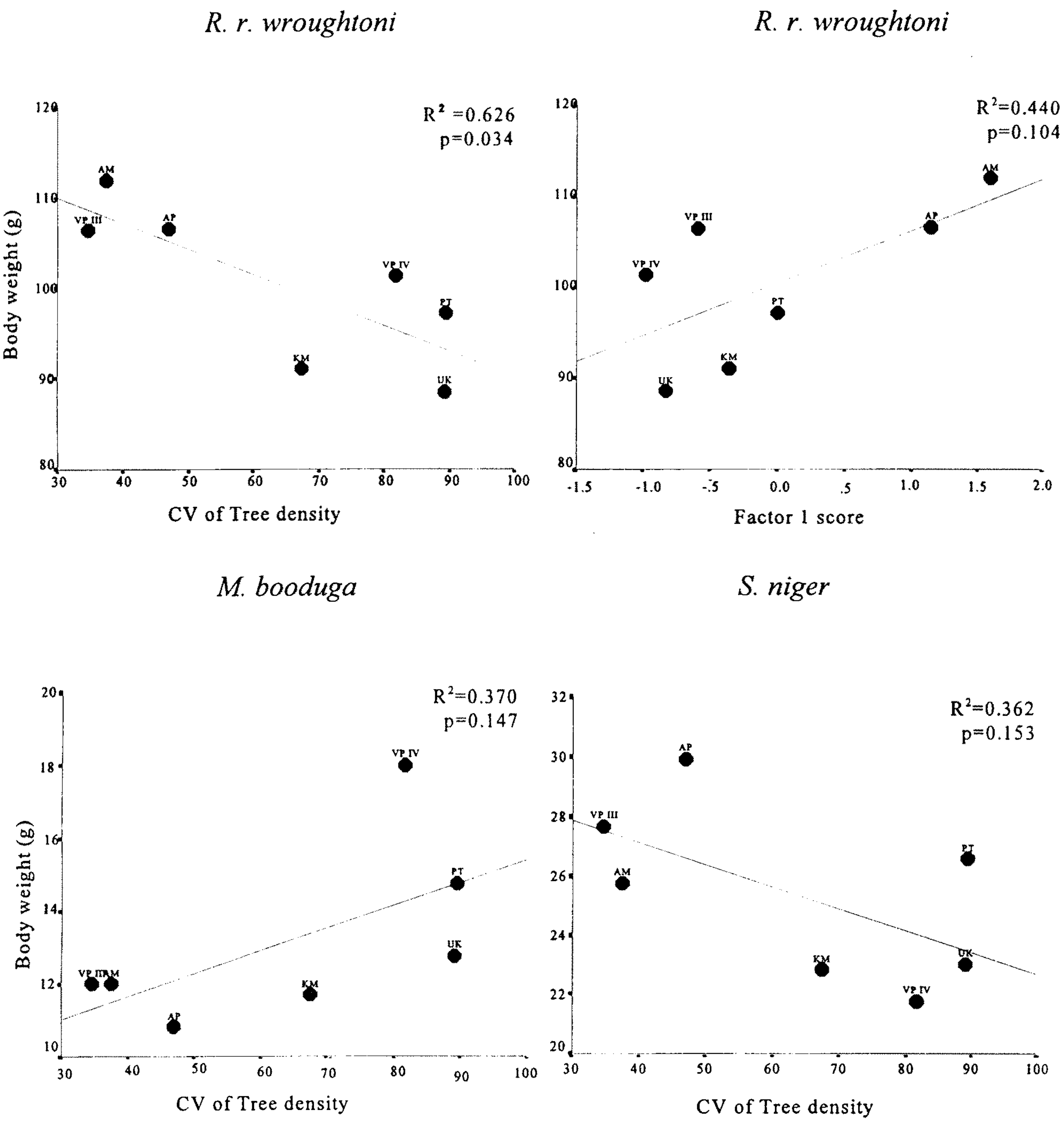
Mean body weight was compared with the two Factors, and habitat parameters and their CVs (Table 4.3). The body weight of *R. r. wroughtoni* increased linearly with Factor 1 scores, which represented the status of habitat, the relation being nearly significant (R^2=0.440; *df*=5; P=0.104) (Figure 4.11). The body weight also tended to decrease with increasing habitat heterogeneity, as measured by Factor 2 (R^2=0.309; *df*=5; P=0.195), even though this was not significant. Among the individual parameters, the body weight increased with increasing tree density (R^2=0.596; *df*=5; P=0.042), and decreased with increasing CV of tree density (R^2=0.626; *df*=5; P=0.034) and CV of canopy height

Table 4.3 R^2 values for linear fit of body weight with the two Factor scores, and several habitat parameters.

	R.r.wroughtoni		*S.niger*		*M.booduga*	
Parameters	R^2	P	R^2	P	R^2	P
Log area	0.207	0.305	0.253	0.250	0.142	0.404
Tree density	0.596	0.042*	0.159	0.376	0.244	0.260
Canopy height	0.107	0.474	0.345	0.166	0.224	0.283
Tree density (CV)	0.626	0.034*	0.362	0.153	0.370	0.147
Basal area (CV)	0.107	0.474	0.121	0.445	0.019	0.770
Canopy height (CV)	0.511	0.071	0.098	0.494	0.121	0.444
Undergrowth (CV)	0.279	0.223	0.001	0.949	0.047	0.640
Factor 1	0.440	0.104	0.190	0.329	0.274	0.227
Factor 2	0.309	0.195	0.010	0.835	0.002	0.932

*$P<0.05$

($R^2=0.511$; *df*=5; P=0.071). The other habitat variables and CVs were not significantly correlated, ranging from 0.107 to 0.279 with body weight ratios (Table 4.3). The body weight of *S. niger* did not show any significant correlation with the two Factors (Table 4.3) and the habitat parameters, the best fit being a negative correlation with the CV of tree density ($R^2=0.362$; *df*=5; P=0.153) (Figure 4.11). The body weight of *M. booduga*, which varied from 10.8 g to 18 g, also did not show any significant correlation with the two Factors. The best correlation (positive) was with the CV of tree density ($R^2=0.370$; *df*=5; P=0.147) (Figure 4.11). Thus, in *R. r. wroughtoni* body weight seemed to increase with tree density and decrease with increasing CV of tree density, while that of *M. booduga* increased with CV in tree density. Body weight in *S. niger* also showed to decrease with CV of tree density.



R. r. wroughtoni

R. r. wroughtoni

M. booduga

S. niger

Figure 4.11. The body weight of *R. r. wroughtoni, S. niger* and *M. booduga* in relation to spatial heterogeneity (CVs) of tree density and Factor 1 score.

4.4 DISCUSSION

4.4.1 Seasonality in Breeding

The seasonal and annual population fluctuations in tropical rodents are not very striking, compared to that in temperate regions, for example in microtine rodents. Nonetheless, seasonal cycles in the production of foliage, fruits and seeds, and insects which are related to temperature and rainfall do occur in tropical rain forests (see Leigh *et al.* 1982). This is reflected in the breeding of rodents and shrews. Thus, rodents breed in the late summer or the wet season that follows in Africa (Iyawe 1989; Wirminghaus and Perrin 1993), neo tropics (Fleming 1973; Smith and Vrieze 1979; Murúa *et al.* 1987; Mares and Ernest 1995) and Asia (Yu 1993; Shanker 1998). The species differences in breeding season might be related to their diet, since fruiting often peaks in late summer and foliage production in wet season (Leigh *et al.* 1982; Murúa *et al.* 1987; Yu 1993). Similarly, ground shrews which are primarily insectivorous (Hutterer 1985) breed in late summer when insect abundance peak (Yu 1993). In India, *R.r.wroughtoni* has been reported to breed mostly in the early wet season (southwest monsoon) in the upper Nilgiris (Shanker 1998), and during both southwest and northeast monsoons in the lower altitudes in northern Kerala (Bhat *et al.* 1987). In the Deccan plateau, *M. booduga* has been reported to breed in October-November (Chandrahas 1974), the peak rainy season in the area. In the Anaimalai Hills, when data from fragments were pooled *R.r.wroughtoni, S. niger* and *M. booduga* seemed to breed in late summer or early wet season, as indicated by the greater subadult to adult ratio in the wet season I. Late summer is the peak fruiting time in the Anaimalai Hills (Kumar 1987), and early wet season is the foliage flush season.

However, young ones were present in all the seasons in all species, as has been the case in most tropical areas.

Population abundance, when data was pooled for fragments, was highest in wet II season, the season following breeding season, in *R. r. wroughtoni* and *S. niger*, while in *M. booduga* it was in summer. A reduction in the population is possible during or just prior to the breeding season due to dispersal or emigration especially by males in search of females (Jurgiel 1992). Thus the lowest capture rate was in summer in *R. r. wroughtoni* and *S. niger* and in wet I, in *M. booduga*, either during or just prior to the breeding season. Emigration of males is indicated in *S. niger* by the high adult sex ratio in summer. In *R. r. wroughtoni* and *M. booduga*, however, the sex ratio was high in all seasons except in one season, wet II in *R. r. wroughtoni* and winter in *M. booduga.*

4.4.2 Seasonal Synchrony

When fragments were considered separately, there was no seasonal synchrony among them in any demographic parameters in *R. r. wroughtoni* and *M. booduga*, The shrew had greater synchrony than the rodents. Rodents have been known to show demographic synchrony at a regional scale, because of similar environmental fluctuations that they experience, dispersal to a large extent dampening local variation in resource abundance. In temperate regions such demographic synchrony may occur over a distance of 30-40 km, beyond which synchrony declines (Steen *et al.* 1996). Disruption of such regional synchrony is a consequence of habitat fragmentation (Adler 1994). Due to the absence of or infrequent dispersal from forest fragments, the isolated populations respond to variation in resource abundance within the fragment. Since tropical forests show a high patchiness in the distribution of resources, synchrony in resource abundance could be expected to be low among fragments. Another factor that might promote asynchrony is demographic stochasticity that small populations in fragments are susceptible to (Gilpin and Soulé 1986). Adler (1994) found that fragmentation promotes asynchrony in a frugivorous rodent in Panama, which was due to differences in food availability. Shanker (1998) concluded that demographic stochasticity might be an important factor promoting demographic asynchrony among fragments.

In the Anaimalai Hills, *R. r. wroughtoni* and *M. booduga* did not show any synchrony while *S. niger* showed some synchrony. This might be related to their diet and dispersal capabilities. There is no information on the diet of these species. Plant species composition varied considerably among fragments, species richness generally decreasing with area (Umapathy 1998). There might be considerable differences in the food availability among fragments for the rodents thus promoting asynchrony. There is no information on the abundance of insects in forest fragments and its impact on the insectivorous shrew, *S. niger*. It has been, however, reported that ground insects increase in rain forest fragments (Malcolm 1991). Dispersal among fragments is an important factor that could affect demographic synchrony among fragments. In the upper Nilgiris, *R. r. wroughtoni* dispersed within a 600 ha fragment, the sites within which showed demographic synchrony. In contrast, there was no dispersal among fragments (Shanker 1998). Synchrony also declined with increasing distance between fragments.

A fragmented landscape with a mosaic of vegetation types can be a filter, allowing some species to disperse over long distance (Kozakiewicz and Jurasińska 1989; Jurgiel 1992; Kozakiewicz 1993; Diffendorfer *et al.* 1995) . However, the proportion of animals moving out might decrease as fragmentation increases (Diffendorfer *et al.* 1995). In the Anaimalai Hills, *R. r. wroughtoni* has been captured in the matrix around the fragments (tea and coffee estates) at a rate of 0.71/100 trap nights (Anon. 1997). This rate is considerably lower than in the fragments (2.62/100 trap nights). This probably indicates their low use of the matrix and low dispersal. Being an arboreal species to a large extent, canopy contiguity may be important for the dispersal of this species (Chandrasekar-Rao and Sunquist 1996). In contrast, *Suncus* spp. was trapped at a higher rate (0.38/ trapped) relative to its abundance in fragments (1.52/100 trap nights). It is likely, therefore, that *S. niger* might disperse among fragments, and are able to maintain a greater synchrony as was seen. *Mus* species was captured only at low rate (0.10/100 trap nights) from the matrix around (Anon. 1997), even though being a generalist and introduced species, a greater capture rate was expected. It is likely that the commensals, *M. booduga* and *M. musculus*, by and large disperse to fragments from human settlements (with a source-sink relationship) and hence might show little synchrony among fragments. In the absence of any estimate of the densities of these species, in fragments the role of demographic stochasticity cannot be assessed.

There was considerable variation in the vegetation in and around the fragments depending the proximity of human settlement. This has been reported to be a major factor governing the occurrence of arboreal mammals such as lion-tailed macaque, in the same area (Umapathy 1998). When human settlements were nearby, the orchards raised near the fragments often contained species such as jack fruit, mango, and guava. Fragments without human settlements nearby, were often surrounded by estates. This difference could not only affect the abundance of fruits and seeds to the rodents, but also their seasonality, in the fragment. This may be another factor that caused the asynchrony among fragments.

Thus, the asynchrony among fragments, in the demography of *R. r. wroughtoni* and *M.booduga* may be related to both variation in food abundance and lack of dispersal among fragments. The shrew, *S. niger*, shows greater synchrony because the abundance of insects may show lower variation among fragments. Moreover, *S. niger* might also show greater dispersal capabilities.

The only parameter which showed some synchrony among fragments was body weight. In *R. r. wroughtoni*, body size was greater in the small fragments in the wet months while in the large fragments it was greater in the dry season (summer and winter). In *S. niger* and *M. booduga*, the body weight was greater in the wet season. This might reflect the greater food availability in the wet season. Seasonal variation is common in small mammals, which often related to body weight and fat content and reflect food availability and breeding time (Wirminghaus and Perrin 1993). The factors that affect body weight seem to be the same as those that regulate abundance. In *R. r. wroughtoni*, both body weight and abundance increased with increasing Factor 1 score, and decreased with Factor 2 score, while the reverse was true for *M. booduga*. *S. niger* did not respond to these either in abundance or in body weight. The body weight of small mammals on islands have been reported to increase due to lack of predation and competition (Foster

1964; Angerbjorn 1985). However, forest fragments on mainlands are not comparable to islands, due to the presence of predators as well as invading competitors. In fact, predator density might be higher in disturbed forests due to invasion by new predator species to which the native small mammals might be ill-adapted (Batzli 1985; Johns and Skorupa 1987; Soule *et al.* 1992; Batzli and Lesieutre 1995; Smith and Quin 1996). A reduction in body weight in native species might be therefore expected. A decrease in body weight might also be expected due to a decline in food abundance or preferred microhabitats in fragments, or increasing competition from invading species such as M. booduga and M. musculus. The relative importance of these factors cannot be evaluated now in the absence of information on predators, the nature and extent of competitive interactions between species, and resource requirements of the species. The segregation of species in microhabitat selection is discussed in Chapter 5.

4.5. SUMMARY

1. In this Chapter I examined the seasonal variations in population abundance, age ratio, adult sex ratio and body weight of the three most frequently captured species in this study, *R.r.wroughtoni, S.niger,* and *M.booduga.* The synchrony among forest fragments in the seasonal variation of the above parameters was also examined.

2. There was considerable seasonal variation in the abundance of the three species in many fragments. However, when the data from the fragments were pooled the seasonal differences were not very evident in *R.r.wroughtoni* and *M.booduga.* This was due to the absence of synchrony among the forest fragments in population

abundance. There was greater synchrony in *S.niger*, hence seasonal variation in the pooled data was marginally significant.

3. The subadult to adult ratio in the data pooled over all the fragments indicated that the breeding period for all three species was in the late summer and southwest monsoon. This might be correlated with peak fruit and foliage production in the habitat. There was no difference between forest fragments of four size classes in the age ratio in the case of *R.r.wroughtoni*. The differences in the other two species did not show any consistent pattern with fragment area. There was some synchrony among fragments in the seasonal variation of age ratio only in the case of *S.niger*.

4. The adult sex ratio did not show any consistent pattern with fragment area in *R.r.wroughtoni* and *S.niger*, while in *M.booduga* it seemed to decrease with fragment area. Synchrony was evident only in the case of *S.niger*.

5. When data from fragments were pooled, the body weights of all species were greatest during the southwest monsoon. The body weight of *R.r.wroughtoni* increased with increasing tree density, decreasing CV of tree density, and increasing Factor 1 score. *M.booduga* showed an opposite relation with the same parameters while *S.niger* did not show significant relationship. The body weight of all species was greater in the wet seasons in most fragments.

6. The lack of demographic synchrony among forest fragments in the case of *R.r.worughtoni* and *M.booduga* might be related to the differences among fragments

in plant species composition and lack of dispersal between fragments. Invertebrates, the major food of *S.niger*, may show less variation among fragments. Moreover, this might be also able to disperse between fragments. These two factors might promote greater demographic synchrony among forest fragments in the case of *S.niger*.

5. MICROHABITAT SELECTION

5.1 INTRODUCTION

Habitat selection and use in small mammals have been studied at two levels the macrohabitat and microhabitat. The former refers to features of the habitat that vary across the home range of an individual, while the latter refer to features that vary within the home range (Adler 1988). The influence of macrohabitat features on the occurrence and abundance of small mammals was discussed in Chapter 3. The microhabitat is the template over which species interaction and competitive hierarchies are expressed (Bowers and Flanagan 1988). The availability of microhabitat may therefore determine species abundance at a local scale (Price 1978).

Microhabitat occupancy and segregation often reveal how the ecological communities are structured in an ecosystem (Dueser and Porter 1986; Bowers and Flanagan 1988; Gore 1988; Churchfield and Brown 1997). Microhabitat selection among small mammals depends upon the ecological requirements of the species and resource partitioning among competing species (Porter and Dueser 1982; Putman and Wrattan 1984). Microhabitat selection by small mammals has been extensively studied in the wild (Dueser and Shugart 1978, 1979; Adler 1988; Gore 1988; Prakash *et al.* 1995; Churchfield and Brown 1997; Shanker 1998 and others).

The species segregation at microhabitat level might reduce competition among species, but whether the segregation is due to competition has been a matter of considerable debate. Microhabitat selection may be related to the biology of the animal, such as body weight, diet or denning (Canova and Fasola 1991). Adler (1988) argued that microhabitat structure influences population density more than other demographic variables. Dueser and Shugart (1978) studied four species of forest-floor small mammals, of which three species showed a high degree of microhabitat selection, deciduous canopy, density of trees, bushy shrub cover, and shrub ever greenness, being important microhabitat variables. Ecologically similar species that are also similar body-sized often segregate their microhabitats by competition or specialised food preference. For instance, *Peromyscus* and *Ochrotomys* are similar in appearance, body size and general ecology, however, these species are least similar qualitatively in microhabitat preferences (Dueser and Shugart 1978). Extreme habitat specialisation of small mammals in islands due to intense interspecies competition and can lead to this competitive exclusion by invaders or habitat generalist (Laurance 1994). In the upper Nilgiris in the Western Ghats, *Rattus rattus, Suncus montanus* and *Mus famulus* showed no difference in microhabitat selection, while *M. famulus* showed an apparent preference for ground cover, canopy cover and tree density (Shanker 1998) .

Factors other than competition and microhabitat selection (such as predation and disease) might also be important in structuring small mammal communities. The relative importance of these factors may vary spatially as well as temporally (Batzli 1985; Brown 1987; Bowers and Flanagan 1988; Gore 1988; Kotler and Brown 1988; Batzli and Lesieutre 1995).

Several removal experiments have demonstrated the important role that competition play in microhabitat selection (Price 1978; Schoener 1983; Walter *et al.* 1984; Bowers and Flanagan 1988; Scott and Dueser 1992; Valone and Brown 1996), and that local competitive exclusion might decide which species is represented in each guild (Rosenzweig and Winakur 1969). Several removal experiments, however, have failed to demonstrate the role of competition in structuring small mammal community and have concluded that microhabitat selection and other factor may play roles (Murúa *et al.* 1987; Batzli and Lesieutre 1995; Shanker 1998).

After a review of desert communities, consisting of 29 species in 202 sites in USA, Brown (1987) concluded that the distribution of each species reflected its unique needs and was independent of the occurrence of other species. He also concluded that in a small region it is structural diversity of the habitat that promotes coexistence of species that are regionally present. Rosenzweig and Winakur (1969) were able to relate habitat structure with a particular functional group or rodents, and experimentally test it. Dueser and Porter (1986) also come to the conclusion that in insular islands local species pool, habitat structure and colonisation abilities of species, determined communities, and not competition. Providing specific microhabitats can alter species composition. Prakash (in press) found that stone walls in agricultural fields extended the distribution of Cutch rock-rat (*Cremnomys cutchicus*) a forest species, into agricultural fields. Amarasekare (1994) and Soulé *et al.* 1992 found no evidence of competition between the introduced species (*Rattus rattus* and *Mus musculus*) and native small mammals.

In this chapter I examine the response of four species to variations in microhabitats and discuss its implication in the context of habitat fragmentation.

The objectives are:

a) to examine the microhabitat preferences of the four most abundant species (*R. Rattus wroughtoni, S. niger, M. booduga* and *M. musculus*).

b) to discuss the relative importance of microhabitat preferences and competition in the changes in the small mammal community due to habitat fragmentation.

5.2 METHODS AND ANALYSIS

Microhabitat preferences were examined for the four most frequently captured species (*R. r. wroughtoni, S. niger, M. booduga* and *M. musculus*) on which there was sufficient data. Data from all fragments were pooled for the analysis. Seventeen microhabitat parameters were recorded from 1 x 1 m (1m^2) and 5 x 5 m (5m^2) quadrats centred around the trap station (see Table 2.2). These were recorded at the end of each trapping session for all sprung trap stations, and at every alternate unsprung stations. Data was recorded from 1,255 trap stations in total, of which 704 were from unsprung stations and 551 were from capture sites.

Only the data from 1m^2 quadrats was used for analysis here since this differentiated the species better than the data from 5m^2 quadrats. The expected percentage of captures in each category of a microhabitat variable was compared with the observed percentage of captures, in order to examine microhabitat selection. The expected percentage was calculated from sampling of alternate trap stations. The assumption was that if there was no selection the expected percentages of captures in different categories of a microhabitat variables would be equal to the percentage of trap stations available under these categories. If the observed percentages of captures in a category out of total captures of a species was greater than the expected, it indicated selection, and if lower it indicated avoidance.

Even though I measured 17 microhabitat parameters, several parameters did not show much variation (*e.g.,* sand cover, vegetative and reproductive phases of trees). Therefore, only 11 microhabitat variables were analysed. Of these, five variables (trees, lianas, rattans, fallen logs and buttresses) had only two categories, present or absent. Six variables had three or four categories each, which represented either numbers (*e.g.,* number of shrubs), or grouped classes such as percentage cover and GBH of trees. For each variable only one χ^2 test was done including all species and using the externally generated expected frequencies, with *df* = (number of categories - 1) x (number of species - 1). I also attempted a logistic regression (Hosmer and Lemeshow 1989), with the same microhabitat variables in order to identify the variables that significantly influenced the probability of capture of a species at a trap station.

5.3 RESULTS

5.3.1 Bivariate Analysis

Trees

Five variables (trees, buttressed trees, lianas, fallen logs and rattans) were recorded only as being absent or present from quadrats around the trap stations. A tree was present only in 23.6% of the quadrats. If captures were random with reference to the occurrence of trees near trap station, 23.6% of the captures of each species should be from quadrats with trees. *R. r. wroughtoni, S. niger* and *M.booduga* were all captured more frequently from quadrats with trees (Figure 5.1). In *R. wroughtoni*, 39.2% of the captures from quadrats with trees, compared to an expected percentage of only 23.6%. The percentage of captures were also greater than 23.6% in the case of *S. niger* (36.6%) and *M. booduga* (30.7%). In contrast only 18.2% of the captures of *M. musculus* was from quadrats with trees, thus showing its avoidance of such quadrats. The above difference between expected and observed percentages of captures from quadrats with trees was statistically significant (χ^2=42.28; *df*=3; P<0.001).

Lianas

Only 13.1% of the quadrats around trap stations had lianas. *R. r. wroughtoni* (30.6%), *S. niger* (224.2%), and *M. booduga* (26.7%) had far greater percentage of captures from

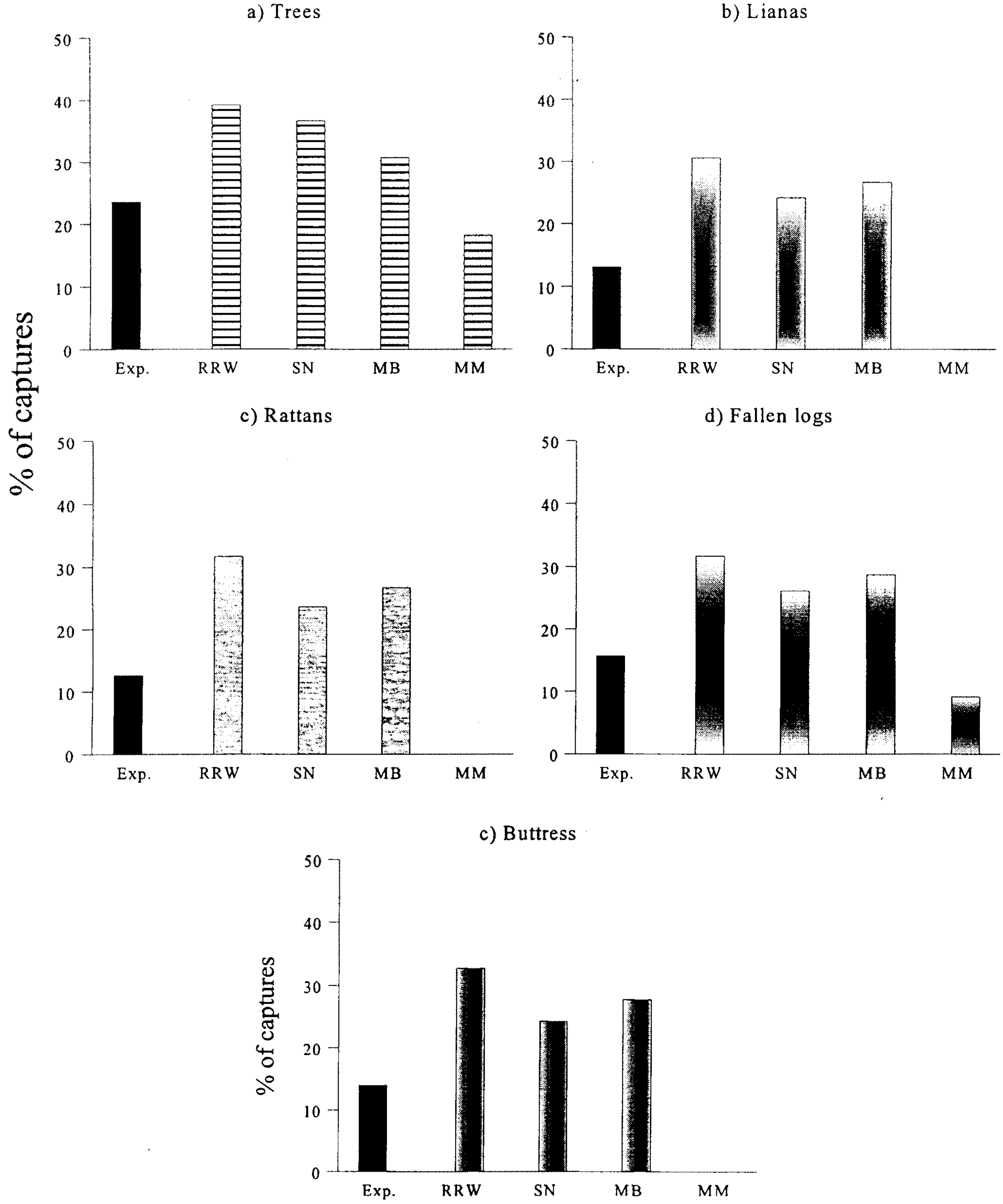


Figure 5.1 A comparison of the expected and observed percentages of captures when at least one tree, liana, rattans, fallen logs and buttress trees was present in 1m^2 quadrat around the trap station (abbreviations:Exp.=Expected percentage of captures out of total captures of a species, RRW=*R. r. wroughtoni*, *SN=S. Niger*, MB=*M.booduga*, MM=*M.musculus*).

quadrats with lianas (Figure 5.1). In contrast, none of the captures of *M. musculus* were from quadrats with lianas. The difference between expected and observed percentages was significant ($\chi^2=95.44$; *df*=3; P<0.001).

Rattans, Fallen logs & Buttressed trees

Similarly, significantly more captures of *R. r. wroughtoni, S. niger* and *M. booduga* and fewer captures of *M. musculus* were obtained from quadrats with rattans ($\chi^2=112.13$; *df*=3; P<0.001), fallen logs ($\chi^2=66.71$; *df*=3; P<0.001), and buttressed trees ($\chi^2=96.19$; *df*=3; P<0.001) (Figure 5.1).

Thus, the captures of *R. r. wroughtoni, S. niger* and *M. booduga* were more associated with quadrats in which trees, lianas, buttress trees, fallen logs and rattan were present. In contrast, *M. musculus* seemed to avoid such sites. The difference between expected and observed percentages of captures was greatest in *R. r. wroughtoni*, for all variables. The difference between expected and observed percentages were similar in *S. niger* and *M. booduga*, except in the case of trees for which *S. niger* seemed to show a greater difference (Figure 5.1). Among the five variables, the difference between expected and observed was greatest in the case of buttressed trees.

For six other microhabitat variables, an assessment of abundance was made in the 1m^2 quadrats. The shrubs were counted, while herb, canopy, litter and rock covers were visually estimated as percentage cover, in categories (0, 20, 40, 60, 80 and 100%). For analysis these were grouped into three categories, <40%, 40-80% and >80%. The GBH

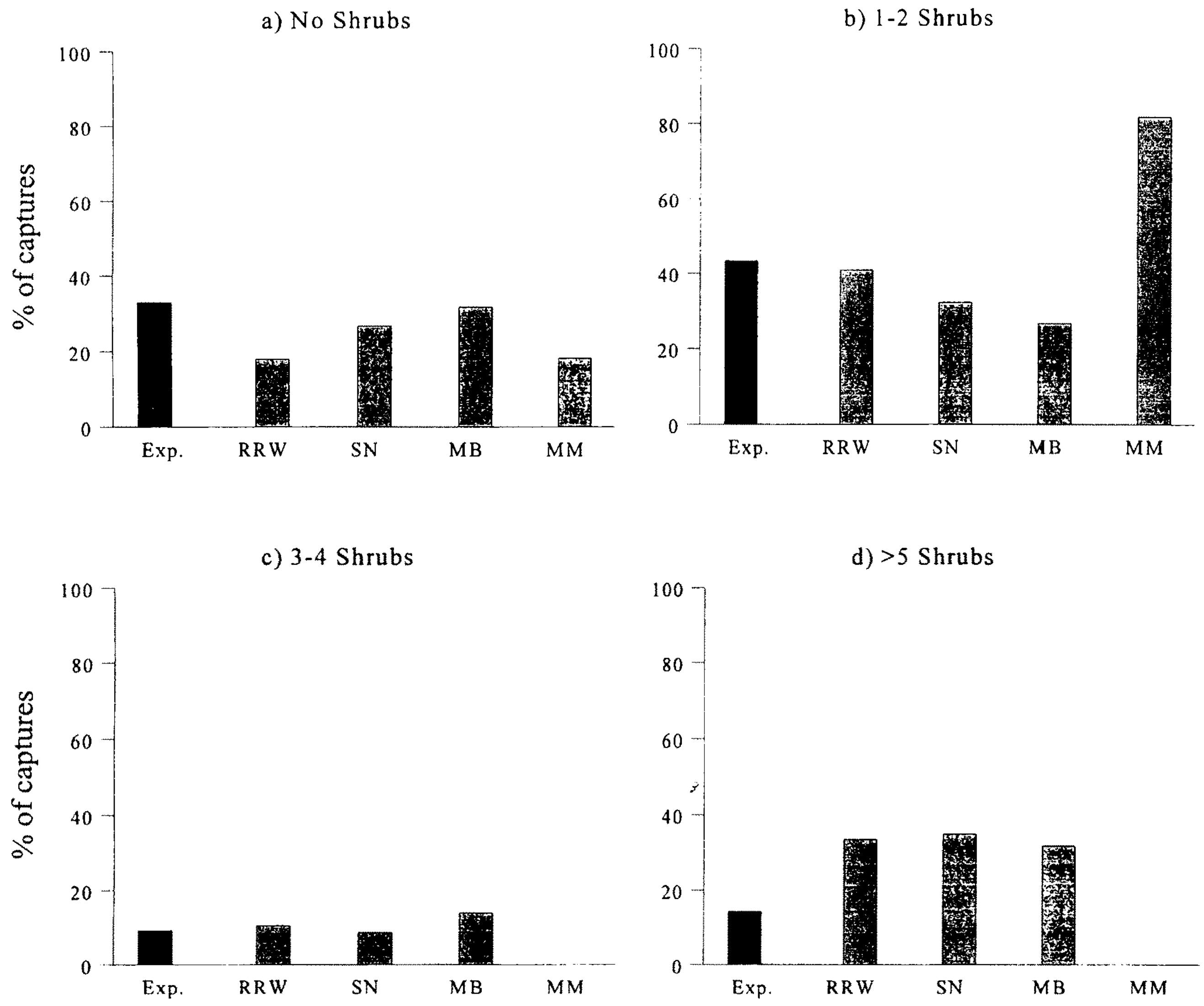


Figure 5.2 A comparison of the expected and observed percentages of captures in different categories of shrub cover (abbreviations as in the figure 5.1).

of the nearest tree (>30 cm) were also grouped into three categories <70 cm, 70-180 cm and >180 cm. The availability of each category of a variable was estimated as the percentage of the randomly sampled trap stations which fell under that category. This was considered the percentage of captures (out of total captures of a species) expected if the species did not show any selection with reference to that variable. A χ^2 test was done for each microhabitat variable, using the expected number of captures and the actual number of captures.

Shrub Cover

The expected and observed percentages of captures in the categories of shrub cover are given in Figure 5.2. The captures for all species were lower than expected when there were no shrubs around the trap station. When there were only one to two shrubs in the quadrat, the captures of *R. r. wroughtoni, S. niger* and *M. booduga* were lower than expected, but that of *M. musculus*, was considerably more. At medium shrub density (3-4 shrubs), the captures of *M. booduga* was greater than expected. At high shrub cover density (>5 shrubs), the captures of *R. r. wroughtoni*, *S. niger* and *M. booduga* were all greater than expected. The difference between expected and observed for all species together was highly significant (χ^2=148.06; *df*=9; P<0.001).

Canopy Cover

The canopy cover was generally high, with about 60% of the randomly sampled quadrats having a cover of more than 80% (Figure 5.3). At low canopy cover (<20%), there was

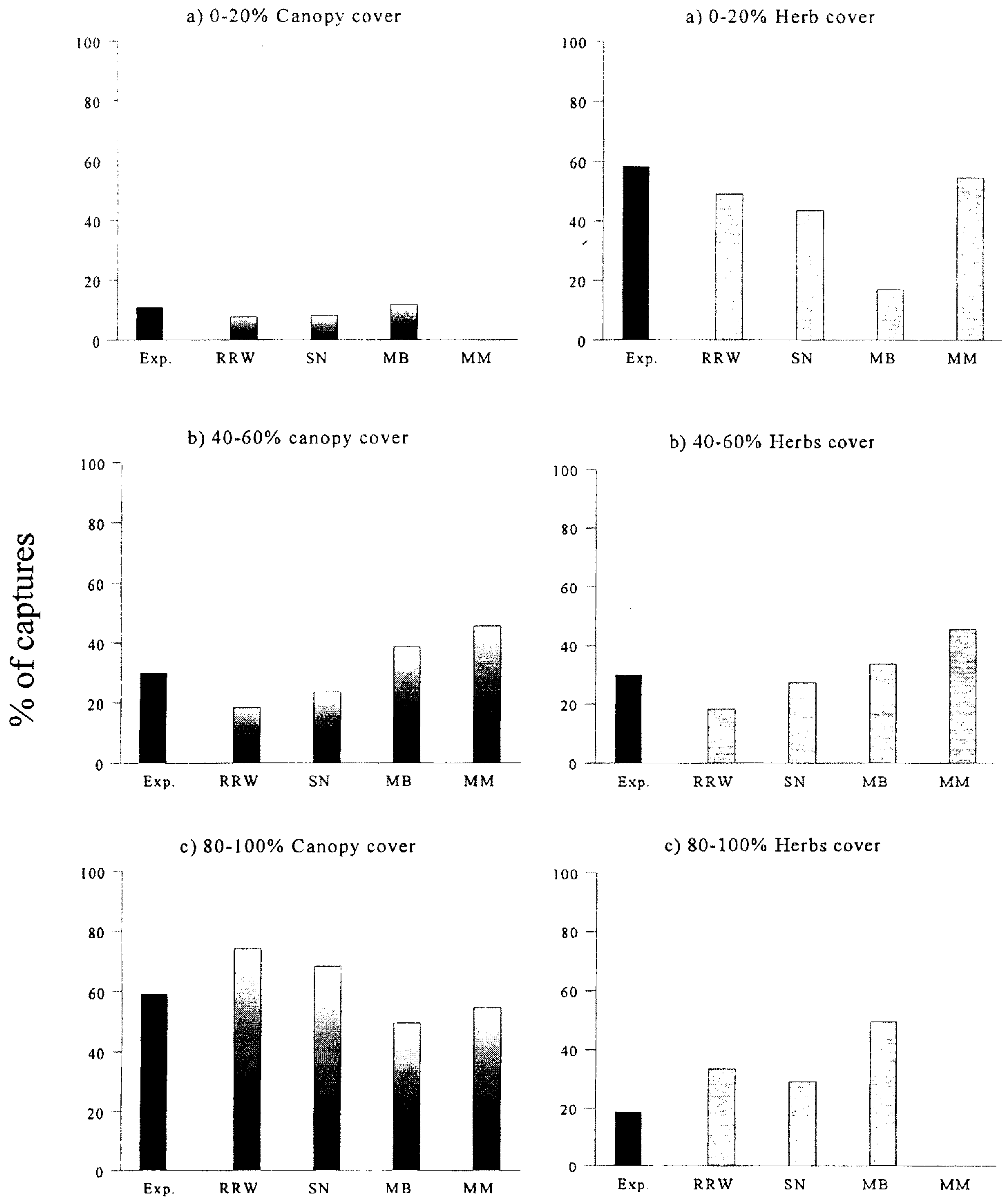


Figure 5.3 A comparison of the expected and observed percentage of captures in different categories of canopy cover and herb cover (abbreviations:Exp.=Expected percentage of captures out of total capture of a species, RRW=*R. r. wroughtoni*, SN=*S. Niger*, MB=*M.booduga*, MM=*M.musculus*).

difference among species in percentages of captures which were similar to the expected percentages. At medium canopy cover, *M. booduga* and *M. musculus* had greater captures than expected, while *R. r. wroughtoni* and *S. niger* had lower captures. At high canopy cover, the opposite was true. The difference between expected and observed captures with reference to canopy cover was, significant (χ^2=37.7%; *df*=6; P>0.001).

Herb Cover

Herb cover was generally low, with nearly 60% of the randomly sampled quadrats having a cover of only less than 20% (Figure 5.3). At low herb cover (<20%), the captures were lower than expected, especially in *R. r. wroughtoni, S. niger* and *M. booduga.* At medium herb cover, *M. booduga* and *M. musculus*, had more captures than expected, especially the latter species. At high herb cover all species, except *M. musculus*, had more captures than expected. The difference between expected and observed was significant (χ^2=147.10; *df*=6; P<0.001). Among the four species, *M. booduga* was more sensitive to herb cover, showing an increase in captures as herb cover increased (Figure 5.3).

Litter Cover

The capture rates of species varied significantly with litter cover (χ^2=94.57, *df*=6; P<0.001). The captures of *M. musculus* was greater than expected at medium litter cover, and that of *R. r. wroughtoni* and *S. niger* at high litter cover (Figure 5.4). *S. niger* seemed to be particularly sensitive to litter cover, the captures being considerably greater than expected at high litter cover.

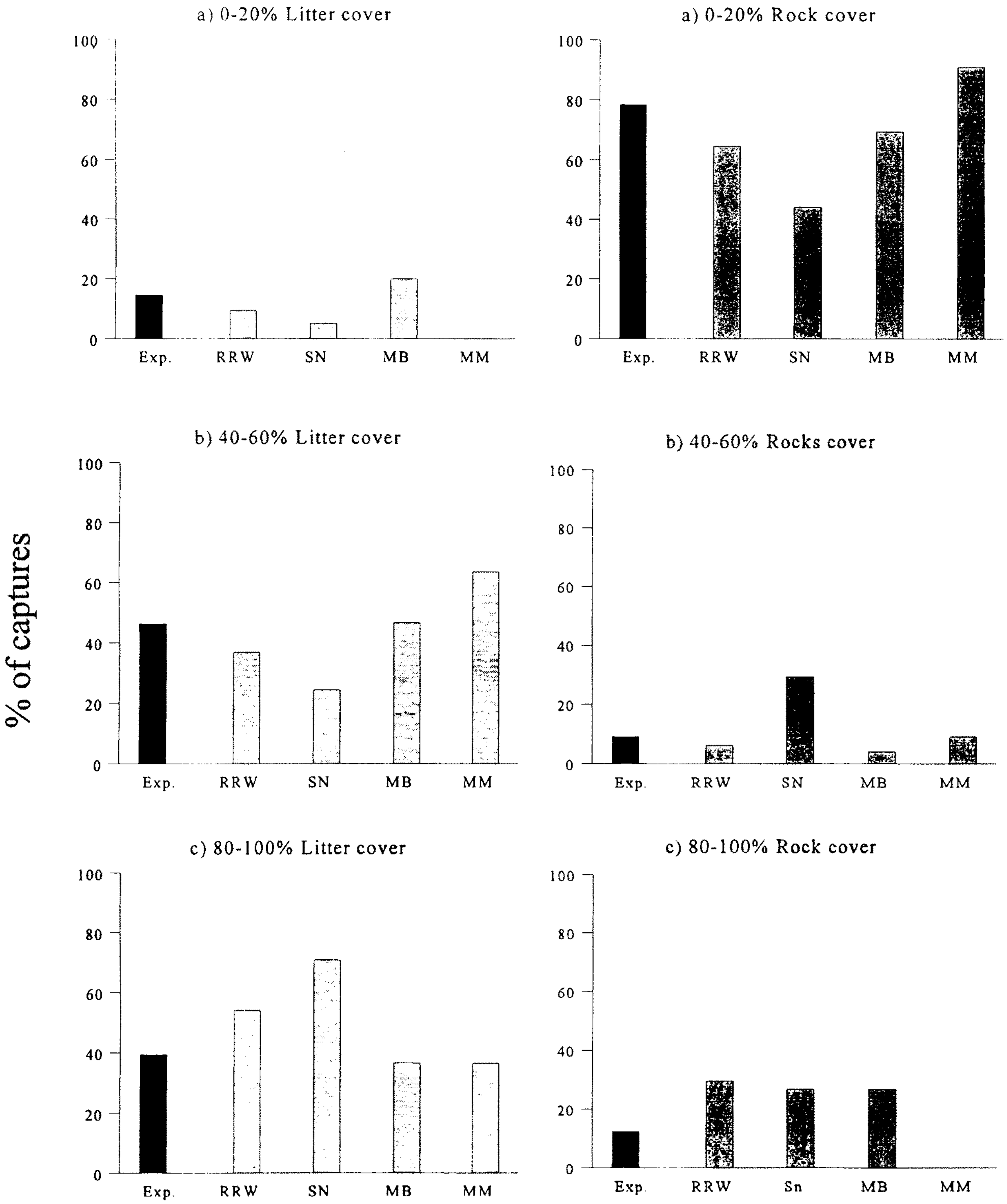


Figure 5.4 A comparison of the expected and observed percentages of captures in different categories of litter cover and rock cover (abbreviations:Exp.=Expected percentage of captures out of total capture of a species, RRW=*R. r. wroughtoni*, *SN=S. Niger*, MB=*M.booduga*, MM=*M.musculus*).

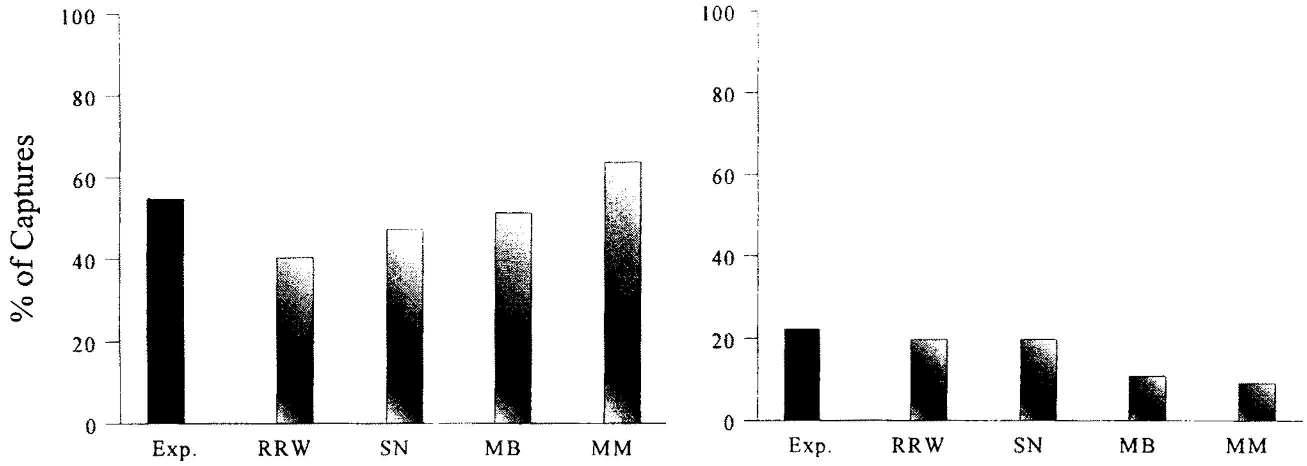
Rock Cover

Rock cover was generally low, with nearly 80% of the quadrats having a cover of less than 20%. Captures showed significant differences with rock cover (χ^2=217.92; *df*=6; P<0.001). The captures of *M. musculus* was greater at low rock cover. At medium rock cover, only *S. niger* had more captures than expected, the other species showing no difference from expected. At high rock cover, all species, except *M. musculus*, had greater captures. *S. niger* seemed to be the most sensitive to rock cover, especially in the preference for medium rock cover, while *R. r. wroughtoni* seemed to prefer high rock cover. *M. booduga* also seemed to prefer high rock cover.

GBH of Nearest Tree

The presence of large trees seemed to affect the captures of all species. Captures of *M. musculus* was greater than expected when small trees (<70 cm GBH) occurred at the trap site, while that of the other species were lower (Figure 5.5). When large trees (>180 cm GBH) were present, the captures of all species greater than expected, the difference being highest in the case of *R. r. wroughtoni*. With trees of medium GBH, the captures were lower than expected for all species. The overall difference was significant (χ^2=75.04; *df*=6; P<0.001).

A comparison the expected and observed percentage of captures with reference to categories within 11 microhabitat variables reveals some similarities as well as differences among the four species that were examined. While *R. r. wroughtoni*, *S. niger*



a) <70 cm of GBH

b) 70-180 cm of GBH

c) >180 cm of GBH

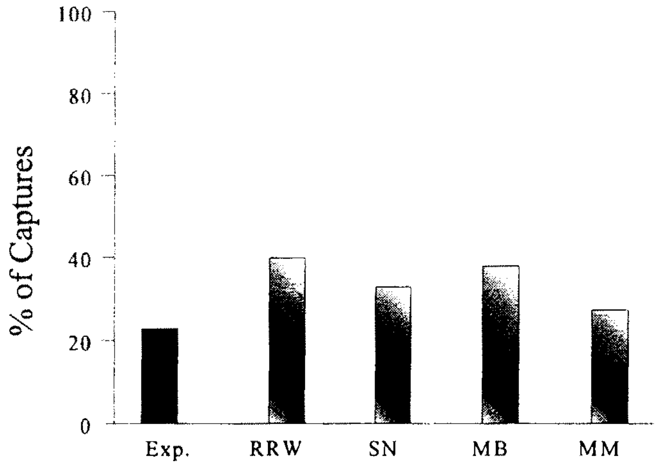


Figure 5.5 A comparison of the expected and observed percentage of captures in different categories of trees' Girth at Breast Height (GBH) (the abbreviations as in figure 5.4).

and *M. booduga* were caught more frequently in trap sites with trees, lianas, rattans, fallen logs, and buttressed trees, *M. musculus* was caught more frequently in sites without these. *R. r. wroughtoni* and *S. niger* were caught more frequently in sites with high shrub density, canopy, herb, litter and rock covers, and large trees. *M. booduga* also showed a similar pattern, but was caught more frequently than expected also from medium shrub density, and canopy and herb cover. In contrast, *M. musculus,* was caught from sites with low shrub density, rock cover and small tree, and medium canopy, herb and litter cover. Thus, *M. musculus*, showed distinctively different pattern compared to *R. r. wroughtoni* and *S. niger*, while *M. booduga* overlapped with *M. musculus* on the one hand and *R. r. wroughtoni* and *S. niger* on the other.

5.3.2 Logistic Regression

Since many of the 11 microhabitat parameters were likely to be interrelated, I attempted to reduce these into a relatively few but independent variables using Principal Component Analysis (PCA). However, five principal components with eigen values of greater than 1.0 accounted for only 50% of the total variance, the first component accounting for only 17%. This might have been due to the presence of a large number of zero values for many variables (*e.g.,* trees, lianas, buttress trees and rattans), and low variability in other variables. Data reduction through PCA was therefore not feasible. Instead, I used logistic regression, to examine the independent effect of each microhabitat variable in determining the occurrence of three species, *M. musculus* was excluded due to small sample size. Some variables were used as categorical variables after converting abundance into presence-absence categories. These included trees, rattans, buttressed trees, lianas and fallen logs. For these variables absence (0) was used as the indicator. For other six variables the measurements were cover in percentages (herb, rock, litter and canopy cover), or numbers (shrubs), and in the case of the largest tree, the GBH for given the difficulties in the interpretation of parameter interactions in a logistic regression (Hosmer and Lemeshow 1989; Norusis 1990). I assessed the importance of each independent variable by examining whether the slope was significantly different from zero. This is based on the *t* value obtained by dividing the slope by its standard error. Variables with significant slopes are taken as important in the selection of microhabitat by a species.

The results from the logistic regression are given in Table 5.1. The occurrence of rattans was the most important parameter that independently increased the captures of *R. r. wroughtoni,* as indicated by the significant slope (slope 1.5606, P=0.0044). Slopes of two other parameters were marginally significant, that of the presence of buttressed trees (slope 0.7612, P=0.0598) and number of shrubs (slope 0.0957, P=0.0683). The presence of buttressed trees, and rattans, and a greater density of shrubs, therefore, increased the capture of *R. r. wroughtoni.*

The slopes of four parameters were significantly different from zero in the case of *S. niger*. An increase in herb cover (slope 0.0301, P=0.000), litter cover (slope 0.0504, P=0.000), rock cover (slope 0.0555, P=0.000) and shrub density (slope 0.1433, P=0.0367), increased the captures of *S. niger.*

Table 5.1. The slopes and their significance values obtained from a logistic regression of the occurrences of three species on several habitat parameters.

	R.r.wroughtoni		*S. niger*		*M. booduga*	
Variable	Slope	P	Slope	P	Slope	P
Occurrence:						
Buttress tree	.7612	.0598	-.5466	.5355	-.1854	.8625
Fallen Log	-.3596	.3914	-.1235	.8255	-.0333	.9647
Lianas	.0619	.9132	-.4495	.5965	-4.4409	.7248
Rattan	1.5606	.0044	.6838	.5250	-5.0759	.7475
Tree	.0252	.9151	.2823	.3717	-.4799	.3794
Cover %:						
Canopy	.0013	.6870	.0071	.1169	-.0098	.0482
Herb	-.0020	.5910	.0301	.0000	.0368	.0000
Litter	.0025	.5176	.0504	.0000	.0070	.2693
Rock	-.0049	.4012	.0555	.0000	-.0241	.0978
No. of shrubs	.0957	.0683	.1433	.0367	.2049	.0145

Higher shrub density (slope 0.2049, P=0.0145) and herb cover (slope 0.0368, P=0.0000) also increased the captures of *M. booduga*, while greater canopy cover decreased it (slope -0.0098, P=0.0482). Rock cover also had a negative, marginally significant effect on the captures of *M. booduga* (slope -0.0241, P=0.0978).

The logistic regression brings into sharper focus the factors that influence the captures of the three species. Increasing shrub density increased the captures of all three species; while presence of buttressed trees, and rattans were important for *R. r. wroughtoni*, herb cover and rock cover were important for *S. niger* and *M. booduga*; the latter variable acting in opposite direction in the two species. In addition, litter cover was important for *S. niger*, while canopy cover had a negative effect on *M. booduga*.

The results from the logistic regression by and large confirm that from the bivariate analysis of the independent variables. For example, the largest difference between the expected and observed captures of *R. r. wroughton*i were in the case of buttressed trees and rattans. Similarly, the effects of rock and litter cover on *S. niger* were also evident in the bivariate analysis. However, some of the nonlinear relations would not have been taken into account in the logistic regression. For example, the captures of *M .booduga* were highest at medium canopy cover.

5.4 DISCUSSION

Bivariate analysis of the data showed considerable overlap among species in their response to variation in the microhabitat parameters, even though some differences among them were also evident. In general, *R. r. wroughtoni, S. niger* and *M. booduga* seemed to respond similarly to microhabitat variation. However, *M. booduga* responded nonlinearly to some variables, captures being greater at moderate shrub densities, canopy and herb cover. Captures of *M. musculus*, on the other hand, were greater at low values of most variables. Logistic regression, seemed to segregate species further, with buttressed trees, rattans and shrub density being important for *R. r. wroughtoni*, rock, litter and herb cover, and shrub density being important for *S. niger*, and canopy, herb and rock cover and shrub density being important for *M. booduga*. The only variable to which all species respond was shrub density. The opposing responses were by *S. niger* and *M. booduga* to rock cover. However, the low slope values for most parameters, and nonlinear responses to some variables by *M. booduga* which was evident in bivariate analysis, make the species segregation in logistic regression less reliable.

Even though microhabitat selection might reflect feeding habits, foraging behaviour and anti predation strategies (Canova and Fasola 1991), functional relationships of microhabitat selection have rarely been demonstrated in small mammals. As in this study, results from elsewhere also demonstrate considerable overlap among the small mammals in the use of some microhabitats, and segregation in others (*e.g.,* Brown 1987; Adler 1988; Scott and Dueser 1992; Batzli and Lesieutre 1995; Morris 1996; Shanker 1998).

The overlap in the use of microhabitats among different species has been attributed to the possibility that trapping may not reflect microhabitat affinities at a finer level (Price 1978). Microhabitat segregation has been reported to be an important factor in structuring small mammal communities in several studies (*e.g.*, Price 1978; Brown 1987; Adler 1988; Bowers and Flanagan 1988).

In Chapter 3, it was demonstrated that the abundance of *R. r. wroughtoni* declined and that of *M. booduga* and *M. musculus* increased, as habitat status declined and habitat heterogeneity increased due to rain forest fragmentation. Whether this change is due to microhabitat changes or due to competition is debatable. There was a clear segregation between *R. r. wroughtoni* and *M. musculus* in microhabitat selection. *R. r wroughtoni* responded negatively to a reduction in shrub density, rock cover, GBH of trees, lianas, buttressed trees and rattans. In contrast, *M. musculus* responded positively to these changes. Therefore, it is not likely that competition between two species is an important factor in the decline of the abundance of *R. r. wroughtoni* in forest fragments. On the other hand, *R. r. wroughtoni* and *M. booduga* responded similarly to the variation in several microhabitat variables, especially shrub density, herb cover, rock cover, and the presence of trees, buttressed trees, lianas, rattans and fallen logs. The invasion by *M. booduga* could, therefore, competitively displace *R. r. wroughtoni.* It is noteworthy, however, that *M. booduga* often showed a nonlinear response to variation in several of these parameters, captures being highest at moderate values (*e.g.,* shrub density, herb cover, and rock cover). It is likely that competitive advantage rest with *M. booduga*, when conditions are less optional for *R. r. wroughtoni* as in habitats with moderate values of the above parameters. Competitive exclusion of *R. r. wroughtoni* by *M. booduga* are represented by moderately disturbed, medium sized forest fragments. In the larger and relatively undisturbed habitats, *R. r. wroughtoni* might be at a competitive advantage, conditions being near optimum for the species.

The decline in the abundance of the subspecific endemic *R. r. wroughtoni* from forest fragments might also be associated with the loss of its microhabitats such as buttressed trees, rattans, and shrub cover rather than due to competition with invading species such as *M. booduga* and *M. musculus*. *M. musculus* is a human introduced invaders in many parts of the world, where it may not have any affect on the native species (Bennett 1990;

Soulé *et al.* 1992; Amarasekare 1994). In the upper Nilgiris in the Western Ghats, Shanker (1998) did not find competition as a major factor affecting community structure.

S. niger also showed considerable overlap in microhabitat selection with both *R. r. wroughtoni* and *M. booduga*. However, it did not show any major change in its abundance following fragmentation. Being an insectivore, it might have little competitive interaction with rodents, in spite of considerable microhabitat overlap. Shrews, have been reported more generalists in its microhabitat preferences than rodents (Canova and Fasola 1991). This might explain their remarkable resistence to habitat fragmentation.

Whether the changes in the community structure in small mammals due to habitat fragmentation and associated factors, result from changes in microhabitat structure or competition can be assessed only through experimental removal of animals and habitat manipulation. The role of competition and microhabitat selection in structuring small mammal communities has been a subject of considerable debate. Several studies have highlighted that microhabitat selection and not competition is the important factor (Dueser and Shugart 1978; Dueser and Porter 1986; Muruá *et al.* 1987; Adler 1988; Canova and Fasola 1991; Scott and Dueser 1992; Amarasekare 1994), and *vice versa* (Price 1978; Schoener 1983; Walter and Halley 1984; Bowers and Flanagan 1988; Kotler and Brown 1988; Probert and Litvaitis 1996). It has also been argued that other factors such as predation and disease, might also be important, and that the relative importance of the these might vary from place to place, as well as time (Batzli 1985; Batzli and Lesieutre 1995; Morris 1996; Shanker 1998). It is also likely that the microhabitat parameters that are often used may not reflect the finer levels of selection that animals make (Price 1978). In fact, empirical models of microhabitat selection have often failed to reliably predict the occurrence of small mammals (Gore 1988). Similarly, competition might also be difficult to determine in the field.

The loss of specific microhabitats has seldom been associated with the extinction of species. In England, the disappearance of the dormouse (*Muscardinus avellanarius*) from forest fragments has been associated with the loss of specific microhabitats such as tree holes and three food plants. In Kalakkad-Mundanthurai Tiger Reserve in the Western Ghats the occurrence of the endemic Malabar spiny dormouse has been associated with buttressed trees and lianas (Anon. 1997). In Anaimalai Hills, this species was caught only from the largest and least disturbed fragment, Akkamalai shola. The other fragments that I sampled had been logged, which removed large buttressed trees and lianas. This might explain the absence or low occurrence of this species from these fragments.

The low occurrence of *G. ellioti, R. blanfordi* and *V. oleracea* might also be due to the lack of specific microhabtiats. All these species are widely distributed in India, but were caught infrequently from forest fragments. However, there is no information on the microhabitat preference of these species. Most of the captures of *G. ellioti* were from thick undergrowth of weeds such as *Lantana, Eupatorium* and *Solanum*, often along road sides. Prakash *et al.* (1995) have also reported that this species occur under thick undergrowth. This microhabitat is currently limited to few patches in the highly disturbed fragments.

5.5 SUMMARY

1. In this Chapter, I examined the differences among four species in their responses to variation in several microhabitat parameters. The microhabitat variables were measured from $1m^2$ quadrats centred on trap stations. The measurements were made from all trap stations with capture, and several random stations. The response to each microhabitat variable was examined by comparing expected captures estimated from the random stations with observed captures. In total 1255 trap stations were sampled.

2. Bivariate analysis showed considerable overlap in the responses of *R. r. wroughtoni* and *S.niger* to variation in almost all macrohabitat variables. Captures of both the species were greater in quadrats with trees, buttressed trees, liana, rattan and fallen logs. Captures of both species also increased with an increase in canopy, herb, litter and rock cover and shrub density. *M. booduga* also showed a similar response, but captures were greatest at medium values of some variables such as shrub density and herb cover. In contrast, captures of *M. musculus* were greater at low values of most of these variables, and at mid values of the others.

3. Logistic regression segregated these species better, with reference to their response to the microhabitat variables. *R. r. wroughtoni* showed significant response to the presence of rattans, buttressed trees and shrub density. *S. niger* responded to shrub density, rock cover, litter cover and herb cover. *M. booduga* responded to shrub density, rock cover, herb cover and canopy cover, the last of these negatively. Thus, shrub density was the only variable to which all three species responded. However,

nonlinear response of *M. booduga* to some variables may not be represented adequately in the logistic regression.

4. Since *R. r. wroughtoni* and *M. musculus* are almost totally segregated in their microhabitat selection, competition among them is unlikely to be a factor in the decline of *R. r. wroughtoni* in forest fragments. Overlap with *M. booduga* in microhabitat selection can lead to competition among them in moderately disturbed fragment, and thus to the decline of *R. r. wroughtoni.* However, loss of optimal microhabitats of *R. r. wroughtoni* could also be a reason.

5. The insectivorous, *S. niger* is more generalist than rodents, and is unlikely to be affected by competition with invading or resident rodents, in spite of considerable overlap among them in microhabitat preferences.

6. CONCLUDING DISCUSSION

6.1 Species Richness

Studies on small mammals conducted in recent years in the Western Ghats, especially in the tropical rain forests, shows a generally impoverished fauna in the taxa that I studied (Families Muridae and Muscardinidae; and Order Insectivora). As discussed in Chapter 3, this might reflect the impoverished fauna in these taxa in India. What is perhaps surprising is that species richness and endemism among these taxa in the Western Ghats seems to be lower than in the other nonvolant mammals, noted for their low species richness and an endemism of only about 10%. Other mammalian Families exhibit greater species richness and endemism in the Western Ghats: Cercopithecidae - four species and two endemics; Viverridae - four species and two endemics; Herpestidae - four species and two subspecific endemics; and Sciuridae - at least seven species with two full endemics and three subspecies endemics. In comparison, Muridae one of the most speciose Families of the world, seem to be particularly species poor in the Western Ghats. Only 13 murid species have been reported from the Western Ghats, most of which are human introduced and widely distributed, and none are endemic at the subspecies level. Among Insectivora eight species have been reported to occur in the Western Ghats, even though the occurrence of the tree shrew (*Anathana ellioti*) and the small footed-hedgehog (*Erinaceous micropus*) in Western Ghats proper is doubtful. Three of these are probably endemic to the Western Ghats (*S. niger, S. dayi* and *S. perroteti*). This faunal impoverishment may also reflect the faunal distribution in India.

There are, however, two reasons to suspect that species richness in these taxa might be greater than currently known. The first is the poor taxonomic expertise that has been available on these faunal groups. A systematic taxonomic evaluation of specimens might increase the number of species that occur in the Western Ghats. Secondly, a systematic collection of these taxa has never been attempted in the Western Ghats. Most of the recent field studies have used only standard Sherman traps, which might be insensitive to species with the low body weight. Different kinds of traps, such as Sherman traps, pitfall traps and snap traps are known to sample different species, with snap traps giving more captures as well as species (Woodman *et al.* 1995; Ishwar Prakash pers.com). The use of snap traps, however, kills not only the target animals, but others too, perhaps the reason why they have not been widely used in the Western Ghats. In this study an Indian pitta (*Pitta brachyura*) was trapped in a Sherman trap and later released (Prabhakar 1998), which would not have been possible with snap traps. However, the terrestrial rodent and shrew fauna need to be sampled systematically using a variety of techniques, before the surprising faunal poverty of these taxa is accepted.

6.2 Changes in the Community Structure

This and other recent studies reveal the drastic changes taking place in the small mammal communities in rain forest fragments. There has been a general increase in the species richness of this community in forest fragments, compared to the contiguous forest. This has been due to the invasion of several species, most of which are human introduced. These include *M. booduga*, *M. musculus*, *R. blanfordi*, *V. oleracea* and *G. ellioti.* These are associated with human settlements, with agricultural fields, or drier forest types.

It is likely that some of these regularly disperse from human settlements and orchards around them into fragments and survive in fragments because of the source-sink relationship (*e.g., M. musculus*). At least one species *G. ellioti,* occur only in weed dominated edges of fragments or along road sides. With on going degradation of habitat, it is likely that *M. musculus*, *R. blanfordi*, *V. oleracea* and *G. ellioti* which were caught infrequently, would increase in abundance.

While the above species have increased in abundance, two species show a definite decline. *P. lasiurus* seem to have been the most severely affected. There is also decline in the absolute and relative abundance of *R. r. wroughtoni.* The disappearance of *P. lasiurus* from the smaller fragments seem to be due to the loss of its microhabitats, consisting of buttressed trees and lianas, which are typical of undisturbed rain forest. Being a highly arboreal species, canopy contiguity is perhaps very important for this species. Its rarity or absence from disturbed forest fragments is interesting since it has been reported to be a pest in arecanut -pepper orchards along the foothills of the Western Ghats.

It is worth examining whether *R. r. wroughtoni* would have increased in abundance, in the absence of invading species. It is interesting to note in this context that several species that occur naturally in the rain forests have increased in abundance following habitat fragmentation, and habitat disturbance. In the study area, these include the Malabar giant squirrel and flying squirrels (Umapathy 1998).

Whether the decline in the absolute and relative abundance of *R. r. wroughtoni* in forest fragments is due to a decline in its preferred microhabitats or due to competition from invading species is remain un resolved. There is considerable overlap between this species and *M. booduga* in the use of microhabitats. However, for some parameters, such as shrub density and herb cover, *M. booduga* seemed to also prefer moderate values, while *R. r. wroughtoni* preferred higher values. Thus decline in *R. r. wroughtoni* is more likely to be related to changes in microhabitats in fragments than to competition. *M. musculus* showed very little overlap with *R. r. wroughtoni* in microhabitat selection and thus might offer no competition to *R. r. wroughtoni.*

6.3 The Role of Habitat Heterogeneity

This study demonstrates the role of spatial heterogeneity in determining species richness in small mammals. Habitat heterogeneity, which was highest in the medium sized forest fragments, facilitated an increase in microhabitats which were occupied by invaders such as *M. booduga* and *M. musculus*. It has been reported from elsewhere that the small mammal community is a function of physical structure or heterogeneity of the habitat, the regional species pool and colonisation abilities of species (August 1983; Adler 1988; Kerley 1992; Steen *et al.* 1996). What fragmentation and associated disturbance have done is to increase the heterogeneity of the habitat and also to facilitate the colonisation of the newly created microhabitats by introducing several commensal species.

6.4 Synchrony

This study has also demonstrated one of the typical and expected impacts of habitat fragmentation, on demography. There was a general absence of synchrony among the isolated populations of *R. r. wroughtoni* and *M. booduga* in most of the demographic parameters that were examined. Unlike contiguous population which show synchrony at a regional scale, fragmented populations seem to be tracking resources at the local scale, probably of their habitat fragments. The asynchrony also demonstrates the lack of dispersal among fragments. *S. niger*, an insectivore, showed some demographic synchrony among fragments. This might indicate either dispersal among fragments, or synchrony in the abundance of insects which probably respond to environmental parameters such as temperature and humidity.

6.5 Ecological Implications

There are several reasons to expect that the changes in the small mammal community would have a cascading effect on the ecosystem in which they live. Rodents are major seed predators, therefore, changes in their abundance could affect the long term vegetation dynamics of their habitat (see Osankoya 1994). Rodents also form a major prey base for several reptilian, avian and mammalian predators. For example, nearly 33% of the small carnivore scats in Thailand contained rodent hairs (Walker and Rabinowitz 1992), up to 90% of the diet of the lesser cats may be murids (Mukherjee in prep.). The Western Ghats has 14 species of small carnivores, consisting of four each of civets, mongoose and lesser cats and one species each of marten and ratel. Murid rodents most likely form an important prey base of the lesser cats, and probably of other species such as the marten and the small Indian civet. Changes in the community structure of rodents, therefore, might have important implications for the small carnivore community.

Another major ecological consequence of the changes in community structure of rodents, especially invasion by several species, in the introduction of new parasites and pathogens. Parasites and pathogens, especially those introduced anew by invading carriers, can pose major threats to a community in a restricted environment (McCallum and Dobson 1995; Holmes 1996). Animals in restricted environments, are often under considerable stress due to poor nutrition and greater interspecific competition, which make them more vulnerable to introduced parasites and pathogens. A reduction in the number of parasitic species, with an increase in the percentage of infected animals and parasitic load, has been reported in fragmented populations of bank voles (Kozakiewicz 1993). As populations become increasingly fragmented and isolated, diseases come to play a greater role in community ecology and survival of species (Holmes 1996). It is in this context that the invasion of forest fragments in the Western Ghats by commensal species become important. *M. booduga* a major invader, has been reported to be capable of circulating Kysanore Forest Disease (Krishnaswami 1965), which killed several monkeys in northern Western Ghats in the 1960's. *M. platythrix* another invading species reported from the Anaimalai Hills (Chandrasekar-Rao and Sunquist 1996), is a potential carrier of plague (Krishnaswami 1965). Thus, the invasion of murids into the fragmented landscape of the Western Ghats may pose serious health hazards to other taxa, as well as to human beings.

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APPENDIX-I

Rodents and Insectivores Reported from the Western Ghats.

S.No.	Order	Family	Common Name	Species Name	Distribution
1	Rodentia	Sciuridae	Large Brown Flying Squirrel	*Petaurista petaurista*	Peninsular India, some part of northeastern India, Sri Lanka and Myanmar.
2	”	”	Small Travancore Flying squirrel	*Petinomys fuscocapillus*	South India and Sri Lanka
3	”	”	Malabar Giant Squirrel	*Ratufa indica*	The peninsular India.
4	”	”	Grizzled Giant Squirrel	*R. macroura*	The Western, Eastern Ghats of India and Sri Lanka
5	”	”	Three-striped Palm Squirrel	*Funambulus palmarum*	South India and Central India
6	”	”	Western Ghats Striped Squirrel	*F. tristriatus*	The Western Ghats (Bombay region south to Kerala)
7	”	”	Layard's Striped Palm Squirrel	*F. layardi*	The Western Ghats (Kerala) and Sri Lanka
8	”	”	Dusky Striped Squirrel	*F. sublineatus*	The Wester Ghats and Sri Lanka
9	”	Hystricidae	Indian Porcupine	*Hystrix indica*	Through out India, Sri Lanka and West Pakistan to Syria, Lebanon and Israel in the west.
10	Rodentia	Muscardinidae	Malabar Spiny Dormouse	*Platacanthomys lasiurus*	The Western Ghats.

S.No.	Order	Family	Common Name	Species Name	Distribution
11	Rodentia	Muridae	Long-tailed Tree Mouse	*Vandeleuria oleracea*	Peninsular India some parts of north-eastern India
12	”	”	Common Indian Rat	*Rattus rattus*	Through out India
13	”	”	White-tailed Wood Rat	*R. blanfordi*	Peninsular India and parts of northeast and Sri Lanka
14	”	”	Ellerman's Rat	*R. elvira*	South India
15	”	”	The Metad	*Millardia meltada*	Peninsular India and Nepal
16	”	”	House Mouse	*Mus musculus*	Throughout the world
17	”	”	Commmon Indian Mouse	*M. booduga*	Peninsular India, some parts of northeast, W. Pakistan and Myanmar
18	”	”	Fawn-colored Mouse	*M. cervicolor*	Peninsular India, Nepal and Myanmar
19	”	”	Bonhote's Mouse	*M. famulus*	South India, some parts of northeast, Myanmar, China and Thailand
20	”	”	Brown Spiny Mouse	*M. platythrix*	Peninsular India, and W.Pakistan
21	”	”	Indian Bush rat	*Golunda ellioti*	Peninsular India and part of northeast.

S.No.	Order	Family	Common Name	Species Name	Distribution
22	Rodentia	Muridae	Lesser Bandicoot Rat	*Bandicota bengalensis*	Throughout India, Nepal Sri Lanka, Myanmar, Malaysia and Indonesia
23	”	”	Large bandicoot Rat	*B. indica*	South Asia
24	Insectivora	Soricidae	Common Musk Shrew	*Sorex caerulescens*	Generally India
25	”	”	Mouse-coloured Shrew	*S. murinus*	Generally India
26	”	”		*S. montanus*	The Western Ghats
27	”	”		*S. dayi*	The Western Ghats
28	”	”	Rufescent Shrew	*S. serpentarius*	Southern India, Myanmar and Sri Lanka
29		”	Nilgiri Wood-Shrew	*S. niger*	The Western Ghats
30	”	”	Nilgiri Pigmy-Shrew	*S. perroteti*	Southern India
31	”	Erinaceidae	Small-footed Hedgehog	*Erinaceus micropus*	South India
32	”	Tupaiidae	Elliot's Tree Shrew	*Ananthana elliot*i	South India, parts of Orissa and Bihar.

Source; Ellerman (1961); Biswas & Tiwari (1966); Sterndale (1982); Kumar *et al.* (1998); Shanker (1998).

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