BIODIVERSITY AND CONSERVATION OF A CULTURAL LANDSCAPE IN THE WESTERN GHATS OF INDIA

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ABSTRACT

Biodiversity and Conservation of a Cultural Landscape in the Western Ghats of India

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The effects of landscape modification on the diversity of three contrasting groups of organisms – trees, birds and macrofungi that are biologically and ecologically different – are presented in this study. Three competing heuristic frameworks – island biogeography, metapopulation theory and landscape ecology – which have been advanced to study the fragmented landscapes, are discussed. The most suitable framework for studying biodiversity of the three groups of organisms in landscapes that are modified but not fragmented, is determined. Two contrasting approaches to biodiversity conservation – strict protection, and community-based conservation – are assessed and the conditions where they are suitable are determined.

Methods were used for rapid assessment of biodiversity and evaluation of landscape structure in Kodagu district, in the Western Ghats of India, where few data are available on the distributions of trees, birds and macrofungi and where detailed spatial information is not easily accessible. The comparisons of diversity across the three groups of organisms were made based a maximum value that permitted calculation of rarefied species richness for all samples. The shaded coffee cultivation in the study area has been instrumental in maintaining a continuous cover of native trees, despite long-term management by humans. As an ecological consequence of high tree cover, recognised landscape measures such as patch area and isolation are inadequate for explaining the patterns of diversity in Kodagu, unlike in many other insularised fragmented landscapes. A slight climatic gradient and an increasing degree of human disturbance in patches farther from the contiguous forest reserve influence tree diversity, in addition to the isolation of the patches from the reserve. The intervening matrix of coffee plantations is not a strong barrier to the movements of many forest-dwelling birds. The influence of landscape structure on moisture content of the air affects sporocarp production of macrofungi. The integrity of the surrounding landscape matrix has an influence on diversity of birds and macrofungi within patches.

There is no congruence in diversity of the three groups of organisms studied. It is inferred that their differences in biological and ecological characteristics as well as their ranges of dispersal are responsible for the distinct patterns of distribution in the landscape. It is proposed that the strategy for biodiversity conservation should consider landscape-level conservation in order to capture the range of biodiversity that exists outside the formal reserve network in Kodagu. While maintaining the integrity of the strictly protected forest reserve; of traditionally conserved sacred forests; and of privately owned coffee plantations is essential, conservation practice should emphasise the involvement of local communities in management of landscape rather than imposing biodiversity conservation by force.

Dedication

To the loving memories of

my grandfather,

Ramchandra Shankar Bhagwat,

who cherished traditional values while he wrote about modern science;

my grandmother,

Pramila Anant Kodolikar,

who epitomised determination, as is required for completing a task like this,

&

my uncle,

Ramesh Anant Kodolikar,

who embodied hard work, another of the qualities essential for writing a thesis.

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Shonil Bhagwat, April 2002, Oxford.

CONTENTS

1 . INTRODUCTION

The loss of forest biodiversity is one of the priority issues discussed during the sixth meeting (held between 7 and 19 April 2002) of the Conference of the Parties on the Convention on Biological Diversity (CBD 2001). According to the United Nations' estimates, about 0.5% of the world's tropical forests disappeared each year throughout the 1990s (FAO 2001). WWF, one of the leading non-governmental organisations working for conservation, estimates that "every minute of every day we are losing some 26 hectares of forests, somewhere in the world due to threats such as illegal logging, land clearance for agriculture, road building, mining, and forest fires" (WWF 2001). Clearly, loss of forest due to human activities resulting in the fragmentation of landscape is one of the top current global conservation concerns.

In response to this problem, the designation of protected areas is increasing at a faster rate than ever before – there are now over 30,000 such areas, covering nearly one tenth of the Earth's land surface, which represents a conservation estate equivalent to the combined areas of India and China (FAO 2001; WCPA 2001). However, many of these areas are facing several challenges to their existence due to the resentment that such an approach invites from the local people, who depend on the resources within protected areas for their livelihoods (ICPL 2001). In relation to these threats and recognising that these special areas can only achieve their conservation objectives within a context of local acceptability, a new conservation paradigm is widely promoted – in order to be effective, the management of protected areas must be based on a collaborative approach with local people (*e.g.* McNeely 1996; Adams 1998).

In this chapter, the effects of landscape fragmentation on forest biodiversity are reviewed, the conceptual framework surrounding the subject is explored and findings of empirical research on fragmentation are analysed. Further, the rationale behind the two current conservation paradigms embodied in strict protection of natural resources and community-based conservation are examined. This is followed by a discussion of the relevance of contemporary fragmentation research and the approaches to biodiversity conservation for the present study.

1.1 Landscape fragmentation and modification

Human activity is unarguably the most important factor in contemporary alteration of natural landscapes (Morris 1995). Direct removal of natural habitat from the landscape is the most visible effect of human activities. Such activities create gaps in the continuous habitat; the gaps increase in size while the continuous habitat shrinks with increasing intervention by man. This gradual process of breaking up of natural landscape into small patches of habitat is called fragmentation (Forman 1995). Human activities such as clearing forests for cultivation of arable crops lead to perforation of landscape while construction of roads result in the dissection. Growing human settlements cause the shrinkage of natural habitat. A combined effect of all these processes is summarised in the term fragmentation. Thus fragmentation involves a combination of the loss of natural habitat and isolation of fragments (Wilcox 1980; Saunders *et al.* 1991; Murcia 1995; Borges 2000).

However, less intensive activities such as selective logging or agroforestry may cause just modification of landscape rather than fragmentation. For example, Johns (1991) points out that a landscape with a mosaic of patches of unlogged forest, regenerating forest and open areas forms more heterogeneous habitat in comparison with a landscape with habitat 'islands', predominantly because of greater diversity of vegetation in the former. Thus, in a modified landscape, there is spatial variation in habitat quality with no apparent barriers between patches (*e.g.* Opdam *et al.* 1993). Such mosaics will be expected to lose habitat specialist species in a similar way to fragmented landscapes. However, mosaic habitats may, if areas of natural forest are of reasonable size, be less susceptible to physical changes, such as in microclimate, than are forest isolates. As a result, the intensity of physical changes may be lower and a reduced number of species will face habitat loss.

In forested landscapes, for specialist forest-dwelling organisms, fragmentation and modification means reduction in the size of their habitat, which results in:

1. Reduction of population size for plants (immobile organisms);

2. Over-crowding and restriction on the movements across the landscape matrix, for animals (mobile organisms).

In contrast, habitat-generalist species that are pioneers will be benefited by fragmentation and modification because of the creation of habitats that are suitable for their colonisation. This will result in the increase in population sizes of such species.

As a consequence of habitat transformation in modified landscapes and its effects on habitat-specialist and generalist species, the proportional distribution and abundance of such species will be altered, ultimately causing changes in community diversity. The effects are at two scales:

1. Patch-scale effects (*e.g.* edge effects): include changes in the physical environment, namely, temperature, humidity, sunlight *etc.*;
2. Landscape-scale effects: include reductions in the areas of habitat-patches, changes in their shapes, isolation *etc.* as well as changes in the habitat quality of the surrounding landscape matrix along with patches.

Most of the published literature on forest fragmentation addresses the effects of insularisation of forested landscapes on biodiversity. The literature is reviewed and the implications of the results for landscapes that are modified but not fragmented, are discussed in the following sections.

1.1.1 Effects of fragmentation on physical environment in patches

The modification of the physical (and consequently biological) environments within a forest patch occurs mainly at new edges between forest and non-forest habitats. Edges are exposed to a hostile physical environment that causes a reduction in the effective area of a habitat patch (Kapos 1989; Saunders *et al.* 1991; Laurance *et al.* 1997; Meffe & Carroll 1997). Current research has documented five primary physical variables affected by forest fragmentation: air flow (wind), sunlight (photosynthetically active radiation), temperature, humidity and soil moisture (Freidenberg 1998). Forest edges experience increased airflow, sunlight and temperature while humidity and soil moisture are reduced (Geiger 1965). The scientific literature on fragmentation suggests that many organisms may be affected by the changes in their physical surroundings (Saunders *et al.* 1991; Murcia 1995). At a patch-scale, it has been shown that some habitat-sensitive insects, birds and small mammals are edge avoiders and their abundances decrease at edges (Laurance 1997a). At the same time, the abundances of invasive species increase (Lynam 1997; Viana *et al.* 1997). As a result, total diversity may increase at edges; however, such increase is due to the arrival of invasive species and persistence of habitat-generalist species.

Edge effects are known to alter forest structure (Kapos 1989). This is believed to result in an increase in stem density (Laurance *et al.* 1997), decrease in basal area (Bierregaard *et al.* 1992), high variability in canopy height and closure (Laurance 1997a) and an increase in the density of lianas (Viana *et al.* 1997). Furthermore, where a landscape is highly man-made, the effects of forest management will compound the effects of the altered physical environment on forest structure. However, in modified landscapes, edge effects may occur at a lower intensity than in fragmented landscapes, because the boundaries between patches and the surroundings in the former are not as distinct as in the latter.

1.1.2 Effects of fragmentation on patches and the surrounding landscape

Organisms living in fragmented landscapes are affected by the combined interaction among three factors: characteristics of a habitat patch, its distance from other similar patches in the landscape or a larger patch of the same habitat, and the landscape-matrix between patches (*e.g.* Opdam *et al.* 1993; Gustafson 1998). These three factors are important in determining diversity within a patch as well as similarity between different patches of the same habitat. Three competing heuristic frameworks have been advanced to address the biological effects of patches in fragmented and modified landscapes:

1. Island biogeography (MacArthur & Wilson 1967);
2. Metapopulation theory (Hanski & Gilpin 1991);
3. Landscape ecology (Forman 1995).

A detailed review of the three heuristic frameworks, which have now grown into separate approaches in fragmentation research, is beyond the scope of this work. The suitability of the three prominent approaches for research in modified landscapes is examined in this section.

Island biogeography

MacArthur & Wilson's (1967) Equilibrium Theory of Island Biogeography (island biogeography theory) proposed that the number of species on an island represents the balance between extinction and immigration. The theory predicted that large islands that are nearer to the mainland would have more species than smaller ones that are farther away. In a fragmented landscape, patches can be imagined as islands of habitat in a sea of altered landscape. Therefore, many scientists turned to island biogeography theory in search for predictive models of the implications of fragmentation for species diversity (Whittaker 1998). For habitat patches in fragmented landscapes, island biogeography theory would predict lower species diversity in small patches than in the large ones. Similarly, patches farther away from the contiguous forest will have lower diversity in comparison with the nearer ones. In addition to the extinction and immigration of species proposed by the island biogeography theory, two other effects are relevant to terrestrial fragmented landscapes:

Species relaxation: Islands connected by land-bridges that have subsequently been submerged due to rising sea-levels in the geological past face losses of species as a consequence of the loss of the land connection (Terborgh 1974). Similarly, species will be lost in the process of progressive landscape fragmentation over time due to disconnection of patches.

Rescue effect: Islands close to an immigration source can be expected to have lower extinction rates than predicted by the island biogeography theory, because declining populations can be bolstered by immigrants of the same species (Brown & Kondric-Brown 1977). Similarly, patches within the dispersal distance for most organisms inhabiting the landscape will have higher diversity than those beyond it.

The island biogeography theory became a popular management tool due to the analogy of habitat patches as islands, and based on the theory, rules for nature reserve design were proposed (Willis 1974; Diamond 1975; Shafer 1990). This initiated a debate over the SLOSS (single large or several small) problem and examination of the empirical evidence, if a single large reserve is better for long term maintenance of biodiversity than several small ones. Simultaneously, the island biogeography theory received a great deal of criticism from the scientific community (Simberloff & Abele 1976; Margules *et al.* 1982; Simberloff 1991) and generated further debate on its usefulness.

The following are some important shortcomings of the theory:

- There are very few empirical tests of the theory since they require datasets for many islands, species and generations (Hoopes & Harrison 1998). Whatever empirical research for terrestrial landscapes there is, claiming support to the theory, it has failed to consider hypotheses for situations other than mainland-island systems (Meffe & Carroll 1997) – the theory assumes one-way immigration from mainland to islands, whereas in reality, there may be multiple sources of immigration as far as the patches in fragmented landscapes are concerned.

- The probability of all species migrating to or becoming extinct from an island was considered equal in the theory, thus making the theory neutral at the level of species. Whittaker (1998) and Hubbell (2001) argue that the theory does not recognise the differential dispersal abilities of species.

- Many species have no problem crossing the gaps between two patches of similar habitat in a terrestrial landscape, as opposed to oceanic islands. Thus, the theory does not recognise species' ability to move through a heterogeneous habitat (Forman 1995).

There is now unanimous agreement among ecologists that diversity in habitat patches cannot be explained by the two simple factors of area and isolation (Hoopes & Harrison 1998). However, the theory may work for certain highly insularised landscapes and species, which are unable to use the intervening landscape matrix. The types of landscapes and species, which the theory may apply to, will be discussed later in the chapter (*pp. 14-15*).

Metapopulation theory

Metapopulation theory examines the dynamics of sets of semi-independent populations connected by dispersal (Hanski & Gilpin 1991). Levins (1970, cited in Hanski 1997) was the pioneer of the metapopulation theory. In his model, a metapopulation is a network of extinction-prone subpopulations of a species occupying a variety of habitat patches. These subpopulations inhabit identical patches and are subject to equal but independent probabilities of extinction and recolonisation. The difference between the theory of island biogeography and the metapopulation theory is that the former predicts the number of species as a function of area and isolation while the latter predicts the population dynamics of a single species inhabiting isolated patches (Whittaker 1998).

Conservation scientists have suggested the creation of metapopulations for endangered species as a means of maintaining their populations across habitat patches in increasingly fragmented landscapes. However, a spatial model created by Lamberson *et al.* (1992) in order to predict how the populations of the Northern Spotted Owl (*Strix occidentalis caurina*) will survive in patches surrounded by logged forest, eventually failed to predict realistic minimum viable populations of the bird (Harrison *et al.* 1993). As a result of un-acknowledged risks to the owl, the populations of the bird declined in a pattern not predicted by the metapopulation models. On the other hand, some butterfly species have been found to behave according to the predictions of metapopulation models (Thomas & Hanski 1997). Therefore, the more basic question is: how broadly and to which species does the metapopulation theory apply in habitat fragments?

Four scenarios of landscape structure are common in fragmented landscapes (*Figure 1.1*):

1. Where patches are roughly of equal size and dispersal distances are comparable to the distances between patches (Classic or Levins' metapopulations);
2. Where patches are so unequal in size and / or habitat quality, that most immigrations are in one direction – from large to small patches; extinctions and recolonisations that occur in very small populations are inconsequential (Mainland-island or source-sink metapopulations);

3. Where patches are so close together relative to dispersal distances that they support a single population and not a metapopulation (Patchy populations);
4. Where patches are so far away relative to dispersal distances that the populations are not interconnected and the assembly ceases to be a metapopulation (Non-equilibrium metapopulations) (Harrison & Taylor 1997; Hoopes & Harrison 1998).

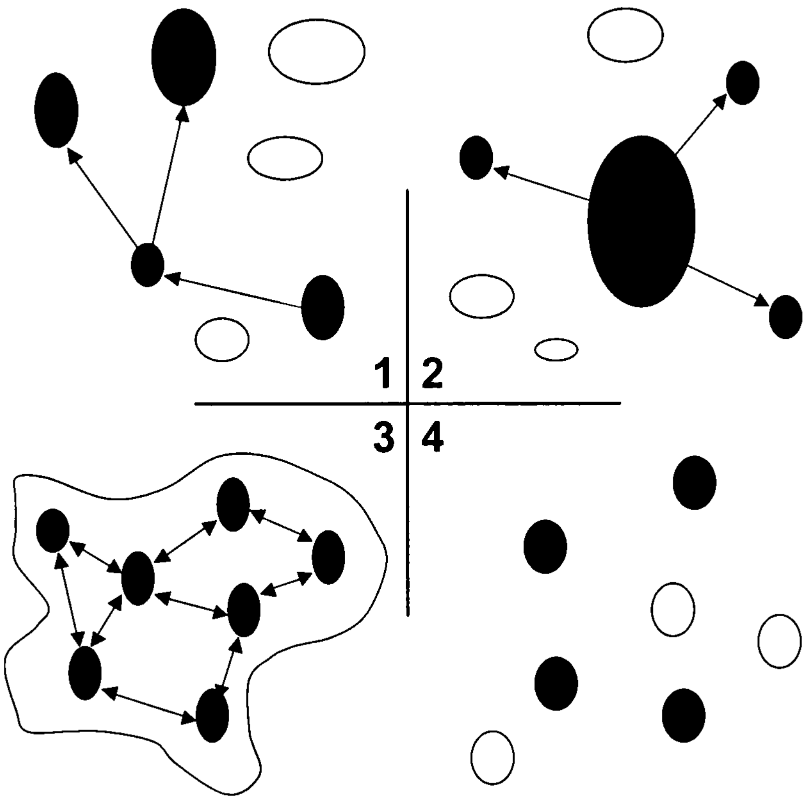

Figure 1.1 Structures of metapopulations that can arise from fragmentation (Adapted from Hoopes and Harrison, 1998)

Thus, whereas patches of tall forest in a savanna landscape may be treated within the framework of metapopulation theory, the concept may not be suitable for forest patches in a highly tree-covered landscape. Furthermore, an insurmountable barrier for one group of organisms may be easily navigated by another – for sunbirds, forest patches spread over a hundred square kilometre landscape may sustain a metapopulation, while for dispersal-limited snails, they can at best sustain isolated populations. Metapopulation models, therefore, are not generally applicable to all organisms in fragmented systems (Fahrig & Paloheimo 1988). Hoopes & Harrison (1998) caution against the general use of such models in conservation decision-making.

The following are some important shortcomings of the metapopulation approach:

- Several authors have noted that metapopulation models are extremely data-demanding and usually require data that are very difficult to obtain (Kindvall &

Ahlen 1992; Doak & Mills 1994). Moreover, model results tend to be very sensitive to poorly estimated parameters and the predictions of such models are dangerously inaccurate for the species in question (*e.g.* Harrison *et al.* 1993; Wilson *et al.* 1994).

- Most empirical examples of metapopulations pertain to single species or a group of interacting species (Hanski & Gilpin 1997) but not to multispecies ecological communities (Whittaker 1998; Hubbell 2001).
- Most metapopulation models assume no distance effects (Dawson 1994, cited in Whittaker 1998; Fahrig & Merriam 1994), although in practice dispersal abilities vary from species to species. For instance, metapopulations of frogs may be influenced by the availability of suitable habitat in the surrounding 500 m, whereas for birds this distance may be 3 km, because of large differences in mobility resulting in the different abilities of frogs and birds to disperse. The issue of 'scale' has therefore been considered important in studying fragmented landscapes (Gardner *et al.* 1989; Doak *et al.* 1992): an issue, which metapopulation models do not address.
- It is important for conservationists to recognise that many local populations may not be at equilibrium and regional processes may be critical in sustaining metapopulations (Hanski 1996, Brown 2000).

In conclusion, metapopulation theory offers a useful framework for thought, but if the concept is to be useful as a theoretical framework for conservation decision-making, it must be extended from the original simplistic models to allow for the differing degrees of population connectivity in fragmented landscapes and differing forms of inter-patch relationships, as in real-world systems (Harrison 1994; Hanski 1996) (*Figure 1.2*).

Landscape ecology

Although landscape ecology lacks a sound theoretical framework (Wiens *et al.* 1993; Wiens 1997), its principles are the closest to the real world of fragmented landscapes amongst the three approaches. The principal tenet behind the landscape approach is that a landscape is made up of several elements of different types (Forman & Gordon 1986). Landscape ecology, unlike either island biogeography theory or metapopulation theory, recognises that the attributes of the matrix that separate habitat patches play an important role in determining the characteristics (such as species richness) of the patches themselves (Forman 1995). From the landscape ecological perspective, the interplay between patch size, isolation from other similar patches or a contiguous area of the same habitat, and characteristics of the intervening matrix, is an important determinant of diversity (*e.g.* Turner 1989; Opdam *et al.* 1993; Gustafson 1998). The utility of these three parameters (patch size, isolation and nature of the intervening matrix) for landscapes and groups of organisms with different characteristics is examined in the rest of this section. The shortcomings of the landscape ecological approach are identified, followed by a discussion on the best framework or combination for the present study.

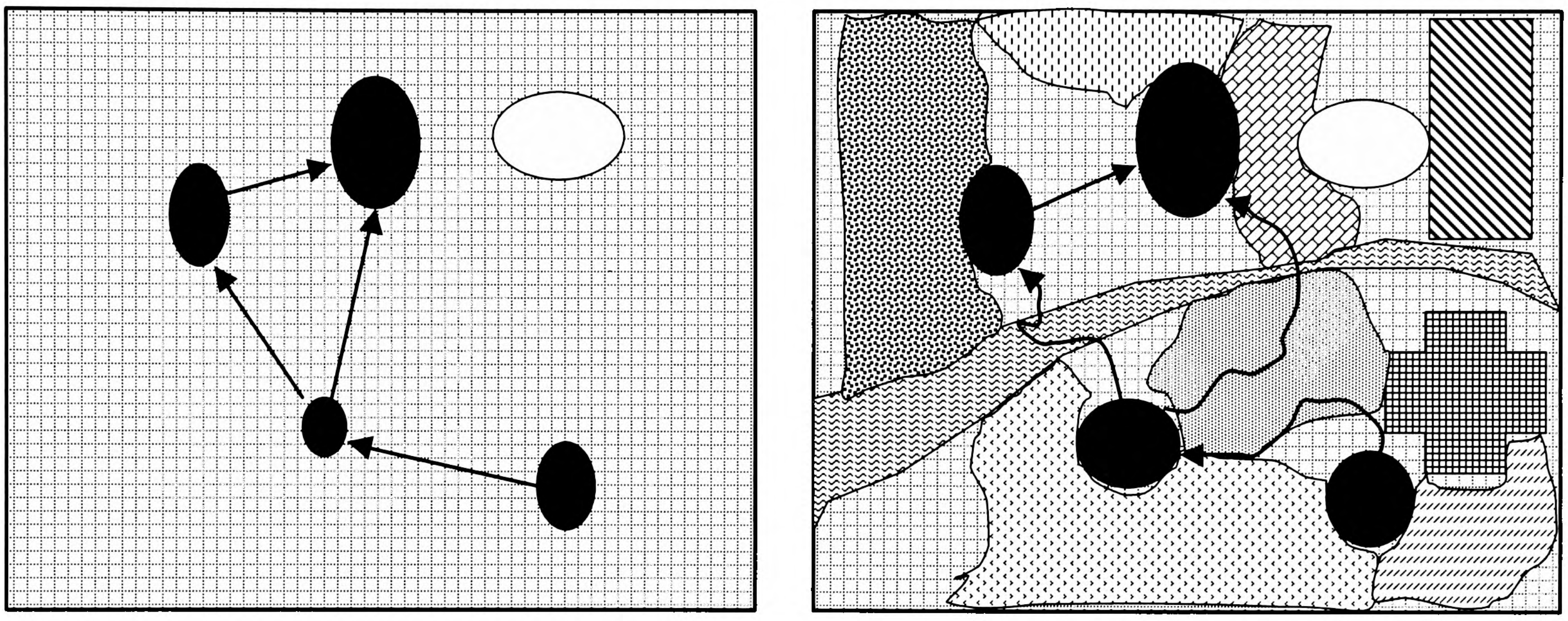

Figure 1.2 Metapopulations in theory (left panel) are represented in a simplistic way, however, in practice (right panel) they are much more complex because of the surrounding landscape mosaic (Adapted from Wiens, 1997)

Patch size

Patch-size is one of the simplest and the most intuitive parameters that can be measured in a fragmented landscape. The idea that patch-size is related to diversity is not new. The island biogeography approach also considers patch size as important. The relationship between number of species and area is summarised by the equation – $S = c A^z$, where S is the number of species occupying an area A, c and z are constants (Arrehnius 1921; Preston 1960). The values of c and z are different in different biogeographic zones (Rosenzweig 1995).

Mean patch size is often used to provide a summary description of a particular patch-type within a landscape (*e.g.* Gustafson & Parker 1992). However, averages might be misleading due to a possibility of a skewed distribution. For instance, the majority of patches in the landscape may be one hectare in size along with a single contiguous patch of 300 ha. Thus, average patch size would not reflect the true landscape pattern (Betts 2000). Instead, by examining the distribution of patch-sizes, it may be possible to determine the percentage of patches that satisfy the criteria of habitat requirement of the species in question. For instance, distribution of habitat-sensitive birds such as the Mountain Imperial Pigeon (*Ducula badia*) may depend on how many patches more than 100 ha in size are available in the landscape rather than average patch size (personal observation).

While patch-size may be ecologically relevant and relatively easy to measure, it may be problematic in landscapes where patches are not easily identifiable. Gustafson (1998) noted that many ecological and environmental conditions are characterised by gradients rather than discrete boundaries. This may be particularly true for modified (but not fragmented) landscapes. Secondly, what constitutes a patch may be purely arbitrary from the perspective of a particular species, depending on its dispersal ability. Thus, patch-size may not always be an important parameter for assessing diversity. Similarly, perimeter: area ratio (Krummel *et al.* 1987) and patch shape (Forman 1995), considered important in determining diversity, fail to be meaningful in landscapes where patches cannot be clearly defined.

Isolation

Isolation measures the degree to which patches are separated from or connected with other similar patches or a larger habitat area. Isolation of patches is also considered important in the island biogeography approach. In landscape ecological studies, nearest neighbour distance is the most commonly measured parameter (McGarigal & Marks 1995; Hargis *et al.* 1998, cited in Betts 2000). Mean nearest neighbour distance summarises the distance between each patch of a certain type and other similar patches. This parameter can be used along with information on movement capabilities of various species to determine, whether the average distance between patches is close enough for organisms to disperse (Betts 2000). However, this parameter, like patch-size may be problematic in landscapes where patches themselves are not easily identifiable.

Baskent & Jordan (1995, cited in Betts 2000) proposed isolation of patches from large forest as an effective measure for assessing diversity because using an isolation measure, one can compare similarity between a patch and large forest along a distance gradient. The inverse relationship between distance and similarity is well known (Nekola & White 1999). Thus, isolation from large forest can provide a scale for comparison of similarity between an uninterrupted forest and forest patches in modified landscapes.

Surrounding landscape matrix

The surrounding landscape matrix is the most complex parameter for measurement because of characteristics of landscape mosaics that are often composed of a number of different patch types. The island biogeography and the metapopulation approaches do not recognise the surrounding landscape matrix as important. Landscape ecologists have explored several ways of measuring landscape structure – its composition and configuration. Measures such as the Contagion index[1] (Li & Reynolds 1993) are reported to be useful. However, the problem with landscape indices is that they combine all landscape parameters into one index, which might give a misleading picture (Davidson 1998). For instance, Hussey *et al.* (1989, cited in Davidson 1998) evaluated the effects of timber sales on fragmentation of old growth forests. In order to quantify fragmentation they calculated an index based on total area of old-growth interior, total area of old-growth edge and total area of edge without interior. However, the index was unable to address important aspects of fragmentation such as isolation from or connectivity with other similar patches because these parameters operate at scales larger than the scale of patches. Thus, measurement and evaluation of different aspects of the landscape structure, rather than combining them in one index, are necessary.

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[1] Contagion index expresses the degree to which habitat fragments are clustered or dispersed. It is the sum, for each patch-type, of the product of two probabilities, first, that a randomly chosen cell on a grid belongs to a certain patch type and second, that the neighbouring cell belongs to a different patch type (McGarigal & Marks 1995).

The following are the important parameters that the published literature in landscape ecology identifies:

- Landscape continuity or the proportion of the total habitat area (*e.g.* Andrén 1994; Fahrig 1998) that is available for the organism or the group of organisms in question;
- Landscape heterogeneity or number of patch types (*e.g.* Miller *et al.* 1997);
- Landscape complexity or fractal geometry – *e.g.* Nikora *et al.* (1999, cited in Betts 2000) used area: length ratios of patches and patch edges respectively.

Despite the proximity between the landscape structure recognised by landscape ecology and real-world fragmented landscapes, the framework has the following shortcomings:

- There is no 'sound' theory, unlike in island biogeography or the metapopulation theory, behind the interactions of organisms with the landscape structure (Wiens 1997). Little research has been carried out to establish the relationship between the landscape ecological indices and now widely used metapopulation models (Brown 2000).
- Most empirical examples in landscape ecological research are from European and North American landscapes that are fragmented so as to produce regular and simple geometrical shapes (Milne 1988). However, fragmentation in tropical landscapes often results in more irregular and complex shapes without clear boundaries, which landscape ecological models may not be able to address effectively because such models often rely on straightforward mathematical measures of Euclidean[2] geometrical figures (Milne 1988).
- Landscape ecological research requires extensive spatial information in digital forms, collected using sophisticated technology such as remote sensing. Such information is still very expensive and often difficult to obtain for most tropical regions (Longhorn 2001 *pers. comm.*).

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[2] Relating to the geometry, the study of angles and shapes, formed by the relationships between lines described by the ancient Greek thinker Euclid.

Suitability of heuristic frameworks

It is important to examine the three competing approaches discussed in previous sections, from the viewpoint of organisms (Pearson *et al.* 1996). The ability of organisms to disperse (mobility) as well as to use the intervening matrix is important in order to determine the relevance of each of the three heuristic frameworks, for the persistence of the organisms' populations and the overall diversity, in fragmented and modified landscapes. The following examples illustrate this argument:

- The size of a favourable habitat patch is important for explaining bird diversity in an agricultural landscape in Western Australia (Miller & Cale 2000), where 93% of the landscape is without tree cover. In this landscape, island biogeography will usefully explain the diversity of forest-dwelling birds that are unable to use the intervening agricultural matrix.
- In the case of butterflies, many individuals remain within the local population where they first emerge (which means less possibility of error in measuring population sizes). In addition, butterflies have short generation times, which makes modelling of their population dynamics easier. Therefore the metapopulation theory provides a suitable framework for studying their extinction-recolonisation dynamics (Thomas & Hanski 1997).
- The study of diversity and distribution of wind-dispersed, deciduous and habitat-generalist trees that can survive effectively in a range of habitats across a landscape, requires a landscape ecological approach (*e.g.* Borges 2000); island biogeography or metapopulation theory are not sufficient to describe or explain their patterns of distribution or abundance.

In addition to the group of organisms under investigation, the type of landscape will also dictate the suitability of the three heuristic frameworks. For instance, island biogeography may well provide a suitable framework for studying diversity of shade-tolerant trees inhabiting forest patches in a savanna landscape. Metapopulation theory may be useful to test abundances of individual species in a set of forest patches, connected by dispersal, in the same landscape. However, for a highly tree-covered landscape, where a mosaic of many patch types occurs, landscape ecology may be the best choice for investigating the effects of landscape modification on biodiversity.

Finally, the three heuristic frameworks are not mutually exclusive. In fact, there are close intellectual links among researchers who pioneered these frameworks. As Hanski (1997) points out, the two contemporary theoretical ecologists – McArthur (one of the proponents of the island biogeography theory) and Levins (the originator of metapopulation idea) were in close contact with each other. While MacArthur and Wilson completed their monograph on the Equilibrium Theory of Island Biogeography in 1967 (MacArthur & Wilson 1967), Levins' metapopulation ideas were first published in 1969 (Levins 1969). The two theories are similar to the extent that Levins' model is a single-species version of MacArthur and Wilson's theory. According to Forman (1995), landscape ecology integrates much of the previous theoretical understanding and empirical research in ecology and biogeography into a single approach. Forman, the pioneer landscape ecologist, and a Harvard professor, worked in close collaboration with Wilson, another senior Harvard professor and one of the proponents of the island biogeography theory. This is strong evidence for the 'sympatric evolution' of the three approaches and suggests that they cannot be looked at in isolation. Research into landscape fragmentation and modification will therefore require an adaptive strategy for choosing a suitable approach or combination.

Empirical research

The findings of empirical research on biodiversity in fragmented landscapes are reviewed in this section. There is a large body of literature on this subject. The theory of island biogeography alone has been discussed and debated in close to a thousand research papers (Hubbell 2001). Metapopulation theory and landscape ecology have been equally well researched, if not more. It appears from the literature that the effects of landscape fragmentation on diversity are studied at three different levels:

1. Genes
2. Species
3. Communities

The effects at the three levels are explored in the rest of this section. The focus remains on the effects that pertain to patch size, isolation of the patch from large habitat (hereafter 'isolation'), and composition and configuration of the surrounding landscape matrix – the three most important factors identified in previous sections.

Effects of fragmentation on genes

In fragmented landscapes pollination, reproduction and dispersal of plants are often affected across patches (Kanowski & Boshier 1997). For animals, individuals within a patch reproduce more among themselves rather than with the individuals from other patches. This preferential combination of genes has important genetic consequences. Genetic variation is a measure of how different individuals are in their genetic make-up. High variation is considered to enhance the ability of a population to adapt to changing environmental conditions while low genetic variation usually has negative consequences for the population in question. Thus, the preferential genetic combination among a limited number of individuals within a patch results in lower genetic fitness than a similar population with more individuals inhabiting an uninterrupted habitat. The patchy populations may encounter various difficulties.

Inbreeding depression is the effect of mating among close relatives, which produces fewer, weaker and more sterile offspring than average (Charlesworth & Charlesworth 1987). *Genetic drift* takes place in small populations when the gene frequencies change by chance, and may play a role in decreasing genetic variation over time (Lacy 1987). A *founder effect* occurs when the first few individuals of a species that colonise a patch in the fragmented landscape constitute a small population, normally with low genetic variation (*e.g.* Baker & Moeed 1987), leading to a combined effect of inbreeding depression and genetic drift. *Population bottlenecks* occur when a population is greatly reduced in size as a result of landscape fragmentation, the genetic diversity is lost and the population may decline as a result of a lack of overall genetic fitness (*e.g.* Temple 1986).

Although all the effects mentioned above merit consideration while investigating the long-term survival of diverse communities in fragmented landscapes, diversity below species level is beyond the scope of this work. The consequences of fragmentation for diversity at and above species level are examined in the next two sections.

Effects of fragmentation on species

Fragmentation can affect the abundance and population structure of individual species because reduction in habitat size causes reduction in population size. Biologically rare species that typically have small population sizes in fragments are vulnerable to local extinction (Bierregaard & Stouffer 1997; Brown & Hutchings 1997). Similarly, many species with restricted or patchy distributions (Diamond 1980) often disappear from fragmented landscapes due to a 'sampling effect' (Wilcox 1980), which means that due to their biological rarity, they may be absent from a patch only by chance. The patch-scale effects change the quality of the habitat, which further reduces population sizes of many species. The ecological and habitat specialist species, which often avoid or rarely use modified habitats surrounding fragments or are intolerant to habitat changes inside fragments, are vulnerable to elimination from fragments (*e.g.* Didham 1997; Laurance 1997a; Warburton 1997).

Immigration and dispersal are very important in maintaining local populations of many plants (Hubbell & Foster 1986). Trees with heavy seeds are the first to disappear from fragments because their large-mammal dispersers, who have large habitat requirements, cannot survive in fragments (Laurance 1991a). For instance, Harrington *et al.* (1997) found that regeneration of large-seeded trees in Australian rain forests was affected by forest fragmentation, principally because of their strong dependence on animal vectors (who themselves were affected by fragmentation) for seed dispersal. Many species with co-evolved interdependencies (*e.g.* tree species with specialist pollinators) are vulnerable to higher order effects of fragmentation (Gilbert 1980). For instance, *Heterophragma quadriloculare*, a tree that is pollinated by highly mobile carpenter bees occurs in the crest forests in fragmented seasonal cloud forests of the Western Ghats of India. Borges (2000) cites studies from this region that reported no effect of tree isolation on the reproductive success of *Heterophragma qaadriloculare*. On the other hand, *Lasiosiphon eriocephalus,* a shrub in the same environment, is pollinated by sedentary beetles and its isolated individuals were reported to have lower reproductive success than individuals occurring in clumps (Borges 2000).

For birds, large body size and low fecundity are reported to be the main hurdles for survival in fragmented landscapes. Large-bodied birds occur at low population densities; large body sizes are often correlated with low fecundity (*e.g.* Pimm *et al.* 1988), thereby increasing the risk of predation. Thiollay (1994) and Renjifo (1999) report the vulnerability of large-bodied birds in fragments. Sieving & Karr (1997) report higher nest-predation in fragments than in large forests and therefore higher possibility of less fecund birds disappearing from fragments. In addition to the natural causes of bird extinctions, fragmented landscapes also become accessible to hunters (Peres & Terborgh 1995) – large-bodied birds are more vulnerable to hunting than the small-bodied ones.

The effects of the three important landscape factors – patch size, isolation and the characteristics of surrounding landscape matrix – on the types of species known to be vulnerable in fragmented landscapes are summarised below.

Patch size

- For species that have patchy or restricted distributions (Diamond 1980), larger patches will mean higher probability of occurrence than the smaller ones.
- For biologically rare species that have small population sizes (Bierregaard & Stouffer 1997), large patches are better for maintaining minimum viable populations than small ones.
- For less fecund birds, that are vulnerable to nest predation at patch edges (Sieving & Karr 1997), large patches with extensive interiors are beneficial, while small ones have the opposite effect.
- For species with large territorial requirements, patch size is important. Distribution of such species in fragmented landscapes will be constrained by small patch size (Laurance *et al.* 1997).

Isolation

- For plants pollinated by, for instance sedentary beetles (Borges 2000), greater isolation means reduced reproductive success.

- For plants having heavy seeds requiring large-mammal dispersers (Laurance 1991a) isolation will limit the spread of propagules.

Surrounding landscape matrix

- For ecologically sensitive or habitat specialist species (*e.g.* Bierregaard & Stouffer 1997) that avoid modified landscapes, a greater degree of modification will mean restriction on movement and eventual disappearance from fragments.
- On the contrary, for habitat generalist species, fragmentation will make available more suitable habitat, causing increase in their population sizes (*e.g.* Laurance 1997a).

Effects of fragmentation on communities

Fragmentation has effects on the structure of ecological communities by changing:

1. Composition of ecological species groups (*e.g.* Warburton 1997)
2. Community diversity (*e.g.* Bierregaard *et al.* 1992)
3. Community similarity (*e.g.* Milne & Forman 1986)

The empirical examples of such effects are examined in this section and are related to the suitability of the three competing heuristic frameworks for studying modified landscapes. It is argued that landscape ecology offers a powerful tool for studying the effects of landscape modification on the above three community attributes.

Composition of ecological species groups

Fragmentation typically affects those species, that avoid or rarely use matrix habitats surrounding fragments and are intolerant to habitat changes inside fragments (*e.g.* Bierregaard & Stouffer 1997; Warburton 1997). In the case of trees, first, increased disturbance causes increased density of light demanding or pioneer species, which displace the late successional or the shade tolerant ones. Second, disturbance makes fragments open to invasion by non-forest species or exotics while undisturbed habitats are rarely invaded (Brown & Hutchings 1997; Laurance 1997a). A combination of the two effects leads to changes in species composition as well as proportional abundances of plants in the community.

Similar effects are observed on the avifauna in fragmented landscapes – large frugivorous birds are reported to disappear from fragments while most small nectarivores survive fragmentation (Feinsinger 1976; Greenberg 1981; Johns 1991; Thiollay 1994; Stouffer & Bierregaard 1995; Restrepo *et al.* 1997; Sieving & Karr 1997; Warburton 1997; Renjifo 1999). This is often attributed to the ability of nectarivores to use a range of resources along the edges of forest patches as well as in the intervening landscape. On the other hand, large frugivores such as hornbills have special dietary requirements. Usually, birds such as small nectarivores increase in proportion to the large frugivores, in effect, altering community structure.

From the examples of plants and birds, it appears that the quality of the edge habitat and the intervening matrix are important in determining the effects of fragmentation on the composition of ecological species groups, particularly when the patches are very small. The same is true for other groups of organisms such as insects (Didham 1997) and amphibians (Tocher *et al.* 1997). Therefore, landscape ecology, with its spatial approach and consideration of the landscape surrounding the patches, may offer a better scheme for assessing such effects than island biogeography or metapopulation theory.

Community diversity

As seen in previous sections, fragmentation results in shrinking of a habitat, reducing the population sizes of many species. In this process, rare species are lost because of the so-called 'sampling effect' (Wilcox 1980) – more abundant species are disproportionately represented in a sample while rare species are absent by chance. Generally, biological diversity of patches in a fragmented landscape is lower compared with a continuous stretch of the same habitat. This may occur due to the loss of species in the process of fragmentation (Terborgh 1974). In addition to the loss of rare species, habitat specialists disappear from fragments and generalists, or in the case of plants weedy invasive species, take over. If the latter outnumber the former, the result will be a higher diversity at the patch scale contrary to the predicted 'species relaxation' (Terborgh 1974). Thus, high disturbance will result in a community with a large proportion of early successional or weedy species; in contrast, low levels of disturbance will maintain the dominance of habitat specialist or most competitive species. The intermediate levels of disturbance will allow both types of species to co-exit and are therefore likely to maximise ecological diversity. Such increase in diversity is explained by the Intermediate Disturbance Hypothesis (Connell 1978) – community diversity is found to increase with disturbance until the diversity reaches a threshold, after which it starts declining. In other words, the disturbance-diversity curve is unimodal or hump-shaped.

Patch size is thought to affect species diversity. However, empirical studies have conflicting conclusions about the relationship between size and diversity. There is evidence for and against the theoretical predictions. For instance, Levenson (1981) found a strong positive relationship between patch size and woody-plant diversity in Wisconsin woodlots; Hill & Curran (2001) found a similar relationship in forest fragments in Ghana. Miller & Cale (2000) and Opdam *et al.* (1985) found patch area a useful variable in explaining bird diversity in the Western Australian and the Dutch woodlands respectively, while Nilsson (1986) and Honnay *et al.* (1999) found patch area a redundant variable for bird diversity on the Swedish lakes and in the Belgian woodlands respectively. The studies by Levenson, Hill & Curran, Miller & Cale and Opdam *et al.* took place in a predominantly agricultural landscape, where the intervening matrix was unusable by tree and bird species, and the numbers of species within patches were proportional to the amount of suitable habitat available inside them. In the study on the Swedish lakes by Nilsson and in the Belgian woodlands by Honnay *et al.*, the hospitability of the surrounding matrix in terms of nesting and roosting sites for birds may be more important than the size of the water-body or the area of the patch respectively. Thus, while island biogeography is useful in explaining diversity of birds in forest patches surrounded by agricultural fields, landscape ecology may be more suitable for studying diversity in highly tree-covered landscapes.

Community similarity

As seen from the previous sections, fragmentation restricts movement of species across modified landscapes. Thus, the greater the isolation of a habitat patch, the fewer the number of individuals that may be able to reach that patch from, for example, a continuous source habitat, provided there is no immigration or dispersal from neighbouring patches. Research has shown that isolation causes a decline in community diversity. Milne & Forman (1986) and van Dorp & Opdam (1987) observed this phenomenon for woody plants in Maine, USA and for forest birds in Dutch woodlands respectively.

Although Brown & Kondric-Brown (1977) describe a 'rescue effect' on terrestrial islands, where species diversity in patches close to a source is bolstered by greater immigration than in more distant patches, loss of species in fragments is more a rule than an exception. Such losses may cause reduced inter-specific competition for resources allowing the persisting species to achieve unusually high densities (MacArthur *et al.* 1972). This has been attributed to 'frustrated dispersal' (Terborgh 1997) for some species, resulting in a large number of individuals of the same species inhabiting the patch. As a consequence of this process, the isolated patches have some species in disproportionately high numbers dominating the species-abundance distribution. Luiselli & Capizzi (1997) observed that isolation of forest patches is important for explaining the distribution and relative abundances of some Mediterranean forest snakes that are, presumably, unable to use the intervening non-forest landscape. These species become disproportionately abundant in isolated fragments, thus changing the community structure and making isolated patches increasingly dissimilar.

While the island biogeography theory would predict lower species diversity in isolated patches, it cannot provide an explanation for the external effects on species diversity. Metapopulation theory also fails to recognise increases in population abundance of certain species, which are adapted to disturbance, in isolated patches. Landscape ecology, with its spatial approach, and recognition of the characteristics of the landscape between patches, is more suitable approach for explaining the processes related to isolation and community similarity in fragmented landscapes.

1.2 Approaches to biodiversity conservation

There are two ways in which biodiversity conservation can be achieved:

1. *Fortress conservation*: strict protection of natural landscapes and conservation of biodiversity within their boundaries through bans on any human activities;
2. *Community-based conservation*: conservation with the involvement of local communities by allowing sustainable human use and local management of the land.

The origins of the two conservation paradigms are explored in the following sections and their strengths and weaknesses identified.

1.2.1 Fortress conservation

Human activity is seen as a major threat to the world's biodiversity (Wilson 1992). The conventional approach to conservation has been through the creation of protected areas that exclude people as residents, prevent consumptive uses and minimise human impacts. This type of conservation often requires an initiative from local or national governments (Primack 1998). Such approaches to conservation are termed top-down approaches (Gadgil & Guha 1995) where a governing authority imposes conservation measures in the areas that they might feel appropriate and where their efforts have to be backed up by formal rules and regulations on human activities inside protected areas. In addition, the international community may play a role in deciding how and where such strictly protected areas should be established. The World Conservation Union (IUCN 1994) has developed a system of classifying protected areas into six types depending on their protection status (*APPENDIX 1*). Such system is believed to offer guidelines for organised protected area management efforts internationally (IUCN 1994).

The earliest documented evidence of state-initiated protected areas dates back to 252 BC when Ashoka, the emperor of the Indian subcontinent, passed an edict for the protection of animals, fish and forests by declaring certain areas as protected (MacKinnon & MacKinnon 1986). The modern conservation movement began with the establishment of national parks in the United States (Kramer & van Schaik 1997). The first one, Yellowstone National Park was established in 1872 (Hays 1987). Currently, the world's 30,000 protected areas cover over 13.2 million km^2 (nearly 10%) of the land surface (WCPA 2001). However, the coverage varies dramatically between countries – Austria and Germany have nearly 25% of their land under strict protection while Greece and Turkey have only about 0.3% (Primack 1998). Protected areas offer an effective apparatus for maintaining contiguous habitats that large mammals require (*e.g.* Kerr 1997). In addition, they provide environmental services across national boundaries – "Protected areas bring water to cities, protect downstream communities from flood, shelter the homes and sustain the livelihoods of indigenous groups, safeguard biodiversity for humanity as a whole, and provide contact with nature for an ever-more urbanised society" (WCPA 2001). However, the approach of maintaining large conservation areas faces various challenges, which are described below.

Although the momentum for establishing protected areas increased throughout the twentieth century, it dropped in the 1980s, reflecting the lack of political will on the part of governments to designate more protected areas. Primack (1998) speculates that protected areas will never cover a large percentage of earth's surface due to the perceived needs of human society for natural resources. Kenya is one country that has achieved the IUCN target of 10% (IUCN 1994) by protecting about 60,000 square kilometres of its land area. However, a recent study calculated that had this land not been protected, it could support 4.2 million people through agricultural and livestock production with a net return of $ 203 million, or 2-8% of the country's GDP (Norton-Griffiths & Southey 1995, cited in Inamdar *et al.* 1999). Therefore, there are economic limitations in developing countries as to how much of the total land surface they can cover, especially where there is growing economic pressure on land. Furthermore, the management of protected areas is very expensive because it requires a large infrastructure and funds; and compensation for lost opportunities for economic use, which poor nations cannot afford.

The 'fences and fines' approach (Wells *et al.* 1992) to biodiversity conservation (also metaphorically referred to as: 'bullets and barbed-wire' or 'guards and guns' approach) has been widely used in developing countries but has led to conflicts with the economic interests and rights of the local people. In many developing countries protected areas tend to be unpopular, especially with politically powerful interest groups keen on commercial exploitation of natural resources (Durning 1993, cited in MacKinnon 1997). Indonesia is a classic example of the misuse of political power for the usurpation of natural resources (Jepson *et al.* 2001). In addition, strict protection is also resented by many local communities that depend on natural resources within strictly protected areas for their livelihoods. In some cases, assertion of local rights over traditional lands has involved violent confrontations with government authorities seeking to acquire the land (Gadgil & Guha 1992). Human-wildlife conflict is another aspect of this resentment as seen every year, when not only several cattle are killed by leopards but also people are killed by elephants in the southern Indian jungles (personal observation). There are three main reasons for the strong resentment. First, the developing nations in the tropics have nationalised some 80 percent of forest land often without regard to the traditional arrangements that preceded their authority or that of the previous colonial masters (Panayotou & Ashton 1992). Second, local communities do not always perceive the government as representing their interests and therefore the people are unwilling to accept the government authority. Third, the local people incur opportunity costs from foregone development activities (Brown 2000).

Many protected areas are located on land considered to be of little economic value. Madagascar provides an example where protected areas are frequently established in inaccessible or low-productivity areas that are unsuitable for agriculture (Smith 1997). Ramesh *et al.* (1997) argues that protected areas in India are declared on an *ad hoc* basis without consideration of biological or ecological criteria for biodiversity conservation. The experience in Indonesian New Guinea suggests that the locations of protected areas are not determined by biological principles (Diamond 1986). Many such efforts, for instance Project Tiger in India (Panwar 1982), are focused on one or a few flagship species that have a great public appeal, while ignoring other elements of biodiversity (Kerr 1997). Schonewald-Cox & Bayless (1986) caution that the administrative boundary of a protected area is different from the natural edge of a particular ecosystem under protection. This is because protected areas are often designated by political or legal, rather than ecological considerations. As a result many species, communities or ecosystems are only partially protected within the area of their extent. Outside this range they are often vulnerable to land-use pressures.

1.2.2 Community-based conservation

This type of conservation originates at the grassroots level. In many societies protection of forest patches for religious beliefs is common. Historically, local people have often established local systems of rights to natural resources (Dasmann 1991), which, in the contemporary world, are sometimes recognised by their governments. Historical records of sacred forests in South-East Asia, Australia, the Americas and Europe can be found in the literature (Bridgewater 1992; Hughes & Chandran 1998). This practice of setting aside no-use areas for protection of watershed and other environmental benefits is believed to have its origins in hunter-gatherer societies (Kosambi 1962; Gadgil & Vartak 1976). This approach to conservation is called the bottom-up approach (Gadgil & Guha 1995), where the local communities are able to decide the fate of biological diversity. In the modern world, this practice is called community-based natural resource management (Getz *et al.* 1999). According to FAO (2001) the rationale for the community-based conservation is, "The participation of local communities in resource planning and management can both improve the effectiveness of conservation efforts and help ensure that local communities benefit from conservation". In the Indian state of West Bengal, the local communities committed themselves to protecting degraded forests from fires, grazing, illegal cutting and agricultural encroachments while Government forest officers agreed to share the financial benefits from forest products with village members. This arrangement called Joint Forest Management was received with enthusiasm in many other Indian states and elsewhere in the world (Poffenberger 1996; Khare *et al.* 2000). Thus, community-based conservation encourages local people to take the initiative in protecting biodiversity while the State apparatus recognises their authority to a certain extent. Such an approach is economically sustainable because it does not involve the costs of maintaining a large infrastructure such as required by protected areas. However, Prins (1992) is one of the sceptics, who are of the opinion that conservation has little to do with development of local communities and it will be an unprecedented act to allow local people to invade protected areas. The community-based conservation approach has the following limitations:

It may not be effective in protecting large, contiguous areas of natural habitat required by some species of conservation importance, such as tigers, that have large home ranges and require a wide range of habitats within contiguous areas (Kerr 1997).

Although the aspirations of conservation, sustainable use and benefit sharing inscribed in this approach are laudable, it has been difficult to demonstrate any tangible success of community-based conservation approach on the ground (Brandon 1997; Inamdar *et al.* 1999). A recent review published by the World Bank concerning Indonesia concluded that "few integrated conservation and development projects can realistically claim that biodiversity conservation has been, or is likely to be, significantly enhanced as a result of current or planned activities" (World Bank 1997).

Community-based conservation involves several actors unlike the government-imposed strict protection. In some cases a community's cultural values and practices can provide a solid basis for an effective conservation programme while in some others it will be doubtful if communities can actually resolve conflicts among themselves and implement conservation effectively better than a centralised authority. This makes the approach more difficult to implement and to measure. Such an approach will not work in the case of groups who lack strong social cohesion; where existing social cohesion is undergoing rapid change; or where there is no unified vision of the kind of community development desired (Brandon 1997). Furthermore, there are obvious limits to the extent that particular national or international conservation targets can be made to coincide with the desires of the local actors (Adams 1998; FAO 2001).

1.3 Biodiversity conservation in India

The concept of natural resource conservation in India is ancient. Maintenance of sacred forests as conservation areas has been practiced in India since pre-agrarian hunter-gatherer societies (Kosambi 1962). The tradition continued over several millennia and is still retained in modern India largely due to the ecological prudence ingrained in the religious traditions of the country (Gadgil & Guha 1995). The traditional nature reserves are often very small and patchy, dotting the Indian countryside. Sacred forests are reported to exist in nearly all regions in India, in different biogeographic zones

(Malhotra *et al.* 2001). Apart form vegetation, they contain various other forest-dwelling life forms and geographical features that are protected with the belief that "to keep them in a relatively undisturbed state is expressive of an important relationship of human beings with nature" (Hughes & Chandran 1998). Although they may cover a tiny total area of the country, the number of such forests is estimated to be between 100,000 and 150,000 in India (Malhotra 1998). Therefore, such sacred forests are an integral part of the modern-day cultural landscape[3].

The formal system of nature reserves has its origins in the former hunting preserves that were declared by the Hindu, Muslim and in more recent history, British rulers (IUCN/WCMC 1992). The British colonial government recognised three types of forest land: reserved forests (most regulated under the authority of the Forestry Department), protected forests, and the unclassified or village forests in which some utilisation by surrounding communities was allowed (Commander 1986). In addition to that, shifting cultivation was banned; hunting preserves were declared, where the local villagers were not allowed; and stringent rules and regulations on land management within such reserves were formulated. Most forests were managed for timber, and the system of privileged hunting in the specially maintained game reserves continued (Rangarajan 1994). Although traditional conservation practices were not officially recognised by the British, in some cases, local management was allowed (Brandis 1897). After Independence in 1947, the first national forest policy was enacted in 1952, which states that the colonial forest policy was the basis for the post-Independence Indian forestry (Guha 1983). According to this policy, many already available hunting preserves were declared as wildlife sanctuaries and national parks. Since then the system of protected areas has expanded further and many new wilderness areas have been declared (Rodgers & Panwar 1988). Currently, India has a network of 87 national parks and 485 wildlife sanctuaries covering 4.75% of the total land area (Forest Survey of India 1999). It appears however, that the formal system of protected areas in India is "flagship species" driven – species such as tiger (*Panthera tigris*) were hunted in the past and subsequently became of serious conservation concern due to their dwindling numbers (Panwar 1982).

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[3] In a cultural landscape two broad component systems, ecological and social, exist – "Both these systems are interlinked in such a way that the 'humans-in-nature' situation is sustainable, wherein, the local social system develops management practices, based on ecological knowledge, for dealing with the dynamics of the ecosystem with which it interacts" (Berkes & Folke 1998).

Furthermore, protected areas are located in lands that are uninhabitable and unsuitable for any other use. A study by the Indian Institute of Public Administration (Kothari *et al.* 1989) found that in India there are few protected areas in the regions with the greatest density of humans, cattle, goats and sheep; and a large number of national parks and wildlife sanctuaries in regions that are remote and sparsely populated (Singh 1999). The formal reserve system seems to miss out the habitats that are situated in the matrix of rural landscape. In the forests of the southern Western Ghats in India, the protected areas have failed to protect patchy populations of butterflies such as the Travancore Evening Brown (*Parntirrhoea marshalli*) and the Malabar Banded Swallowtail (*Papilio liomedon*) (Soubadra Devy & Ganeshaiah 2001 *pers. comm.*). It is likely that the informal system of traditional nature reserves – sacred forests – that are situated in the land outside formal reserves, are instrumental in protecting habitats and elements of biodiversity that are excluded from the formal system (Bhagwat 1998). The ubiquity in distribution of sacred forests in the Indian landscape suggests that they can have a range of different habitats in natural and semi-natural conditions protected within their boundaries.

Although the conservation approach in India is gradually changing, until recently, the conservation policy in the region had its focus on strict protection. The State Forest Departments have wildlife divisions that are in charge of managing protected areas (Kothari *et al.* 1989). The approach to wildlife conservation entails displacement of villages within protected areas in order to prevent any possible human intervention (Kothari *et al.* 1997). Outside protected areas, numerous sacred forest patches maintained by local people exist (Malhotra 1998). However, in most cases, the current government policies do not recognise such forests as community-based conservation areas (*e.g.* Bhatt & Kothari 1997).

Kodagu (formerly Coorg) is a coffee-growing district in the mountain range of the Western Ghats in India, which also has a high density of sacred forests – one forest every 300 hectares – in the landscape (Kushalappa & Bhagwat 2001). While the practice of maintaining sacred forests continues to be commonplace in Kodagu, the formal protected areas are a comparatively recent conservation measure taken by the Government. Today, both these practices exist simultaneously. This, therefore, provides an ideal situation for comparing and contrasting the relative effectiveness of the two systems for the conservation of biodiversity. Because of the long history of landscape modification in the district, the study area provides a promising location for examining long-term effects of landscape modification on biodiversity. The region maintains more than 80% of its land under tree cover (Forest Survey of India 1999), which distinguishes Kodagu from many other tropical landscapes that have faced severe forest fragmentation (*e.g.* Whitmore 1997). This offers an excellent opportunity for comparison between the effects on biodiversity in fragmented landscapes in many tropical countries and those on biodiversity in modified forest landscape of Kodagu. Such information will be useful for formulating a biodiversity conservation strategy in the region and for recommending conservation management of other similar tropical landscapes.

One of the most fashionable approaches is conservation of biodiversity in 'biodiversity hotspots' (Myers 1989, 1990; Myers *et al.* 2000), which are geographical areas that rank particularly high on one or more axes of species richness, levels of endemism, numbers of rare or threatened species and intensity of threat (Reid 1998). The Western Ghats of India has been described as one of 25 global biodiversity hotspots (Myers *et al.* 2000). However, it is argued that the application of this approach fails at 'fine-grained' – *i.e.* landscape – geographic scales (Williams *et al.* 1996; Csuti *et al.* 1997; Margules & Pressey 2000). Furthermore, the use of a hotspot approach in setting priorities for conservation rests on the assumption that patterns of diversity among relatively well-studied 'indicator' groups such as birds, mammals and plants are good predictors of patterns of diversity in less-studied groups such as insects and fungi (*e.g.* Reid 1998). However, such an approach is ineffective, amongst other reasons, for its failure to predict overall patterns of diversity based on the diversity of one or a few taxa (*e.g.* Kerr 1997; Mace *et al.* 2000). This study proposes to measure species and community level diversity in three contrasting groups of organisms – trees, birds and macrofungi, using similar methods; it will compare and contrast the differences in diversity patterns at a landscape scale, in order to examine whether there is congruence in the distribution of the three groups of organisms in Kodagu.

Although the wider meaning of the term biodiversity[4] is recognised, diversity of species[5] - or in more precise terms, richness of recognisable taxonomic units (Oliver & Beattie 1996) - is referred to as 'biodiversity' for convenience in the rest of the thesis.

Bierregaard *et al.* (1997) identify the following problems in the studies of biodiversity in fragmented tropical landscapes:

- Different methods;
- Different habitats and biotas;
- Different target taxa;
- Different spatial scales;
- Different matrix habitats and land uses surrounding fragments;
- Different fragment ages.

In the present study, trees, birds and macrofungi, which are biologically and ecologically different, are surveyed in the same landscape using similar methods. Trees are stationary (although their propagules - pollen and seeds - are not) and have long generation times; birds are mobile and relatively short-lived. Little information is available on dispersal and longevity of macrofungi, and the processes that govern their fruiting are not well known. Nevertheless they have important role in ecosystem, such as forming mycorrhizal associations with plants and recycling nutrients (Fogel 1980; van der Heijden *et al.* 1998; Dahlberg 2001). While the effects of fragmentation are very well documented and researched for trees (*e.g.* Turner *et al.* 1996; Kapos *et al.* 1997; Thébaud & Strasberg 1997) and birds (*e.g.* Thiollay 1994; Warburton 1997; Renjifo 1999), there is no information on such effects on macrofungi. This study will attempt to compare and contrast the differences in diversity in the three groups of organisms.

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[4] Biodiversity refers to the variety and variability among living organisms, the ecological complexes in which they naturally occur and the ways in which they interact with each other and with the geosphere (CBD 1992).

[5] Species diversity refers to the variety of living species at the local, regional and global scale. It can be measured in a number of different ways that differentially weigh presence and absence versus frequency of different species at a given locality. The species is the unit most commonly used by biologists to categorise the variation of life. It is also the unit best understood by laypeople. As a result of this, coupled with the pioneering efforts of taxonomists, much of the attention on biodiversity has been focused at the species level (Sanderson & Redford 1997).

In conclusion, the effects of fragmentation and landscape modification at two scales, patch and landscape scales are identified in this chapter. The focus is on the landscape-scale effects. Three prominent heuristic frameworks – island biogeography, the metapopulation theory and landscape ecology – have been advanced in order to study fragmented landscapes. The suitability of each for explaining diversity at species and community levels is explored in this chapter. While diversity[6] below species and above community level is important, it is beyond the scope of this work. The consequences for diversity, of the interplay between size of a forest patch, its isolation from large forest and the surrounding matrix are discussed in this chapter. Two contrasting conservation approaches – strict protection and community-based conservation – currently in practice are considered, and the possibility of comparing and contrasting the conditions under which small-scale community-based conservation areas are more effective than fortress conservation areas and *vice versa*, is explored in this chapter.

In summary, this study addresses the following questions:

- How does landscape modification affect diversity in trees, birds and macrofungi; and what differences are there in the responses of these organisms?
- Which landscape factors are useful in explaining differences in the composition of ecological species groups, community diversity and similarity in trees, birds and macrofungi?
- What types of organisms are the formal protected areas and the community-based conservation areas suitable for?
- What could be the most effective strategy for combining the two contrasting approaches to biodiversity conservation?

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[6] Diversity can be measured in form of different components – genes, species, higher taxonomic levels, communities, biotic processes, ecosystems and ecosystem processes – and at different scales – spatial and temporal. Different components of diversity can be measured in number or relative frequency or both (Magurran 1988).

2 . MATERIALS AND METHODS

2.1 Study Area

2.1.1 Kodagu district in the Western Ghats of India

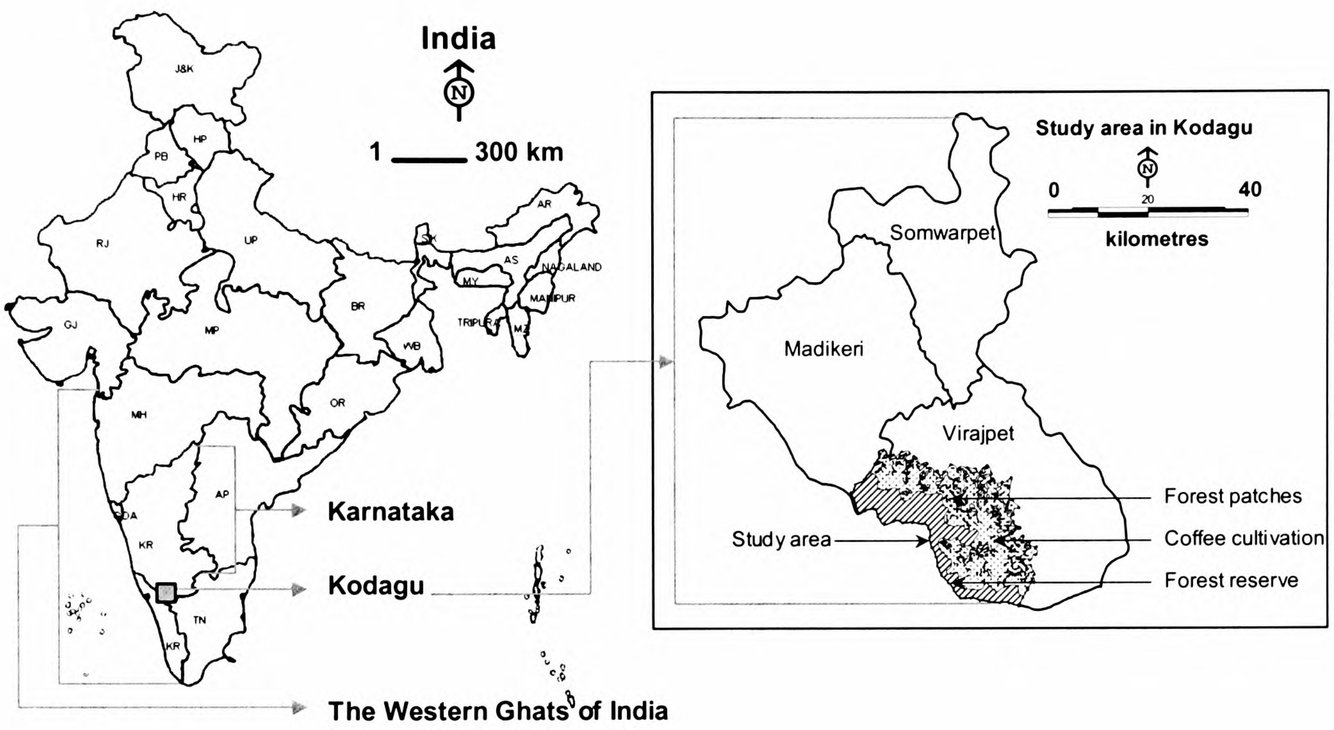

Figure 2.1 The study area lies in the southern part of Kodagu district, called Virajpet taluk (county), in the Western Ghats of India.

Geographical location and physiography: The Western Ghats region of India (*Figure 2.3*, *Figure 2.3*) has been described as one of 25 global biodiversity hotspots due to high levels of endemism and the necessity to conserve endangered flora and fauna in the face of the dwindling cover of primary vegetation (Myers *et al.* 2000). The Western Ghats is a chain of mountains 1600 km long and 5 – 150 km wide along the west coast of India between 8° 22' – 20° 40' N latitude and 73° – 77° E longitude. Kodagu district, situated mainly on the eastern slopes of the mountain range, extends between 11° 56' – 12° 52' N and 75° 22' – 76° 11' E (Pascal & Meher-Homji 1986). The lowest altitude in the district is about 300 m (FRLHT 2000a) and the highest peak, Tadiandamol, is at 1734 m above sea-level (Pascal & Meher-Homji 1986). The study area (*Figure 2.1*) is in the southern part of the district with an average altitude of 900 m (*Figure 2.4*) (FRLHT 2000a) – the lowest sampling locality at 799 m and the highest at 966 m (*Table 2.1*).

Climate, geology and soil: The mean temperature of the coldest month in the study area ranges between 16 and 23 degrees centigrade; rainfall between 2000 and 5000 mm per year (*Figure 2.2* and *Figure 2.4*) with four months dry (Pascal & Meher-Homji 1986). The main rock formations belong to the most ancient Archaean system; the basement rock is composed of peninsular gneiss, gneissic granites and banded gneiss (Akbar Sha 1987). The soils in the study area are classified as red loamy soils (*Figure 2.4*) (FRLHT 2000a), which have mature profiles, although the surface soils have been eroded from the site of formation in many places, making soils generally poor in plant nutrients (Akbar Sha 1987).

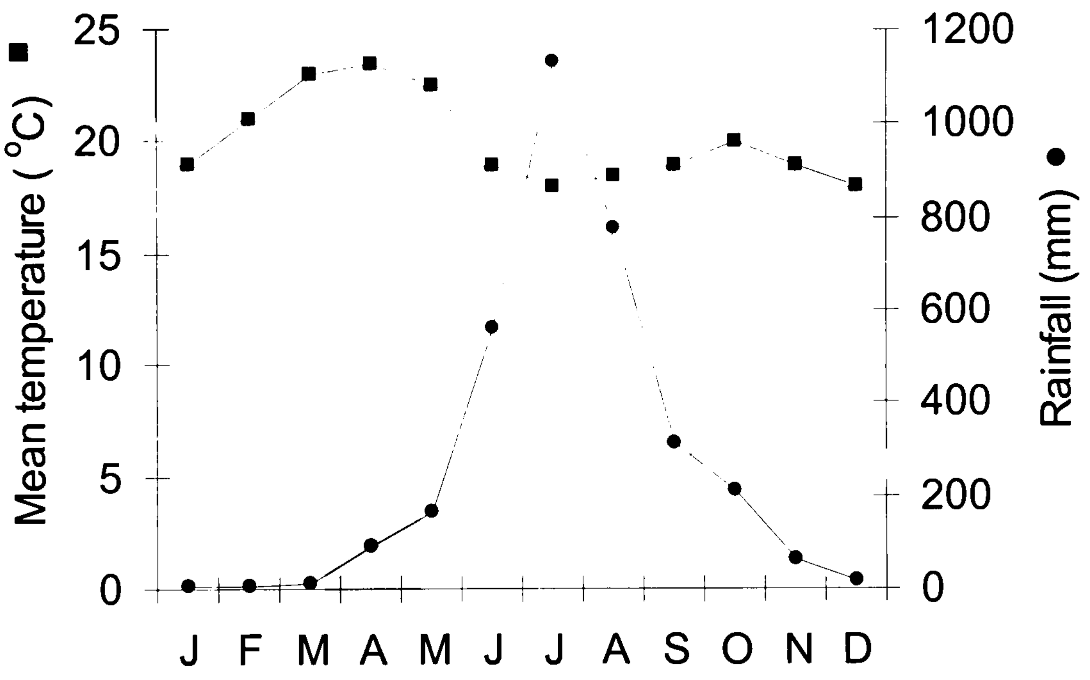

Figure 2.2 Mean monthly temperature and monthly average rainfall at Madikeri in Kodagu (Data collected for 30 years). Sources: Temperatures – Pascal 1988; Rainfall – Akbar Sha 1987.

Figure 2.3 (Please see Page 35) The Western Ghats of India has been designated as one of 25 global biodiversity hotspots (Myers et al. 2000). Source: National Geographic Society, 2001.

*Figure 2.4 (Please see Page 36) Altitude, rainfall, soil and vegetation in Kodagu. Source: FRLHT, 2000a. Note: The study area is marked by a **black** outline.*

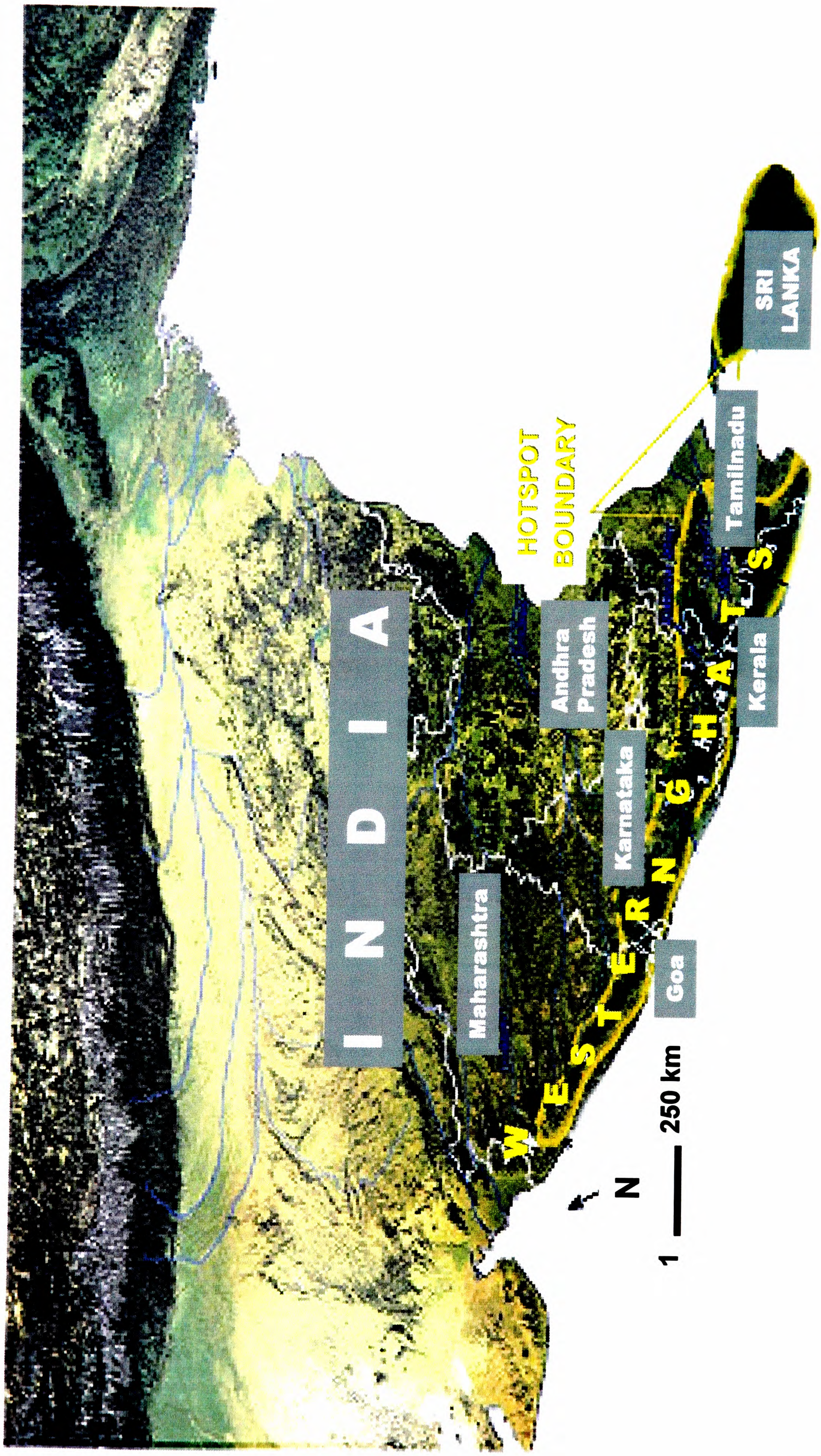

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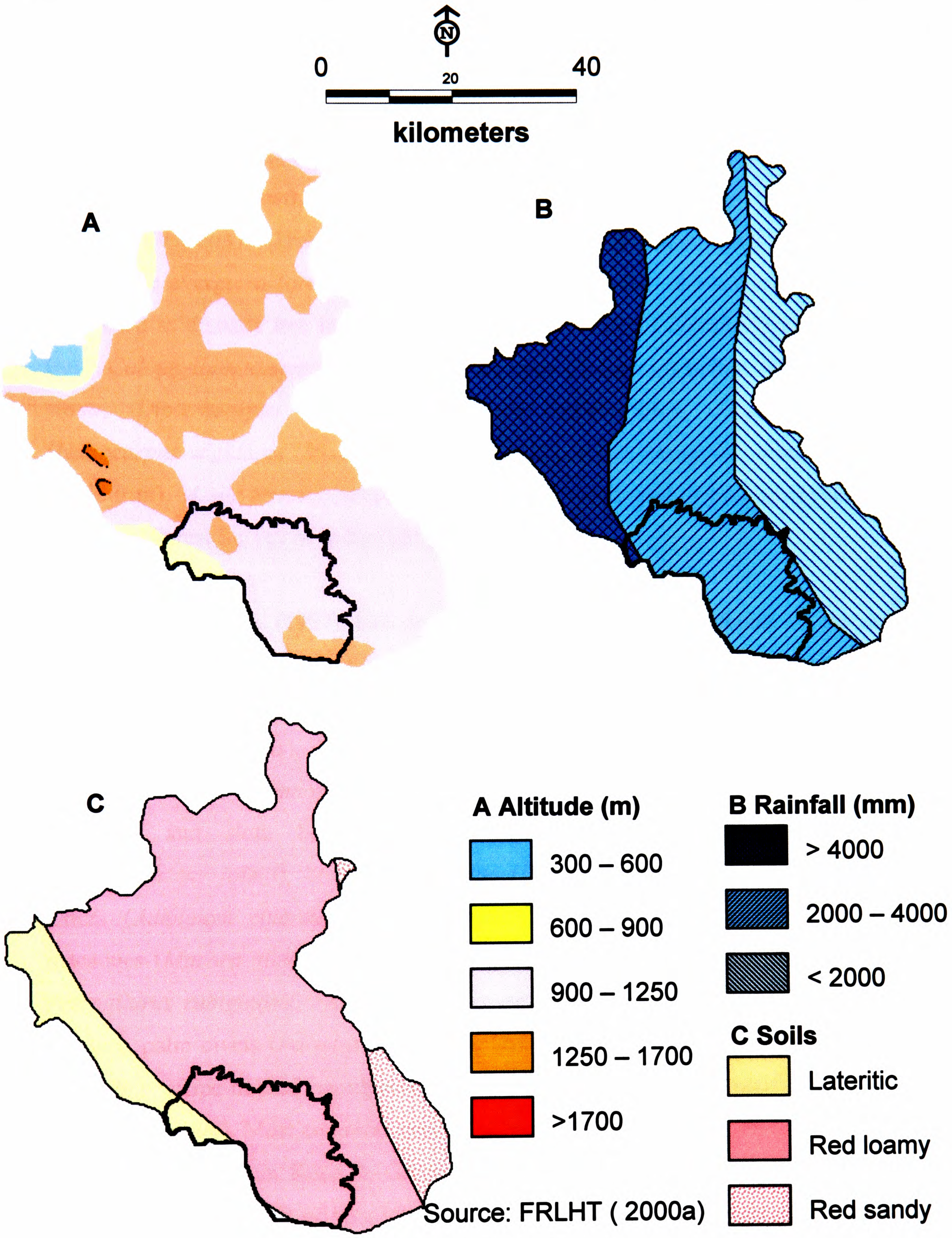
Figure 2.3 - 35



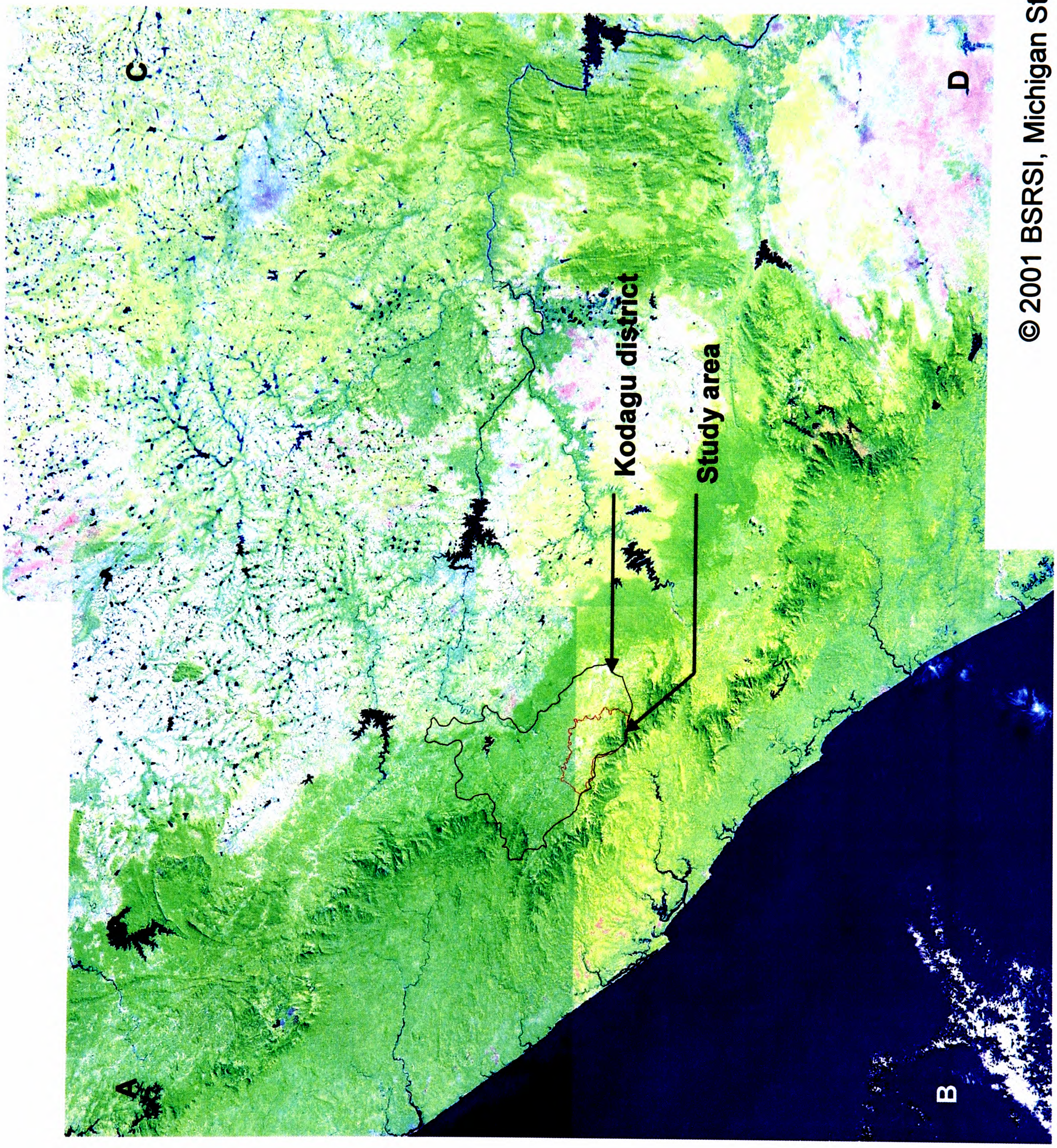
Source: FRLHT (2000a)

Vegetation: The forest in the study area is wet evergreen, and Pascal & Meher-Homji (1986) have defined the vegetation as medium elevation (between 750 and 1400 m) *Mesua ferrea – Palaquium ellipticum* type, which is most common in the region. The important Angiosperm families include: Clusiaceae (*Mesua, Calophyllum, Garcinia*), Sapotaceae (*Palaquium*), Meliaceae (*Aglaia*) and Euphorbiaceae (*Agrostistachys, Mallotus, Drypetes*). Akbar Sha (1987) has listed common and economically important trees in the evergreen forests, most of them used as timber: *Acrocarpus fraxinifolius* (also used as a shade tree in coffee plantations), *Artocarpus hirsutus* (also has edible fruit), *Calophyllum tomentosum*, *Canarium strictum* (also yields Indian damar), *Cedrela toona*, *Dipterocarpus indicus*, *Dysoxylum malabaricum*, *Hopea parviflora*, *Hydnocarpus wightiana* (fruits have medicinal properties), *Mangifera indica* (also has edible fruit), *Mesua ferrea*, *Sterculia alata* (has soft wood used in packaging) and *Vitex altissima* (Forest Survey of India 1995).

Wildlife: Akbar Sha (1987) has described prominent animals found in the area. Common: Asian elephants (*Elephas maximus*), Indian gaurs (*Bos gaurus*), deer – *Cervus axis axis*, *Cervus unicolour*, *Muntiacus muntjac*, Indian wild boars (*Sus scrofa*), tigers (*Panthera tigris*), leopards (*Panthera pardus*), wild dogs (*Cuon alpinus*), Hanuman langurs (*Semnopithecus entellus*), Malabar giant squirrels (*Ratufa indica centralis*) and sloth bears (*Melursus ursinus*). Occassional: Nilgiri langurs (*Trachypithecus johnii*), brown palm civets (*Paradoxurus jerdoni*), Asian Clawless Otters (*Amblonyx cinereus*), Nilgiri Martens (*Martes gwatkinsii*) and lion-tailed macaques (*Macaca silenus*). Rare: jackals (*Canis aureus*), small cats – *Felis chaus*, *Prionailurus rubiginosis*, *Felis bengalensis*, small Indian civets (*Viverricula indica*), common palm civets (*Paradoxurus hermaphroditus*), common mongooses (*Herpestes edwardsi*), stripe-necked mongooses (*Herpestes vitticollis*) and ruddy mongooses (*Herpestes smithi*). Most common species of birds that occur elsewhere in the Western Ghats are also seen in Kodagu. Great hornbills (*Buceros bicornis*), Malabar trogons (*Harpactes fasciatus*), fairy blue birds (*Irena puella*), broad-billed rollers (*Eurystomus orientalis*) and black eagles (*Ictinaetus malayensis*) are reported to be relatively rare (Akbar Sha 1987; Forest Survey of India 1995).

Figure 2.5 (Please see Page 39) Current Landsat imagery shows the study area in Kodagu district. Source: BSRSI Michigan State University (2001). The Landsat imagery is composed of four panels: A – Path 145, row 51; B – Path 145, row 52 (both taken on 20 December 2000); C – Path 144, row 51; D – Path 144, row 52 (both taken on 26 October 2000) ***Note****: The* ***black*** *outline indicates the boundary of Kodagu district and the* ***red*** *outline indicates that of study area.*

Figure 2.6 (Please see Page 40) Change in land cover in Kodagu during 1977 and 1997. Source: Elouard (2000b). The area of medium elevation evergreen forest has been lost by conversion into coffee plantations. ***Note****: The* ***red*** *lines mark the boundary of the study area,* ***dark green*** *indicates evergreen forest,* ***yellow*** *indicates moist deciduous forest zone and* ***pink*** *indicates coffee plantations. Elouard (2000b) states that the 1977 map is based on the interpretation of Landsat image, confirmed through fieldwork (cross-checking land cover and analysing forest types with the help of forest plots established in order to study floristic diversity of the district). The 1997 map is based on the interpretation of image obtained from the Indian Remote-sensing Satellite (IRS 1C), confirmed through fieldwork for notification of the new boundaries and changes in land cover types. Because of the fact that Landsat is not as detailed and precise as IRS 1C, a corrected 1977 map was prepared in order to eliminate the errors. The analyses of the landscape transformation are based on comparison between this map and the 1997 map. The same land cover description was used (Elouard 2000b).*



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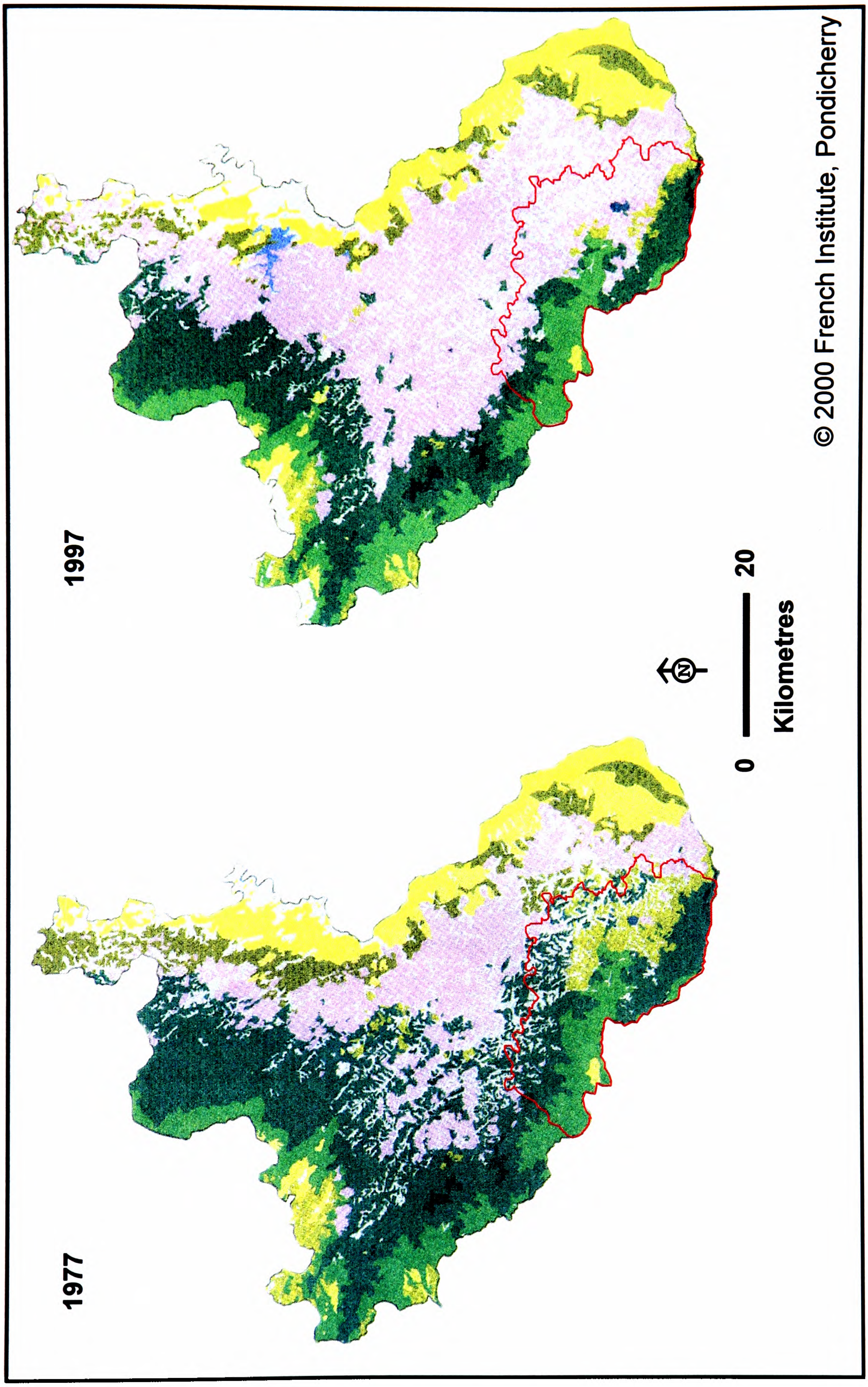
1997
1977
0
20
Kilometres
© 2000 French Institute, Pondicherry

Landscape structure

The current Landsat imagery shows the landscape in the region surrounding the study area (*Figure 2.5*).

Land use

The climate and soil in the region are suitable for coffee cultivation, which is the most prominent land use occupying 29% of the land area of Kodagu district (Elouard 2000a) and nearly 60% in the study area (personal estimate). Pepper, cardamom, ginger and paddy are other important crops. Kodagu is believed to have undergone a rapid landscape transformation since the mid 19th century and 35% of its original evergreen forest area has been planted with coffee (Elouard 2000b). The increase in the area under coffee cultivation between 1977 and 1997 (*Figure 2.6*) has resulted in a loss of forest habitat. However, due to the structural complexity and floristic diversity of trees, shaded coffee plantations are known to resemble the original forest and thereby have relatively high biodiversity (*e.g.* Perfecto *et al.* 1996; Moguel & Toledo 1999). Therefore, coffee cultivation in Kodagu is an example of forest modification where the tree-cover is little altered – total tree-cover in the study area is 81% (Forest Survey of India 1999). Currently, government-owned protected areas occupy about 30% of the study area, 60% is under shade-grown coffee plantations and 2% is occupied by other tree-covered land such as sacred forests (community conserved lands), Paisaris (community-use lands), Jammas (family-owned lands) (Vijaya 2000). Only 8% of the total area is not covered by trees, such as paddy fields (Moppert 2000). Although land tenure in Kodagu is complex (Bar Association Virajpet 1998) (*APPENDIX 2*), the above categorisation into tree covered and non-tree covered land is sufficient for the present study.

Landscape transformation

The ecological history of the Western Ghats in general and of Kodagu district in particular, has been instrumental in shaping the present landscape in the study area. Chandran (1997) speculates that the first inhabitants of the Western Ghats forests were hunter-gatherers in the Palaeolithic and Mesolithic period (13,000 – 10,000 BC) whose activities did not have any major impact on the Western Ghats forests. The agri-pastoralist societies in the Neolithic period (3000 – 1000 BC) were responsible for the initiation of major landscape changes and were also responsible for the conversion of land from tree-covered to non tree-covered types. The societies in the Megalithic period (1000 BC – 1 AD) with their iron tools caused large-scale forest clearance. Kodavas, the present occupiers of Kodagu, are believed to have arrived just before the beginning of the first millennium (Ponnappa 1997). They took up agriculture as their main occupation and cultivated most of the valley bottoms in this hilly terrain with paddy. The subsequent spread of settled agriculture caused a decline in the area of valley forests. The tribal societies co-habiting the land were engaged in shifting cultivation on the hills. The pre-colonial landscape in Kodagu during the reign of the Rajas (regional Hindu rulers before the advent of British rule) could have been as described above – with most areas at medium elevation (900 to 1250 m) under tree cover, while the low-lying ground was under paddy cultivation and high mountain slopes under shifting cultivation.

Coffee cultivation

The British started commercial timber harvesting from natural forests and introduced coffee cultivation in the mid 19th century. This is believed to have brought about more severe landscape transformations than in the past (Elouard 2000b). Cultivation of coffee also continued unabated in post-independence India and it soon became the major commercial enterprise. As a result, there was little dependence on arable crops and the extent of non-tree covered land did not change significantly between about 1850 and 1980 (Moppert 2000). The landscape transformations between 1980 and 2000 have been dramatic because of the intensification and privatisation of coffee cultivation. It can be argued, however, that this is a peculiar example, where the quality of the forest has deteriorated, but the total tree cover is little altered. Thus, this example does not comply with the idea of landscape fragmentation in many other tropical regions (*e.g.* see Laurance & Bierregaard 1997) or landscape transformation as referred to by Elouard (2000b). It could be more appropriately called 'landscape modification'.

Sacred forest tradition

The tradition of protecting patches of forest in cultivated landscapes for local religious purposes is ancient in Kodagu (*e.g.* Chandran *et al.* 1998). These patches are locally referred to as Devarakadus, meaning the God's forests. Brandis (1897) mentions the existence of such practices even before the advent of British rule in India. During British rule, community management of land such as sacred forests, was replaced by government management (Chandran 1997). Currently, sacred forests in Kodagu are owned by Government and managed by local village committees. There are 1214 sacred forests according to official records in Kodagu. Their density is high, with approximately one forest for every 300 ha of land. Each village has at least one, and some have as many as 17 sacred forests. Although the district has large number of such forests, most of them are very small. Out of the 1214 sacred forests, 997 (80%) are less than 2 ha and only 123 exceed 4 ha in extent. The largest sacred forests are about 50 ha (Kushalappa & Kushalappa 1996). Although numerous, the sacred forests in the study area occupy an estimated total of less than 2% of the geographical area.

Protected area network

The earliest attempt to impose formal wildlife protection measures in Kodagu was in 1879 through the Elephant Preservation Act (Akbar Sha 1987), possibly because elephants were useful for transporting timber during British rule. The principal goal of land management, namely exploitation of forests for timber, remained the same in the post-independence Kodagu as seen from the Forest Department working plans (Somiah 1954; Akbar Sha 1987), although some elements of wildlife conservation appear to have been brought into forest management. After independence, the Karnataka Forest Act of 1963 enforced strict protection of the forest reserve. Up until 1987, only Brahmagiri wildlife sanctuary had been formed in order to protect the evergreen forests of Kodagu. Recently, the protected area network has been expanded by the declaration of two more sanctuaries, Talakaveri and Pushpagiri, along the western boundary of the district. The forest reserve in Kodagu acts as a buffer for the three wildlife sanctuaries. The total protected area network occupies more than 30% of Kodagu, which is much above the national and global averages, as well as the IUCN (1994) recommendation of 10%.

Despite the extensive landscape modification, Kodagu appears to provide a remarkable example of a balance between economy and ecology. Coffee plantations generate employment in the region and make it comparatively wealthy, while the extensive tree cover results in the protection of more forest-dwelling organisms in comparison with other regions of India (Akbar Sha 1987). The forest reserve owned by the Government occupies a contiguous stretch of forest that is strictly protected. The rest of the landscape is man-made and typically has patches of forest scattered among coffee plantations and paddy fields.

2.2 Data collection and processing

2.2.1 Measuring biodiversity

The richness of trees, birds and macrofungi was measured in the present study. These three groups were chosen because of the differences in their biology and ecology that are likely to manifest themselves in the responses to long-term landscape modification.

- *Trees* belong to the Plant Kingdom and are among the producers in the ecosystem. Their taxonomy is among the best-studied in the Western Ghats. Keys for the identification of trees are readily available for the region (*e.g.* Pascal & Ramesh 1987).

- *Birds* belong to the Animal kingdom and are one of the many groups of consumers in the ecosystem. Their taxonomy is also very well studied in the Western Ghats and field guides are available for their identification (*e.g.* Ali 1996; Grimmett *et al.* 1998).

- *Macrofungi* are typically the fungi that are visible to the unaided human eye (Watling 1995). Macrofungi are important decomposers in the ecosystem and are also known to form mycorrhizal associations with plants (Dahlberg 2001).The taxonomy of macrofungi is extremely poorly studied in the Western Ghats and there is not much literature available for their identification. Thus, the relevant published information about African and European macrofungi (Pegler 1977; Jordan 1995) was used in order to identify macro-fungal sporocarps. They were identified as morpho-species based on their external characteristics.

On account of the previous experience of working in the Western Ghats (see Ghate *et al.* 1997; Achar *et al.* 2000), I was able to identify trees and most birds in the region. The technique for identifying morpho-species of macrofungi was acquired with the help of literature and assistance from a regional expert.

Site selection

The sampling localities were selected at various places along the whole of the forest reserve, which adjoins the Brahmagiri Wildlife Sanctuary in southwestern Kodagu. Sacred forests were at variable distances from the nearest forest boundary and from each other. Coffee plantations were ubiquitous outside the forest reserve and between the sacred forest patches. With advice from the local experts at the Forestry College, Ponnampet, and after initial reconnaissance surveys, 58 localities (*Table 2.1*, *Figure 2.12*) were selected for sampling in approximately 60 km^2 in the Virajpet taluk (county) of Kodagu (*Figure 2.1*). 25 sacred forests, 23 coffee plantations and 10 localities within the forest reserve were sampled for trees, birds and macrofungi in 1999 and 2000. Patchy landscapes generally have a higher β diversity (species turnover) than contiguous forests due to a higher heterogeneity of habitats at the landscape scale (Lapin & Barnes 1995; Magurran 1988). In order to capture this diversity, a greater number of coffee plantations and sacred forests were sampled than localities in the forest reserve.

A team of five or six forestry graduates including myself carried out the sampling. During sampling, temporary camps were established in 10 villages in the Virajpet taluk. Sampling was carried out by staying at each camp for about a week during the dry season, when most tree and bird sampling was undertaken. During the wet season, when the sampling for macrofungi commenced, camping was not possible. Therefore, sampling localities were visited periodically starting from the headquarters every morning and coming back to the base after the day's work. Most localities were accessible by public transport or by two-wheeler vehicles.

Table 2.1 List of sampling localities from the forest reserve, sacred forests and coffee plantations in Kodagu. The localities are identified by the abbreviated names of villages in which they occurred; the number at the end of each locality name represents the order in which the localities were sampled for the first time.

Sampling locality *	Number **	Latitude – Longitude ***	Altitude (m) ***	Area (ha)	Distance from the reserve (km)
Reserve					
1. Bgrf31	R1	11° 59’ 44” N, 75° 57’ 50” E	870	NA	0
2. Bgrfr32	-	11° 59’ 44” N, 75° 57’ 50” E	870	NA	0
3. Hgrf43	R2	12° 08’ 00” N, 75° 45’ 42” E	878	NA	0
4. Kurfn49	R3	12° 05’ 24” N, 75° 50’ 02” E	927	NA	0
5. Kurfs50	R4	12° 05’ 22” N, 75° 50’ 03” E	923	NA	0
6. Thrfc41	-	11° 59’ 07” N, 75° 53’ 55” E	856	NA	0
7. Thrfn03	R5	11° 58’ 48” N, 75° 54’ 47” E	833	NA	0
8. Thrfs40	R6	11° 58’ 45” N, 75° 54’ 40” E	832	NA	0
9. Torfe36	R7	12° 09’ 36” N, 75° 41’ 57” E	857	NA	0
10. Torfw35	R8	12° 09’ 36” N, 75° 41’ 57” E	857	NA	0
Sacred					
11. Bgdsg30	S1.6	11° 59’ 02” N, 75° 56’ 52” E	935	21.59	1.4
12. Bkdsg26	S2.5	12° 01’ 57” N, 75° 54’ 07” E	855	9.4	2.7
13. Brdsg21	S3.4	12° 05’ 59” N, 75° 55’ 53” E	812	1.3	5.3
14. Brlsg01	S2.1	12° 05’ 45” N, 75° 54’ 30” E	799	2.4	1.7
15. Btdsg42	S3.1	12° 11’ 24” N, 75° 47’ 19” E	879	12.1	4.8
16. Hgdsg18	S1.5	12° 08’ 40” N, 75° 46’ 04” E	917	2.4	1.0
17. Hglsge16	S1.3	12° 08’ 35” N, 75° 46’ 07” E	912	39.7	6.3
18. Hglsgw15	S1.4	12° 08’ 35” N, 75° 46’ 07” E	912	39.7	6.9
19. Htdsg57	-	12° 07’ 32” N, 75° 55’ 36” E	822	8.9	8.2
20. Icdsg53	S3.7	12° 09’ 00” N, 75° 54’ 31” E	858	4.0	8.0
21. Kbdsg51	S2.7	12° 09’ 55” N, 75° 51’ 37” E	843	12.4	4.3
22. Kdpaim54	S3.8	12° 09’ 44” N, 75° 54’ 07” E	966	6.6	8.6
23. Ktdsg44	S3.3	12° 11’ 35” N, 75° 46’ 03” E	918	0.2	5.1
24. Kudsg45	S2.2	12° 06’ 10” N, 75° 51’ 53” E	860	3.7	1.9
25. Kuhdsg48	S2.6	12° 06’ 38” N, 75° 52’ 08” E	857	3.2	2.8
26. Kulsg47	S2.3	12° 06’ 20” N, 75° 51’ 50” E	847	1.4	2.2
27. Pdlsg27	S3.2	12° 01’ 53” N, 75° 55’ 29” E	870	7.1	4.9
28. Pldsg13	S2.4	12° 11’ 13” N, 75° 43’ 05” E	930	2.1	2.5
29. Pllsge08	S1.7	12° 11’ 12” N, 75° 42’ 57” E	935	48.1	1.4
30. Pllsgw11	S1.2	12° 10’ 55” N, 75° 42’ 56” E	956	48.1	4.7
31. Topaij39	S1.1	12° 09’ 28” N, 75° 42’ 44” E	910	NA	1.6
32. Tslsg58	S3.6	12° 02’ 24” N, 75° 57’ 22” E	820	14.0	6.4
33. Vbdsg52	S3.5	12° 09’ 11” N, 75° 52’ 27” E	824	7.6	5.7
34. Wndsg24	S2.9	12° 01’ 07” N, 75° 57’ 12” E	845	2.3	4.4
35. Wnlsg22	S2.8	12° 00’ 54” N, 75° 56’ 49” E	849	18.8	4.3
Coffee					
36. Bgcofd33	-	11° 59’ 02” N, 75° 56’ 52” E	935	NA	1.3
37. Bgcofs34	-	11° 59’ 02” N, 75° 56’ 52” E	935	NA	1.4
38. Bkcofc29	-	12° 01’ 57” N, 75° 54’ 07” E	855	NA	2.9
39. Brcofj02	-	12° 05’ 45” N, 75° 54’ 30” E	799	NA	2.7
40. Brcofn20	-	12° 05’ 45” N, 75° 54’ 30” E	799	NA	2.3
41. Hgcofa19	-	12° 08’ 35” N, 75° 46’ 07” E	912	NA	1.2
42. Hgcofu17	-	12° 08’ 35” N, 75° 46’ 07” E	912	NA	1.5
43. Kdcofl55	-	12° 09’ 44” N, 75° 54’ 07” E	866	NA	8.3
44. Kdcofs56	-	12° 09’ 44” N, 75° 54’ 07” E	866	NA	8.5
45. Kucofp46	-	12° 06’ 10” N, 75° 51’ 53” E	860	NA	2.1

46. Pdcofc28	-	12° 01’ 53” N, 75° 55’ 29” E	870	NA	4.7
47. Plcofc10	-	12° 10’ 55” N, 75° 42’ 56” E	956	NA	3.4
48. Plcofd12	-	12° 11’ 06” N, 75° 43’ 12” E	910	NA	1.2
49. Plcofh09	-	12° 11’ 12” N, 75° 42’ 57” E	935	NA	1.5
50. Plcofr14	-	12° 11’ 24” N, 75° 42’ 57” E	902	NA	2.1
51. Thcofa04	-	12° 00’ 00” N, 75° 53’ 43” E	847	NA	1.8
52. Thcofb05	-	11° 59’ 47” N, 75° 53’ 37” E	865	NA	1.7
53. Thcofg06	-	11° 59’ 11” N, 75° 54’ 08” E	836	NA	8.8
54. Thcofs07	-	11° 59’ 11” N, 75° 54’ 08” E	836	NA	8.0
55. Tocofc38	-	12° 09’ 28” N, 75° 42’ 44” E	910	NA	8.0
56. Tocofj37	-	12° 09’ 28” N, 75° 42’ 44” E	910	NA	7.0
57. Wncofd25	-	12° 01’ 07” N, 75° 57’ 12” E	845	NA	4.5
58. Wncofl23	-	12° 00’ 54” N, 75° 56’ 49” E	849	NA	4.1

*Note: * indicates all sampling localities (Please see Figure 2.12) ** indicate sampling localities that had sufficient data in order to include them in similarity analyses (Please see Figure 3.8); *** indicate GPS readings taken at the nearest possible point from the sampling locality*

Sampling design

There is no standard sampling design for inventories of diversity in multiple taxa at a landscape scale (Bierregaard *et al.* 1997). The sampling design adopted here (*Figure 2.7*) is an attempt to provide a framework for inventories aimed at rapid assessment of biodiversity in more than one taxon. In the present study, the sampling localities were very variable in size. The forest reserve was uninterrupted and large. Sacred forests were patchy. The coffee plantations, although contiguous, were composed of small individual land-holdings. In order to overcome the problem of variable sizes and differences in the biological and ecological characteristics of organisms in question, emphasis was on sampling randomly the same number of individuals (observations in the case of birds and macrofungi) at each site rather than sampling equal areas. An individual (for trees) is an ecologically more meaningful unit than an area (Condit *et al.* 1996). For assessing diversity of birds, numbers of observations are more meaningful for comparison than measures such as the time spent recording, which will result in different number of observations depending upon the time of the day when the sampling is carried out (Bibby *et al.* 1998). Rapid assessment methods that make use of morphological characteristics of an organism for species identification are suitable for measuring diversity in a group such as macrofungi, for which shortage of expertise is compounded by the long time required to sort records down to the level of species (*e.g.* Balmford *et al.* 2000).

At each locality, trees were sampled before birds and macrofungi. Prior to visiting the sampling locality (for tree sampling), random numbers, in multiples of five, were generated. A base line was selected at each locality (marked by 'St' and 'Fn' in *Figure 2.7*), which often ran along a natural or man-made linear landscape feature (*e.g.* a cart track, path, fence, boundary, stream) across the extent of the area. Starting points of individual transects were located on the base line in the same sequence as the random numbers (*Figure 2.7*). For example, if the first random number was 100, transect number one was placed at 100 m from the starting point along the base line on a randomly chosen side – left or right. After completion of sampling along the first transect, the second transect was placed at the distance equal to the second random number (*i.e.* 225 m – see *Figure 2.7*) from the starting point. The process was repeated until a sufficient number of trees had been sampled at each locality.

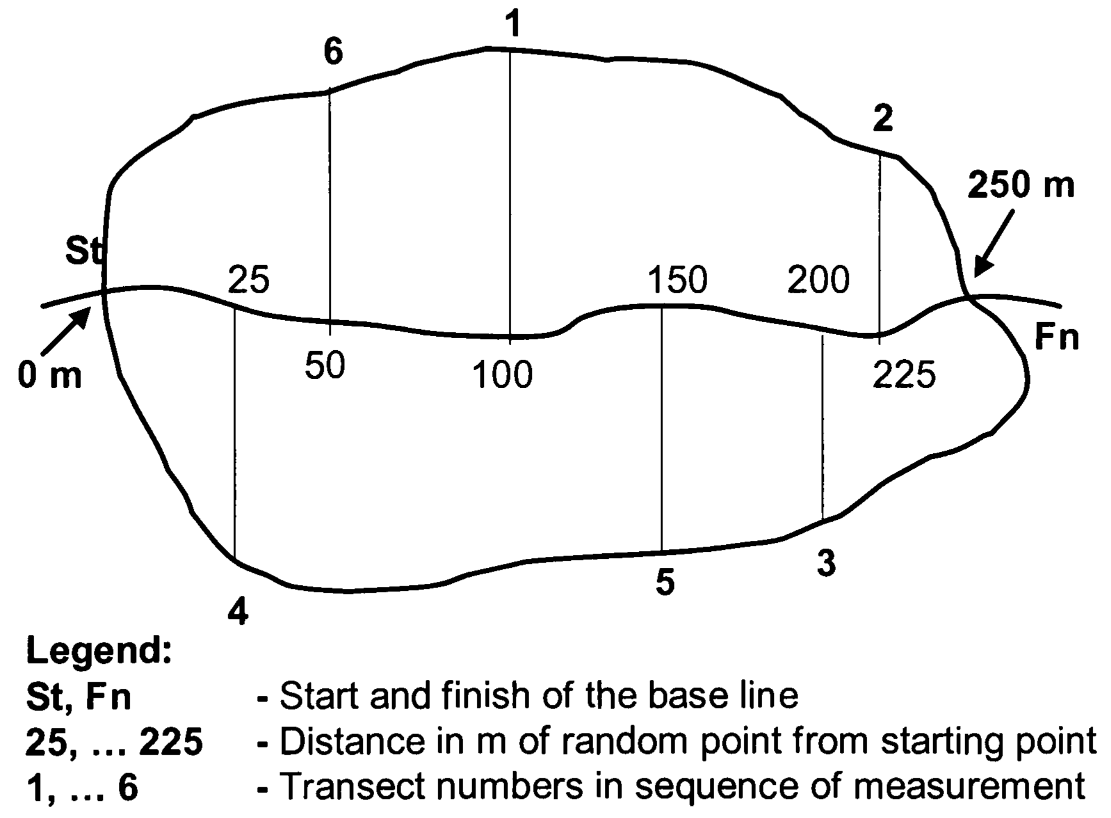

Figure 2.7 Sampling design: A hypothetical sampling locality with a base line running across the patch and the framework of transects placed at random points along the base line, on a randomly chosen side.

The base line was demarcated by painting arrows on adjacent trees using blue oil paint which allowed the relocation of the line easily when the sampling locality was revisited in order to sample birds and macrofungi. The direction of the base line was usually along the cardinal directions. The vegetation transects were established exactly along a North - South line if the base line was roughly East - West and *vice versa*. Thus, at each locality, one base line and several transects running parallel to each other and perpendicular to the base line provided a framework for sampling. The start and the finish of each transect was marked and all the woody plants ≥ 10 cm dbh falling within the boundaries of the transects were marked with blue paint. Sampling for birds and macrofungi was carried out later using the same framework. In order to make the sampling design compatible for inventories of all taxa under investigation and to consider a possibility of re-measuring diversity for monitoring in future, this sampling framework was found to be the most suitable.

Trees

Either rectangular or square plots established in a systematic or random fashion are the most widely used sampling units for trees – the plots range from a few hundred square metres (*e.g.* Pascal 1988) to as large as 50 ha (*e.g.* Condit 1994). In the Western Ghats, due to the undulating topography and rough terrain, researchers have used plots of different sizes ranging from several 10 x 10 m quadrats to a single plot of 1000 x 500 m (Chandran 1993, cited in Ghate *et al.* 1997). However, plot-based inventories are time consuming and expensive due to the work involved in measuring and marking the exact boundaries of plots, which often end up being inaccurate due to undulating terrain. Thus, randomly located transects ranging in size from 20 x 5 m to 100 x 5 m were used in this study in order to sample tree diversity in the forest reserve and sacred forests. Such narrow strip transects have been found useful in studies by Gentry (1990) in southern America and by Daniels *et al.* (1995) in the Western Ghats. The transects in coffee plantations were wider – between 20 x 10 m and 50 x 10 m (see *Figure 2.8*) – because there was a lower density of trees than in other two land use types.

Forest reserve and sacred forests

Typically 100 x 5 m belt transects were established in each sampling locality in the forest reserve and sacred forests (*Figure 2.8 a*). A nylon rope, 100 m in length, was used to mark the central line of the transect while sampling trees. Wooden sticks, 2.5 m in length, were used to measure the distance on each side from the rope. Trees were sampled in the five metre wide belt. On average, in any locality, six transects were established (minimum three and maximum nine). There was variation in the length of transects as well as the number in each sampling locality because of the limited extent of some sacred forest patches and huge variations in stem densities between disturbed and undisturbed sites. In places where the boundary of the sampling locality was less than 100 m from the base line, shorter transects (as short as 20 m) were established. Thus, the effective area sampled was between 1200 m^2 and 3500 m^2 depending upon the extent of the sampling locality and tree density.

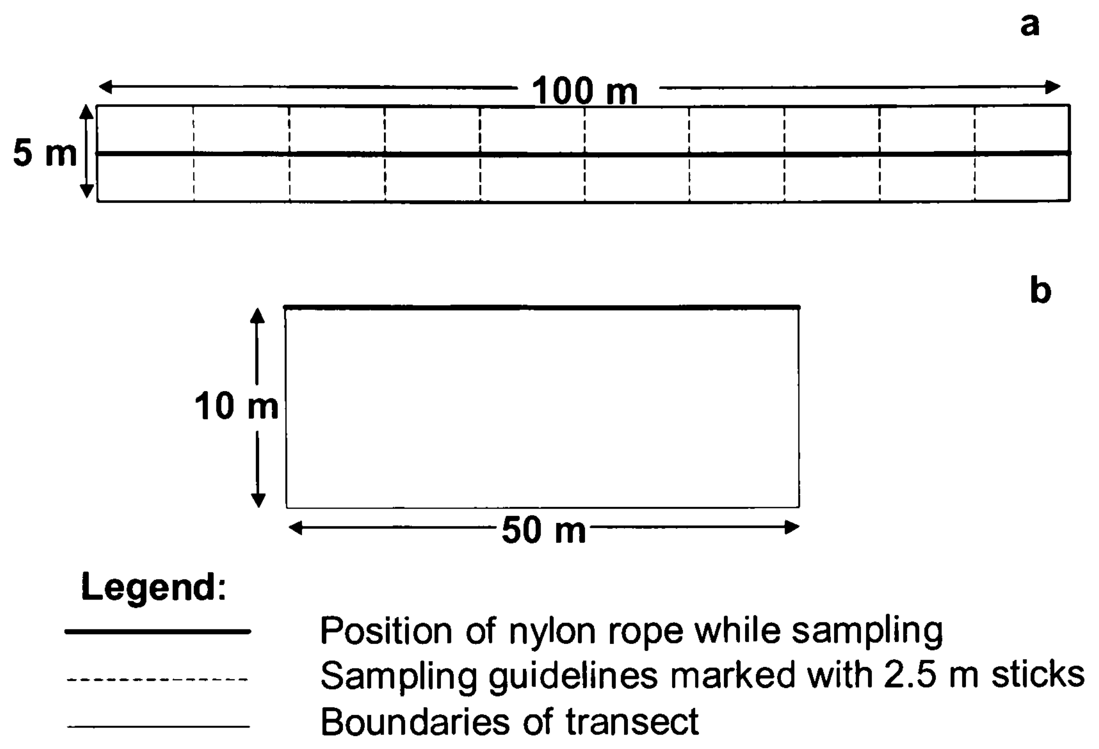

Figure 2.8 ***a****. Design of tree transects in sacred forests and the forest reserve;* ***b****. Design of tree transects in coffee plantations*

All trees ≥ 1 cm dbh in the belt transect were counted and identified by species. Pascal & Ramesh's (1987) field key was used for identification of trees, based on their morphological characteristics. Thus, tree species were identified as recognisable taxonomic units (Oliver & Beattie 1996). Daniels *et al.* (1995) and Utkarsh *et al.* (1998) have used this procedure for identification of trees in the Western Ghats. For all trees ≥ 10 cm dbh, the diameter was measured at the standard height of 130 cm from the base following the international convention. The diameters of buttressed trees were measured at least 30 cm above the buttresses. 1000 individuals ≥ 1 cm dbh is considered a sufficient sample size in tropical forests to carry out an analysis of species diversity (Condit *et al.* 1996). In each sampling locality at least 1100 individual trees ≥ 1cm dbh were enumerated, in order to correct during data processing, any possible discrepancies in counting. If the number of trees ≥ 10 cm dbh was less than 100 at a given locality, more trees that did not fall within transects were measured randomly until the count reached 100. This was done in order to allow comparison between samples from the forest reserve and sacred forests with those from coffee plantations, where only canopy trees were present and the shrub understorey formed by trees ≥ 1 cm dbh but < 10 cm dbh was absent.

Along each transect the trees ≥ 10 cm dbh were painted with a blue mark on the trunks. At each 10 m interval, the distance figures – 10, 20, 30, 40 m *etc.* – were marked in blue oil paint to enable a re-survey of transects in future. The number of lianas in each 10 m sector were recorded but not identified by species because they are poorly known; there are no exhaustive field keys allowing identification of infertile material and the available material on liana stems is often inaccessible.

Coffee plantations

The tree transects were established in a similar fashion in coffee plantations. The only difference was in their dimensions (see *Figure 2.8 b*). Although they were typically shorter (50 m instead of 100) because of the limited extent of the plantations, and wider (10 m instead of 5) because of lower tree density than elsewhere, the effective area sampled by a transect was the same. A nylon rope, 50 m in length was used to mark a line along the length of transect while sampling trees. Another 10 m rope was used to mark the width of transect and trees were sampled in the 50 x 10 m belt. Shade trees in coffee plantations are thinned periodically as the plantations mature. Therefore tree density is lower than that in sacred forests and the forest reserve. In localities where the extent was less than 50 m, the lengths of transects were as short as 20 m and were compensated by increasing the number of transects. However, the total number of trees ≥ 10cm dbh measured in each coffee plantation was at least 100 to allow comparison of diversity between all three land use types.

Birds

Several ways of sampling birds are recommended in the literature. Mist-netting (*e.g.* Stouffer & Bierregaard 1995; Restrepo *et al.* 1997) has recently become popular due to its ability in offering precision while counting individual birds and accuracy in estimating population sizes. However, it is an expensive and time-consuming method. Bibby *et al.* (1998) recommend line transects as the best way to sample birds that occur at low densities in fairly uniform habitats, while point counts are best for sampling species that are less obvious in forests and for working in places with a great variety of microhabitats.

A combination of transect and point count sampling was eventually used because of the limitations on the size of forest habitats in most sampling localities and the variety of microhabitats, such as small canopy gaps, tangles of lianas, reed clumps, *etc.* that affect visibility in the sampling localities. The fixed radius point count method described by Hutto *et al.* (1986) was employed. However, the emphasis in sampling was on the number of observations rather than the amount of time spent at each locality, as recommended by Thiollay (1994), who states that the samples with 50 observations of individual bird species at each locality allow reliable comparisons of bird diversity.

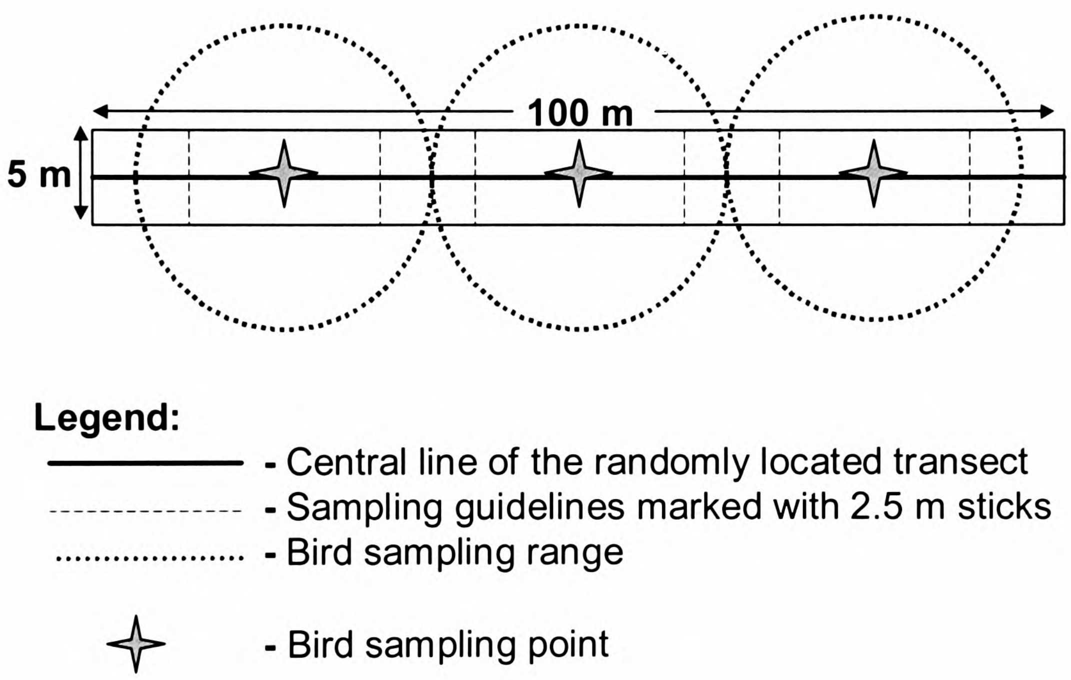

Figure 2.9 The design used for sampling birds – birds were observed at each sampling point for 12 minutes before proceeding to the next point in the sequence.

The framework of a base line and transects that was used for sampling trees, was employed for sampling birds (*Figure 2.9*). At each sampling locality, birds were observed, counted by species and recorded at points 20 – 30 m apart from each other on transects, depending upon the length of the transect and visibility. Each point count lasted for 12 minutes and the next three minutes were spent going to the next point before starting again. Canopy cover varied across land use types and the efficiency of sighting birds was uneven, hence the time for which birds had to be observed in order to encounter 50 individuals was variable. Sampling usually began at 7:00 a.m. because of relatively high bird activity at that time, and continued until the required numbers of observations were completed. All birds that were heard while sampling at a point were recorded but not counted because of the lack of information on the identity of species by bird call alone.

The bird sampling team consisted of at least three people in order to minimise errors in locating and counting birds. At each point, all the birds that were seen and observed through a pair of binoculars (7 x 50 magnification) were recorded by species. The species identification was based on Ali (1996) and the nomenclature was based on Grimmett *et al.* (1998).

Macrofungi

Vogt *et al.* (1992) have reviewed methods for sampling epigeous sporocarps of macrofungi. Various methods have been used including large permanent plots subdivided into quadrats (*e.g.* Ammirati *et al.* 1987), quadrats placed along a regular path (*e.g.* Murakami 1987) or line transects (*e.g.* Wilkins *et al.* 1937). The dimensions of sampling units vary from study to study. However, Parker-Rhodes (1951) has recommended that transects should not be wider than five metres since wider ones might increase the possibility of human error during observation and collection of sporocarps.

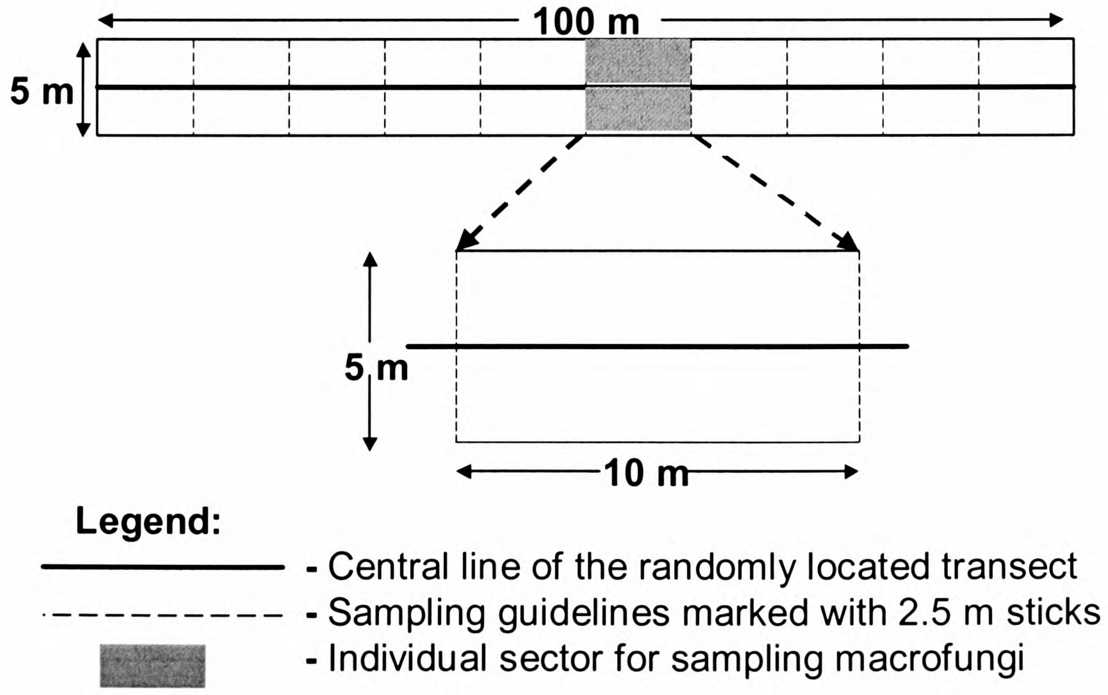

Figure 2.10 The design used for sampling of macrofungi – sporocarps were sampled at each locality in 10 x 5 m sectors on every transect.

The same framework of base lines and transects that was used for sampling trees was employed for sampling macro-fungi in the forest reserve and sacred forests (*Figure 2.10*). Senn-Irlet & Bieri (1999) estimate that a 500 m^2 plot size is adequate to get a reasonable amount of information on fungal diversity in forested habitats. Thus, at least 500 m^2 of area along the framework of transects in all localities, including coffee plantations, was sampled in order to collect fungal sporocarps. The sporocarps were collected on three different occasions at each locality – the beginning of the monsoon rains (June), the peak of monsoon rains (July) and the period of declining monsoon rains (August-September).

A team of at least four people, two on either side of the central line, walked along the transects in order to ensure that no macrofungal sporocarp within the transect belt remained un-noticed. In order to correct for the fact that some macrofungi produce clusters of sporocarps while others produce only single ones, each cluster belonging to a particular type was recorded as one observation irrespective of the number of sporocarps in the cluster. At least two sporocarps of each type were collected in paper bags and brought back to the laboratory. The sporocarps were photographed where it was possible. The morphological characteristics of each, necessary for the identification of morpho-species, were described and recorded in the laboratory and the sporocarps were cut for spore prints. Spore prints were taken on thin, transparent polythene sheets and preserved in polythene bags. The sporocarps were then oven dried and preserved as reference specimens in dried paper bags.

Family level identification was done in the field based on the classification system and colour charts used in Jordan (1995). Later the sporocarps were classified, according to the variation in their morphological characters, into several morphotypes or morphospecies. For instance, five sporocarps belonging to the same genus were classified as five morphospecies of that genus.

2.2.2 Measuring landscape structure

The strongly Euclidean[7] nature of man-made features in European and North American landscapes has focused efforts to quantify landscape geometry on individual patches, *e.g.* patch area, inter-patch distances and perimeter-area ratios (Milne 1988). However, in the Western Ghats, where the landscape has been modified by irregular and variable human intervention over several centuries, fractals[8] (Mandelbrot 1983) may more appropriately describe the landscape structure. The Euclidean parameters mentioned above may not be particularly relevant in the type of landscape investigated in this study where patches of irregular shapes, surrounded by a highly tree-covered landscape, occur. The measures of fractal geometry of the landscape, namely its composition and configuration may be more useful. The surrogate measures for various landscape characteristics (external variables) devised in this study are listed in table *Table 2.2*. These include commonly used Euclidean measures as well as less common non-Euclidean measures.

Table 2.2 The external variables and their ecological interpretation.

Variable	Description of landscape parameter	A measure of
DRF	Distance from the forest reserve	Isolation
AT	Percentage area of tree-covered land	Landscape integrity
NPT	Total number of patches within a buffer[9]	Landscape heterogeneity
LET	Total length of edges within a buffer	Landscape complexity

Note: AT, NPT and LET were measured for 250, 500, 750 and 1000 m buffers.

External variables: measures of landscape quality

Obtaining landscape information

The most recent and detailed information available about the study area is the forest map (Scale - 1:250,000) by Pascal *et al.* (1982). More detailed geographic information

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[7] Relating to the geometry, the study of angles and shapes, formed by the relationships between lines described by the ancient Greek thinker Euclid.

[8] A complicated irregular line or pattern in mathematics built from simple repeated shapes that are reduced in size every time they are repeated.

[9] A buffer is a zone of a given distance surrounding a patch.

is not available for the region, although it is possible to obtain old Survey of India topographic maps, as detailed as 1:25,000. Most maps are restricted to military use and difficult to obtain. Modern satellite images are available, although at a very high cost. Due to these difficulties, landscape information had to be obtained using a combination of locally available maps and field surveys. Processing and analysing this information required digitising hardware and Geographical Information System (GIS) software.

Village land survey maps

The study area consists of 42 villages with areas of 158 – 6867 ha. Land-survey maps (scale - 1:7920) prepared around 1900 were obtained from various local land record offices for each village. These maps have rarely been updated since 1900. The oldest one of the study area was prepared in the late nineteenth century and most others were from the early twentieth century. Lists of land survey numbers[10] and corresponding land-use types were obtained from the local land-survey offices. Although the land parcels (polygons) on these maps could still be located on the ground, many have been subdivided in the century since the maps were prepared. Fortunately, it was possible to determine positions of most sampling localities in sacred forests on the maps because, being a common property resource, sacred forests have not been subdivided. It was possible to locate paddy fields on the maps easily as they were marked by bordering trenches. There have been several land transformations in the last 50-150 years, but the non-tree-covered nature of the paddy fields remains the same (Moppert 2000). Most areas that are neither sacred forests nor paddy fields are under tree-covered land-uses, such as coffee plantations. After the initial conversion of forest into coffee plantations between the mid 19th and mid 20th centuries, little change seems to have occurred in terms of the proportion of tree-covered land, although the forest understorey was planted with coffee. Elouard (2000b) also estimates that open areas such as paddy fields, without any tree-cover, have increased by only 3% during this period. Thus, although the available information about land-parcels is old, these maps can be reliably used after ground-checking for changes that might have taken place. The boundary of the government-owned forest was obtained as marked on a recent (1998) map (scale – 1:126,720) (Bar Association Virajpet 1998). Other landscape features such as cart

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[10] Land survey number is a number by which a land parcel (polygon) is identified on the map.

tracks, streams and rivers were marked but it was not possible to plot more recent developments such as modern roads and habitations on the available village maps. However, they occupy an insignificant proportion (≤ 1%) of the total geographical extent of the study area (personal estimate).

Global Positioning System (GPS) surveys

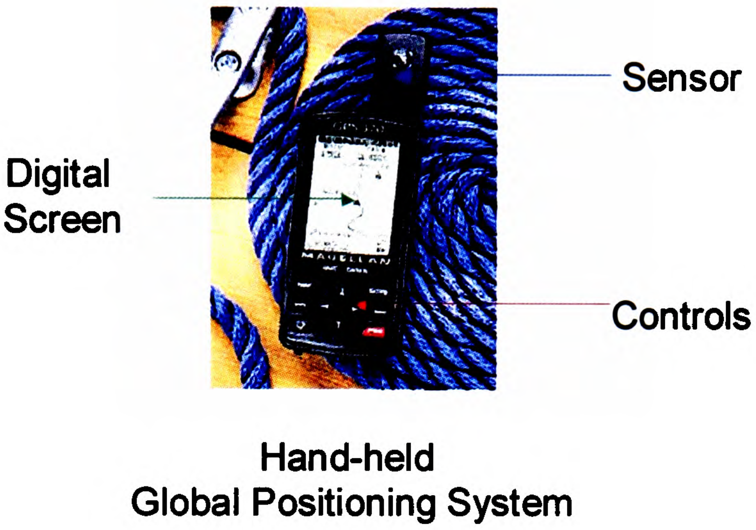

Figure 2.11 A hand-held Global Positioning System used for finding co-ordinates of reference points useful in digitising land-survey maps.

A hand-held Magellan 310 GPS (Magellan Corporation 2000) (*Figure 2.11*) was used to obtain geographical co-ordinates and elevations (at approximately 20 m accuracy) of the points that were located on the maps. Such information was obtained for at least four points for a map (scale: 1:7920) that was less than A2 size. Maps were divided into sections ≤ A2 size (457 mm X 610 mm) for the ease of digitising. In each such section at least four prominent point features were identified. These were then visited on the ground and their geographical co-ordinates measured using a GPS. This was important for establishing accurate control points on each map section. Other landscape features were then digitised in relation to these control points.

Digitising land-survey maps

The land-survey maps were digitised using a CalComp DrawingBoard III digitising tablet (GTCO CalComp Inc. 2000) with a 457 mm X 610 mm active surface area. All maps were digitised in MapInfo (MapInfo Corporation 2001c) format. MapInfo version 5 was used for digitising and subsequent analyses. Landscape diversity is linked to geographic information contained in the landscape. Geographical Information Systems provide an important method for assessment, management and monitoring of landscapes (Parresol & McCollum 1997). MapInfo is one such tool where layers of information can be combined in various ways to generate data that can be analysed to make conservation management decisions.

All 42 digitised maps were saved as one layer in a MapInfo table. The map of the study area had 353 polygons – 160 sacred forests, 192 paddy fields and one forest reserve. The rest of the area was occupied by other tree-covered land, predominantly coffee plantations. Landscape features such as roads and houses that were not marked on the maps, were estimated to occupy about 1% of the total area and were not included in the area occupied by tree-covered land. *Table 2.3* shows the number of polygons of each land-use type and the areas occupied by them, calculated in MapInfo.

Table 2.3 Land use types and the areas occupied by them in the study area.

Land use type	Number of polygons	Total area (ha)	Percentage %
Coffee plantations	-	35493	59
Forest reserve	1	18890	31
Paddy fields	192	4720	8
Sacred forest	160	531	1
Miscellaneous	-	605	1
Total	353	60240	100

The centroids[11] of each of the 58 sampling localities used in this study for sampling biodiversity (23 coffee plantations, 10 areas within the forest reserve and 25 sacred forests) were marked on the map (*Figure 2.12*).

Figure 2.12 (Please see Page 59) Landscape map prepared in MapInfo after digitising 42 village land survey maps (scale 1:7920) in one layer. The sampling localities are numbered according to the list in Table 2.1

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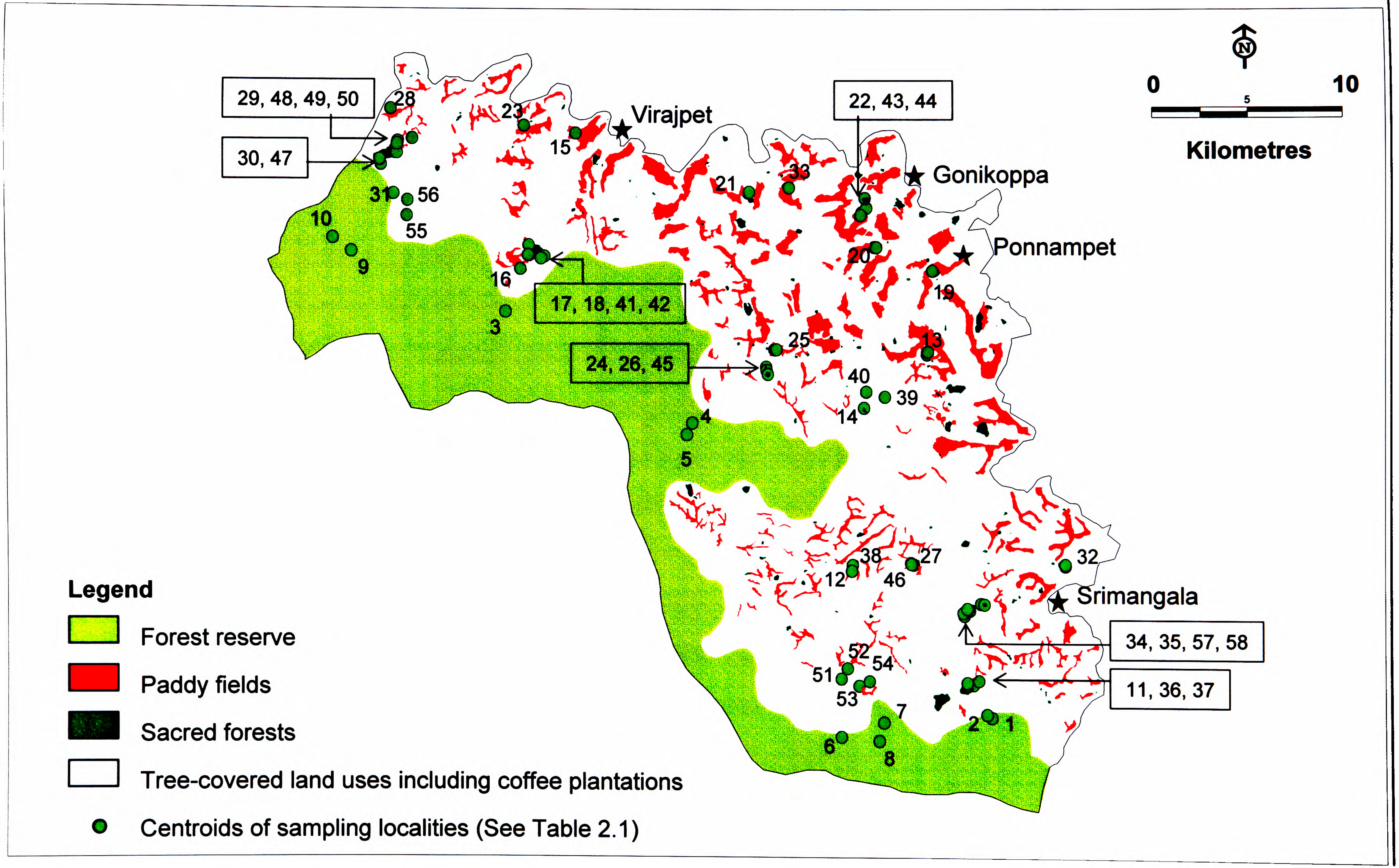
[11] A centroid is the point that is at the greatest distance from all the edges of a polygon.


29, 48, 49, 50
30, 47
28
23
15
Virajpet
31
56
55
10
9
16
17, 18, 41, 42
3
21
33
22, 43, 44
Gonikoppa
20
Ponnampet
19
25
24, 26, 45
13
40
39
14
4
5
0
5
10
Kilometres
N
38
27
12
46
32
Srimangala
34, 35, 57, 58
52
54
51
53
11, 36, 37
7
2
1
6
8
Legend
🟩 Forest reserve
🟥 Paddy fields
■ Sacred forests
□ Tree-covered land uses including coffee plantations
● Centroids of sampling localities (See Table 2.1)

Patch area

It was possible to measure patch areas for sacred forest patches, but not for coffee plantations because there were no well-defined ecological boundaries between coffee plantations – they occupied most of the landscape (nearly 60%) between sacred forest patches.

Isolation

As a measure of isolation of a given patch (*i.e.* the distance of centroid of each sampling locality from the nearest edge of the forest reserve), DRF (*Table 2.2*) was measured[12] (MapInfo Corporation 2001b). The distances of sampling localities within the forest reserve were considered zero.

Intervening landscape matrix

In order to measure characteristics of the landscape surrounding patches, concentric circular buffers[13] of 250, 500, 750 and 1000 m radii were created around the centroid of each sampling locality. Four different buffer sizes were chosen in order to address the differences in the 'scale' of dispersal ability of the groups of organisms in question. Previous landscape studies *e.g.* Opdam *et al.* (1985) for birds, Luiselli & Capizzi (1997) for reptiles, Bowman *et al.* (2001) for small mammals, have found that characteristics of the intervening landscape matrix up to 1 km from patches can influence diversity. In the absence of any such information for trees and macrofungi, which are immobile and hence poorer dispersers than reptiles, birds or small mammals, it was assumed that the intervening matrix within 1 km distance will be sufficient for explaining the patterns of their diversity.

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[12] The 'Distance to Objects' MapBasic utility in MapInfo was used to measure distances.

[13] A buffer is a zone of a given distance and can be generated around map objects.

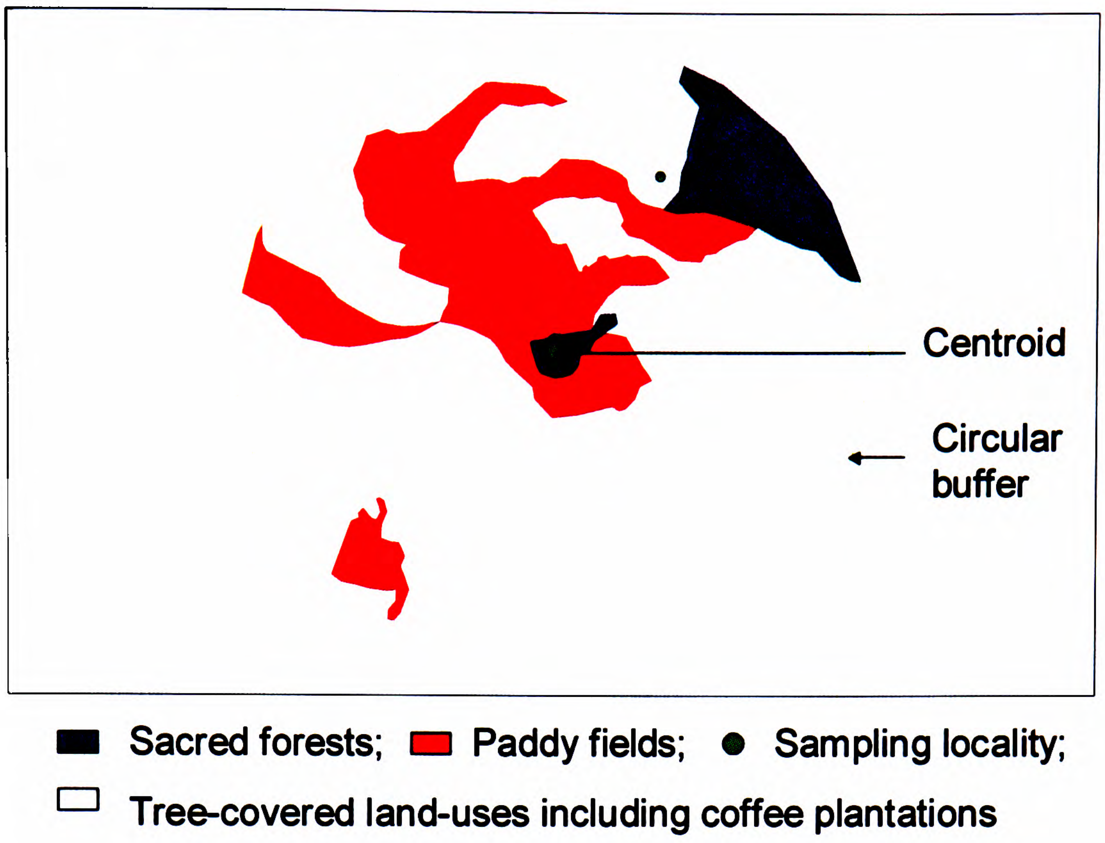

Figure 2.13 An example of a circular buffer used for measurement of landscape structure. Four circular buffers at 250, 500, 750 and 1000 m around the centroid of each sampling locality were created.

All the landscape inside the region created by the buffer was selected[14] (MapInfo Corporation 2001a). Thus, the area enclosed by circular buffers contained all patches (in part or whole) that were within the buffer boundary. Each buffer and the area within it was saved as a different file (thus, resulting in four files for each sampling locality) in order to carry out analyses of the landscape surrounding the centroid of each sampling locality (*Figure 2.13*).

For all four buffer sizes around the centroid of each patch, three landscape variables (except DRF, which was measured using a different method – see *p. 60*) listed in *Table 2.2* were measured:

- *Measure of landscape integrity*: Percentage of tree-covered land was used as an indicator of integrity of the surrounding landscape.

- *Measure of landscape heterogeneity*: The total number of patches of sacred forests and paddy within a buffer were used as indicators of fragmentation that has

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[14] The 'cookie cutter' utility in MapInfo was used.

occurred in the surrounding landscape. A high number of patches was considered to imply greater fragmentation.

- *Measure of landscape complexity*: The total length of edges was used as another indicator of fragmentation. A high cumulative total length of edges of all the patches inside a buffer was considered to correspond with a higher degree of fragmentation of the surrounding landscape.

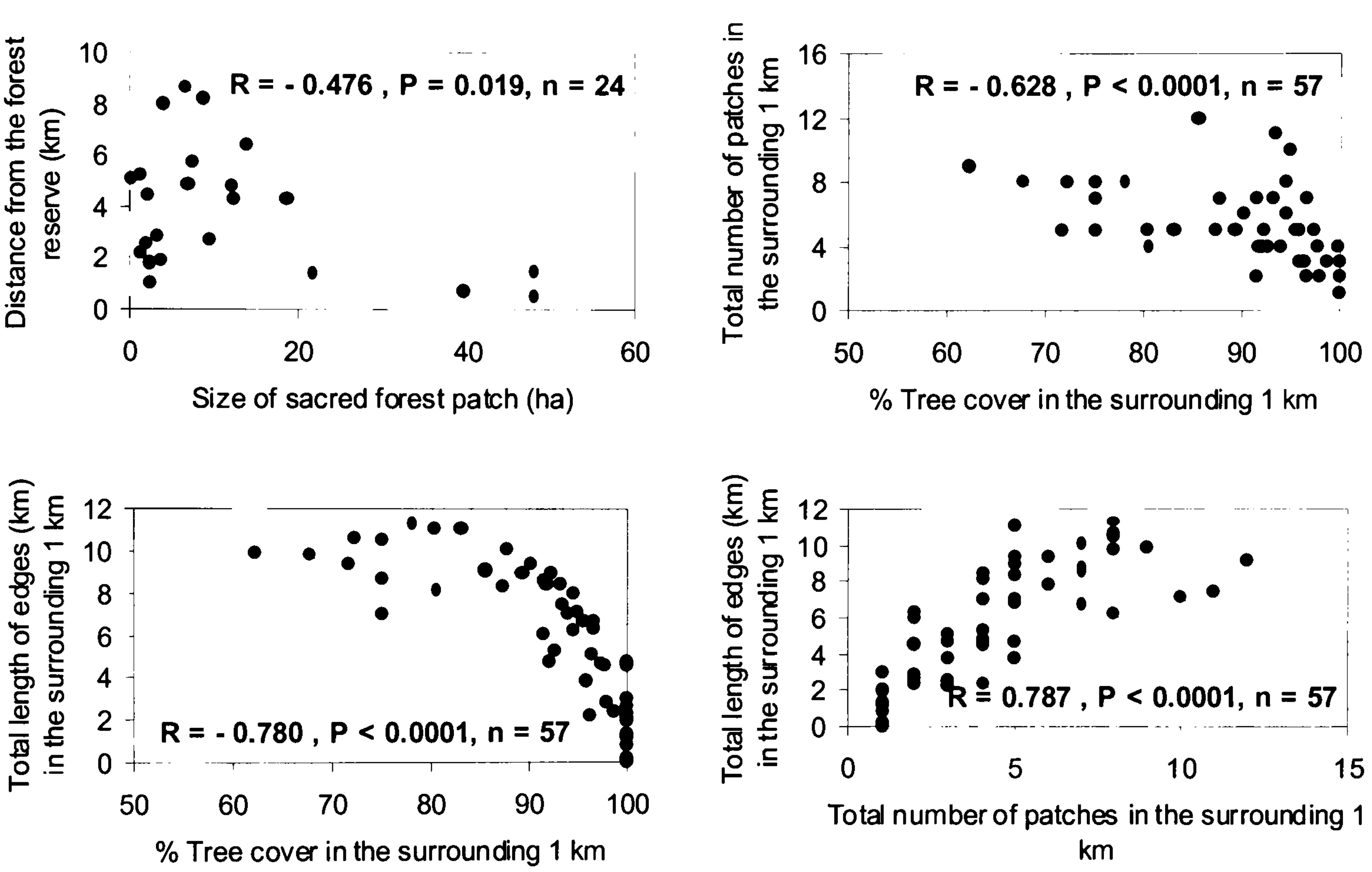

Figure 2.14 Correlations between landscape variables used in the present study; the top-left panel shows the relationship between sizes of 24 sacred forest patches and their isolation from the forest reserve (one outlier is removed); the remaining three panels show data for 57 sampling localities that include forest reserve localities, sacred forests and coffee plantations (one sacred forest outlier is removed).

There was a significant negative correlation between the size of sacred forest patches and their isolation from the forest reserve (*Figure 2.14*). While landscape integrity in the surrounding 1 km was negatively correlated with landscape complexity as well as landscape heterogeneity, landscape complexity and heterogeneity were significantly positively correlated (*Figure 2.14*).

In addition to the landscape characteristics, various habitat characteristics, such as structural diversity, heterogeneity and complexity of forest within a patch (internal variables) were also measured. They are listed in *Table 2.4*.

Table 2.4 The internal variables and their ecological interpretation.

Variable	**Description of structural parameter**	**A measure of**
SD	Number of stems ≥ 10cm dbh per hectare	Forest structural complexity
BAR	Basal area range (5-95 percentiles) in sq. m	Forest structural heterogeneity
HTR	Canopy height range (5-95 percentiles) in m	Extent of disturbance to the ecosystem
CCR	Canopy-scope[15] measurement range (5-95 percentiles)	Micro-habitat heterogeneity in the canopy
LIA	Number of lianas per hectare	Micro-habitat complexity in the understorey

Internal variables: measures of habitat quality

Although structural diversity is an important component of the overall forest diversity, it is seldom defined in objective terms (Brockelman 1998). Structural variables are important in the analysis and management of forest ecosystems (Zenner & Hibbs 2000). Structural characteristics have also been used to define niche requirements of wildlife species (*e.g.* MacArthur & MacArthur 1961) and are important in biodiversity studies.

In the present study, structural variables that determine heterogeneity and complexity of the forest stand were measured along the vegetation transects in all three land-use types. Five internal variables (*Table 2.4*) were measured at each sampling locality:

Stem density per hectare: It was used as a measure of forest structural complexity.

Basal area per hectare: Forest structural heterogeneity was determined by the variability in basal area of trees in a given forest stand. Basal area for each tree was calculated using the formula $BA = \pi d^2/4$. The BAs of individual trees were added to

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[15] The 'Moosehorn' (Garrison 1949) was re-designed as a transparent Perspex screen with a 20 cm cord attached to one corner. The cord is used to ensure that the screen is always held at the same distance from

calculate the total BA per hectare for each sampling locality. If there was a wide range of basal areas of stems ≥ 10 cm dbh, then the site was considered to have higher structural heterogeneity.

Canopy closure: Every 10 m along transects (see *Figure 2.8*), canopy closure was measured using a canopy scope (Brown *et al.* 2000). This technique provides a rapid, cheap and effective way of measuring canopy closure in relatively closed forest and the results correlate with more accurate, but time-consuming and expensive methods such as hemispherical photography (Whitmore *et al.* 1993; Brown *et al.* 2000). The variability in canopy-scope measurements was used to indicate the variability in canopy gap-sizes within a stand. This information was used as a measure of heterogeneity in the forest canopy.

Canopy height: The variability in canopy heights was used to determine quantity and quality of past disturbance episodes that might have taken place in a given stand. It was used as a surrogate measure for the extent of disturbance to the ecosystem. The canopy height was measured every 10 m along the transect using a SUUNTO clinometer.

Density of lianas: The density of lianas is strongly correlated with disturbance, since greater penetration of light into disturbed forest patches allows more rapid growth of climbers and gives them a competitive advantage over tree saplings (Putz 1991, cited in Turner *et al.* 1996). The density of lianas was measured by counting the number of stems in each transect and estimating their numbers per hectare. Lianas often entangle in disturbed forest understories and form impenetrable layers, which can provide various micro-habitats for wildlife. The density of lianas was therefore used as a measure of micro-habitat complexity in the forest understorey.

Monitoring microclimatic conditions

Microclimate monitoring devices recording temperature and relative humidity, referred to as data loggers (*Figure 2.8*) (Onset Computer Corporation 1996-2002b) were set up

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the eye. The screen was engraved with 25 dots, approximately 1 mm in diameter spaced 3 cm apart (centre to centre), in a 5 x 5 – square array. This instrument was re-named as a canopy-scope.

(usually tied to a tree trunk or to an appropriate object about 10 m from the ground with a nylon thread) at six selected localities (since only six microclimate monitoring devices were available). The point chosen for setting up a data logger in sacred forests was one of the 'best' points on the transect network with the most closed canopy in the locality in order to avoid the effects of external factors related to patch edges or canopy openings (*e.g.* Brown 1993). Data loggers were also set up simultaneously in coffee plantations in order to compare the differences in temperature and relative humidity inside the forest and in the plantation during the given period. The devices were kept in the same place for about a week. The data were downloaded and stored in a laptop computer using Box Car Pro software (Onset Computer Corporation 1996-2002a).

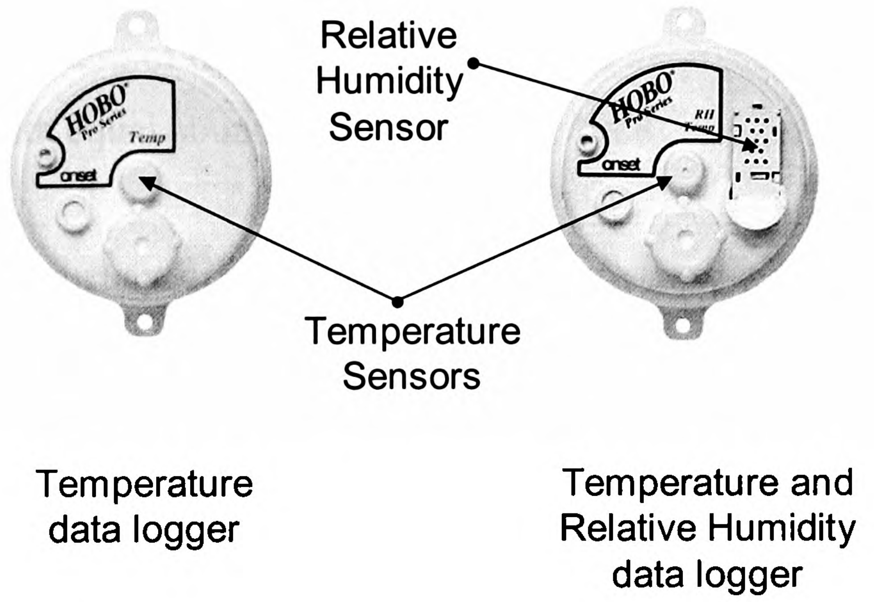

Figure 2.15 Microclimatic data loggers used for monitoring temperature and relative humidity at selected sampling localities.

During the monsoon season in 1999 when the sampling of macrofungi was carried out, the equipment was set up at four localities across the study area (usually in a sheltered place, in order to avoid the saturation of the humidity sensors with rain water). The devices were set up for two calendar months. Rainfall was also recorded at the respective places, during the period in which the data loggers were set up. This information was useful for verifying the correlation between rainfall and relative humidity. Based on the information on temperature and relative humidity, the data logger device is able to calculate dew point[16] (Onset Computer Corporation 1996-2002b). It was intended that this would provide an important variable (moisture content of the air) in relation to sporocarp production in macrofungi.

2.3 Data analyses and interpretation

2.3.1 Univariate analyses

Rarefied species richness

Investigations of biological diversity are often restricted to species richness – a straightforward count of the number of species present. However, relative abundance of species, another component of diversity, is of interest to ecologists since no community consists of species with equal abundances (Magurran 1988).

Rarefaction of the frequency distribution of individuals over species allowed reliable comparisons of diversity (Heck *et al.* 1975; Gotelli & Colwell 2001). Hurlbert's (1971) rarefaction method was employed to calculate the expected number of species in a sample of 'n' individuals selected at random. To allow comparisons across samples, each sample was rarefied to the maximum value that permitted calculation of rarefied species richness for all samples. This number, for trees ≥ 1 cm dbh was 650 individuals, for trees ≥ 10 cm dbh was 50 individuals, for birds it was 15 observations and for macrofungi it was 12 observations. The characteristics of the species-abundance distributions of some sites meant that the rarefied value could not be computed for a larger 'n' although 1100 trees ≥ 1 cm dbh and 100 trees ≥ 10 cm dbh were sampled and 50 observations of birds and macrofungi each were made at each sampling locality.

There was significant correlation between rarefied species richness for trees (≥10 cm dbh) calculated for 50 individuals and number of species (species richness) (*Figure 2.16*). The rarefied species richness was also correlated with four other diversity indices commonly used in ecological studies (Magurran 1988) (*Figure 2.17*).

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[16] Dew point is a measure of how much water vapour is actually in the air. It is defined as the temperature at which saturation occurs if air is cooled at constant pressure without addition or removal of vapour

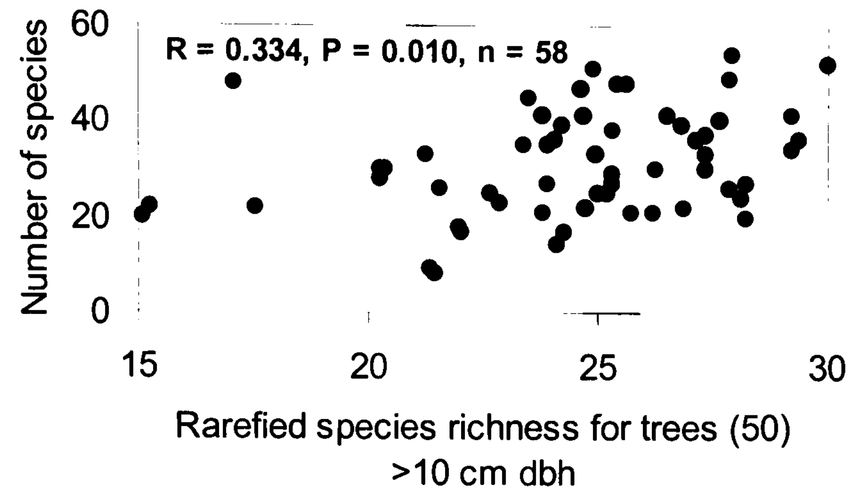


Figure 2.16 Correlation between rarefied species richness for trees (≥ 10 cm dbh) and number of tree (≥ 10 cm dbh) species (species richness), number of sampling localities, n = 58.

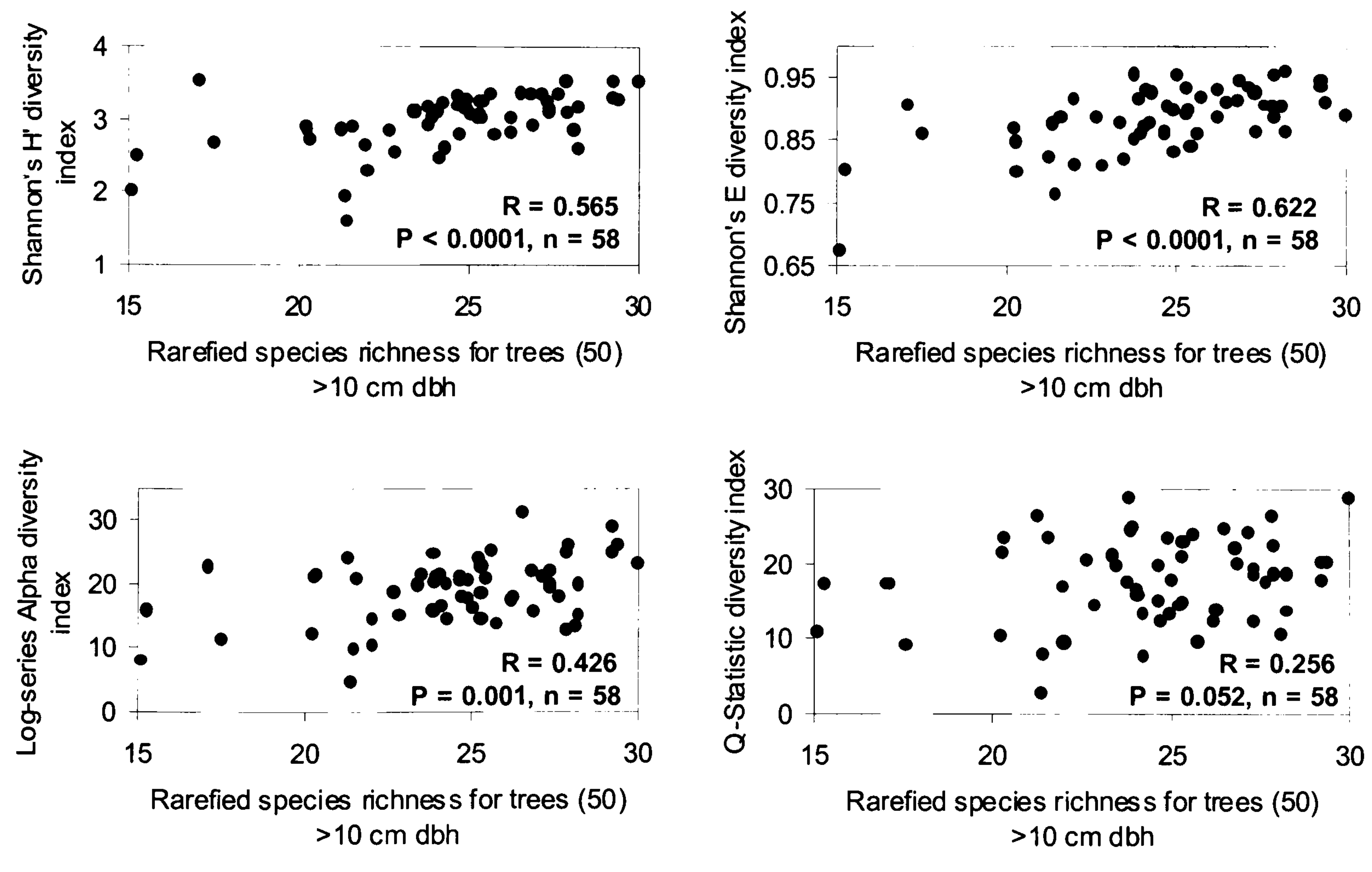


Figure 2.17 Correlations between rarefied species richness of trees (≥ 10 cm dbh) calculated for 50 individuals and four other common diversity indices (Magurran 1988); number of sampling localities, n = 58.

(Barry & Chorley 1987, cited in Glassy & Running 1994).

Mean rarefied species richness for each land-use type – forest reserve, sacred forests and coffee plantations – was calculated for each group of organisms and compared with the other types. One-way analysis of variance (ANOVA), which allows testing of the hypothesis that several means are equal, was used to determine whether the means differed significantly.

Rank abundance curves

In ecological communities some species are common while others are rare. Species can be arranged along a spectrum of abundance, from the most common to the most rare (Tokeshi 1993). The community structure thus arising can be represented in a graphical form using rank abundance curves, where species ranks (based on their abundances in a community) are plotted on the X-axis while the corresponding abundances of species are plotted on Y-axis. Comparisons between the slopes of such curves provide a valuable method for comparing diversity of two or more ecological communities.

Sorenson's quantitative similarity index

In order to compare diversity between a patchy and a contiguous landscape, similarities between the forest reserve and sacred forests at various distances from the reserve, were computed using Sorenson's quantitative similarity index (Magurran 1988). Quantitative modification of the index was used in order to minimise the effect of rare species or occasional observations. Sorenson's quantitative index (C_N) was calculated as:

$$C_N = \frac{2_{jN}}{(_{aN} + _{bN})}$$

Where, $_{aN}$ is the total number of individuals or observations in the first community; $_{bN}$ is the total number of individuals in the second community; and $_{jN}$ is the sum of the lower of the two abundances recorded for species found in both communities.

The forest reserve had 8 localities. There were 9 sacred forest patches at less than 1 km from the reserve, 7 were at 1–1.5 km from the reserve and 8 patches were more than 1.5 km from the reserve. In each of these localities, species were ranked according to their abundances. In each distance band (including the forest reserve) average abundance for the species in each rank was calculated. Sorenson's similarity indices were calculated for four distance bands based on the average rank-abundance distributions of sampling localities contained in the respective bands.

Statistical tests

Chi-squared test

The chi-squared (χ^2) test procedure tabulates a variable into categories and computes a chi-square statistic. This contingency test compares the observed and expected frequencies in each category to test either that all categories contain the same counts or that each category contains a user-specified number of counts (SPSS Inc. 1989-1999).

The χ^2 test was used in order to examine whether the differences in the frequencies of unique species of trees, birds or macrofungi (adjusted to sample size) were significantly different across the three land use types.

Kolmogorov-Smirnov test

The Kolmogorov-Smirnov Z test is a general test that detects differences in both the locations and the shapes of the distributions. The Kolmogorov-Smirnov test is based on the maximum absolute difference between the observed cumulative distribution functions for two samples. When this difference is significantly large, the two distributions are considered different (SPSS Inc. 1989-1999).

In order to examine whether the distribution of tree stems in various size classes in one tree species differed significantly from that in another, the Kolmogorov-Smirnov test was used. The test was also useful in examination of normality in distribution of the variables measured (see *Table 2.2, Table 2.4*).

Kruskal-Wallis and Mann-Whitney tests

The Kruskal-Wallis H test, is the nonparametric equivalent of one-way analysis of variance (used to test the hypothesis that several means are equal) and detects differences in distribution of more than two samples. It assumes that there is no *a priori* ordering of the populations from which the samples are drawn (SPSS Inc. 1989-1999).

The Mann-Whitney U test is equivalent to the Kruskal-Wallis test for two groups. It tests whether two sampled populations are equivalent in distribution. The observations from both groups are combined and ranked, with an average rank assigned in the case of ties. The number of times a score from group 1 precedes a score from group 2, and the number of times a score from group 2 precedes a score from group 1 are calculated. The Mann-Whitney U statistic is the smaller of these two numbers (SPSS Inc. 1989-1999).

The Kruskal-Wallis test was used in order to test the differences in composition of species groups across coffee plantations, sacred forests and the forest reserve. The Mann-Whitney test was used in order to compare the differences in forest structural variables (*Table 2.4*) in the forest reserve and sacred forests.

Mantel's test

Mantel's test (Mantel 1967; Mantel & Valand 1970) is a widely used method for assessing the relationships between two distance matrices. It is classically used to compare genetic and geographical distances in population genetic studies, but it has also been used in ecology and there are a growing number of publications referring to this test in the literature (Legendre 2000). This test involves measuring the association between the elements in two matrices by a suitable statistic, and then assessing the significance of this statistic by comparison with the distribution found by randomly reallocating the order of the elements in one of the matrices (Bonnet 2001). The Mantel's test evaluates the null hypothesis of no relationship between two similarity or dissimilarity matrices.

In order to compare the similarity matrices for distributions of trees, birds and macrofungi, a simple Mantel's test for two matrices was performed using PCORDWIN software (McCune & Mefford 1999). Proxscal Version 1.0 (Data Theory Scaling System Group (DTSS), Faculty of Social and Behavioral Sciences, Leiden University, The Netherlands) was used. Pair-wise comparisons were made across the three groups of organisms. The standardised Mantel Statistic (R) was used as a measure of the strength of the relationship between two distance matrices. R ranges between –1 and 1. ‘T’ indicates the type of association (positive or negative) between distributions using an asymptotic approximation of Mantel (1967). If $T > 0$, the association is positive and if $T < 0$, the association is negative (McCune & Mefford 1999).

2.3.2 Multivariate analyses

Non-metric multidimensional scaling

Non-metric multidimensional scaling attempts to find the structure in a set of community similarity measures. This is accomplished by assigning communities (represented by points) to specific locations in a conceptual low-dimensional space such that the distances between points in the space match the given similarities as closely as possible. The result is a least-squared representation of the communities, which helps to understand the structure of the data in question (SPSS Inc. 1989-1999). Kenkel & Orloci (1986) have shown the use of non-metric multidimensional scaling in ecological studies.

In order to examine the similarity between the forest reserve and sacred forest patches at various distances from the reserve, the average rank-abundance distributions in four distance bands were subjected to non-metric multidimensional scaling. Similar analyses were carried out for rank-abundance distributions in individual sacred forest patches and the localities in the forest reserve. Euclidean distance measure was used in order to create similarity and dissimilarity matrices. The results were evaluated based on the Standardised Residual Sum of Squares (STRESS) measure.

The multivariate analytical methods used to investigate which of the external and internal variables were important in accounting for the patterns of diversity in different groups of organisms are described in the following section.

Decision trees

The Kolmogorov-Smirnov test for normality in distribution revealed that most landscape and habitat variables were not normally distributed. In addition, the landscape variables showed characteristics such as multicollinearity (*Figure 2.14, Figure 2.18*) (due to the effect of overlapping buffers on the measurement of landscape structure around the centroids of sampling localities) that can make the data unsuitable for parametric techniques (Stärk & Pfeiffer 1999). Due to the lack of theory in landscape ecology and due to the lack of specific hypothesis as far as the effects of various landscape variables on biodiversity are concerned, exploratory data-driven modelling methods such as cluster analysis, neural networks and decision trees were thought to be more suitable than theory-driven modelling methods such as GLM (General Linear Models), discriminant analysis and multinomial regressions.

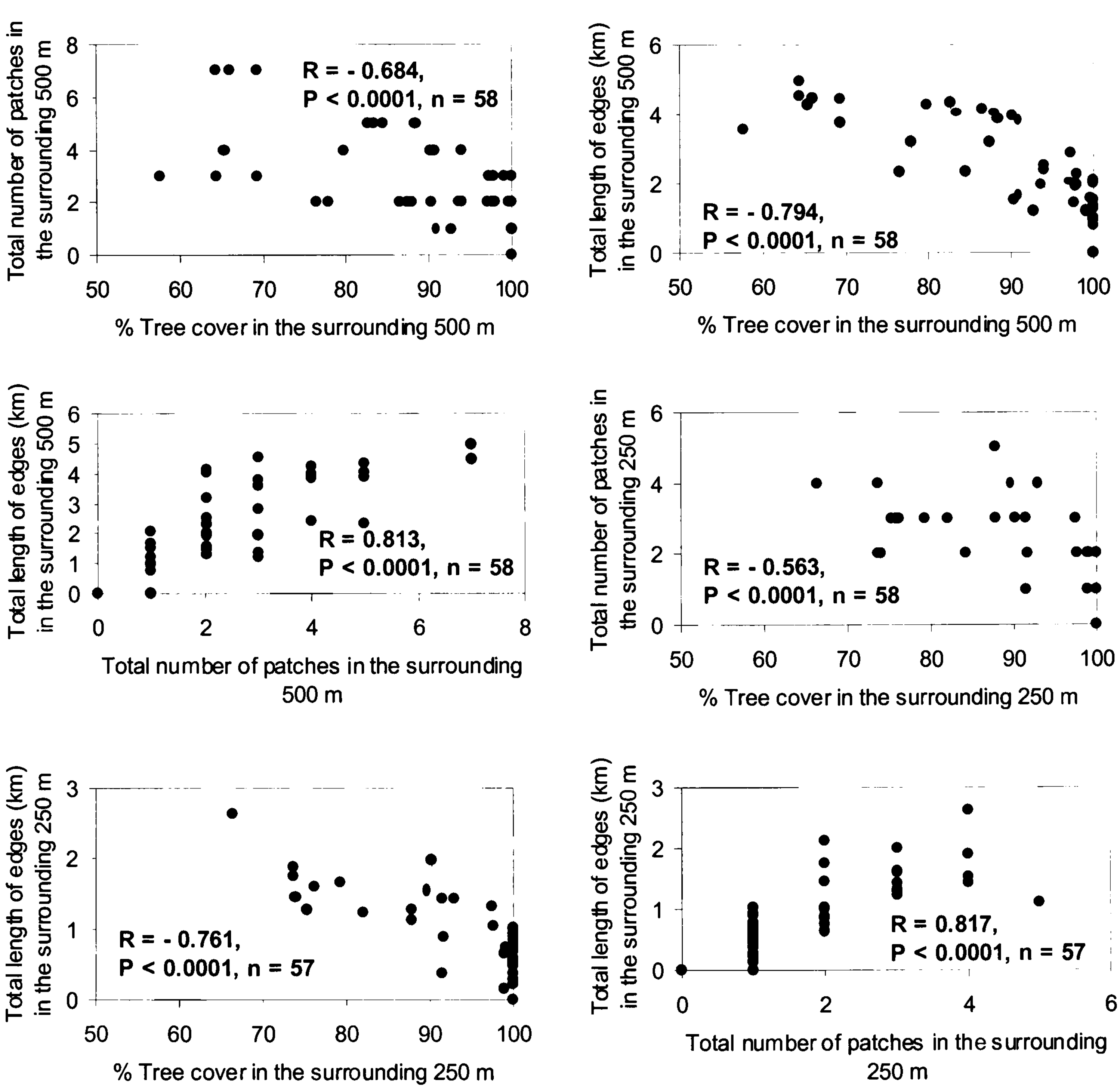


Figure 2.18 Multicollinearity between landscape variables measured in the present study; there were strong correlations among variables listed in Table 2.2, measured within buffer sizes of 500 and 250 m.

In addition to these reasons, it was important that the model developed was easily interpretable and had practical utility for recommending land management alternatives for biodiversity conservation. Decision trees (Breiman *et al.* 1984) were found suitable for building a model that can explain the patterns of diversity in different groups of organisms. This interactive method of building multiple regression models was used because variable selection is best accomplished when models are screened by an analyst who is familiar with the data and subject matter – *e.g.* Henderson & Velleman (1981); James & McCulloch (1990, cited in Miller & Cale 2000).

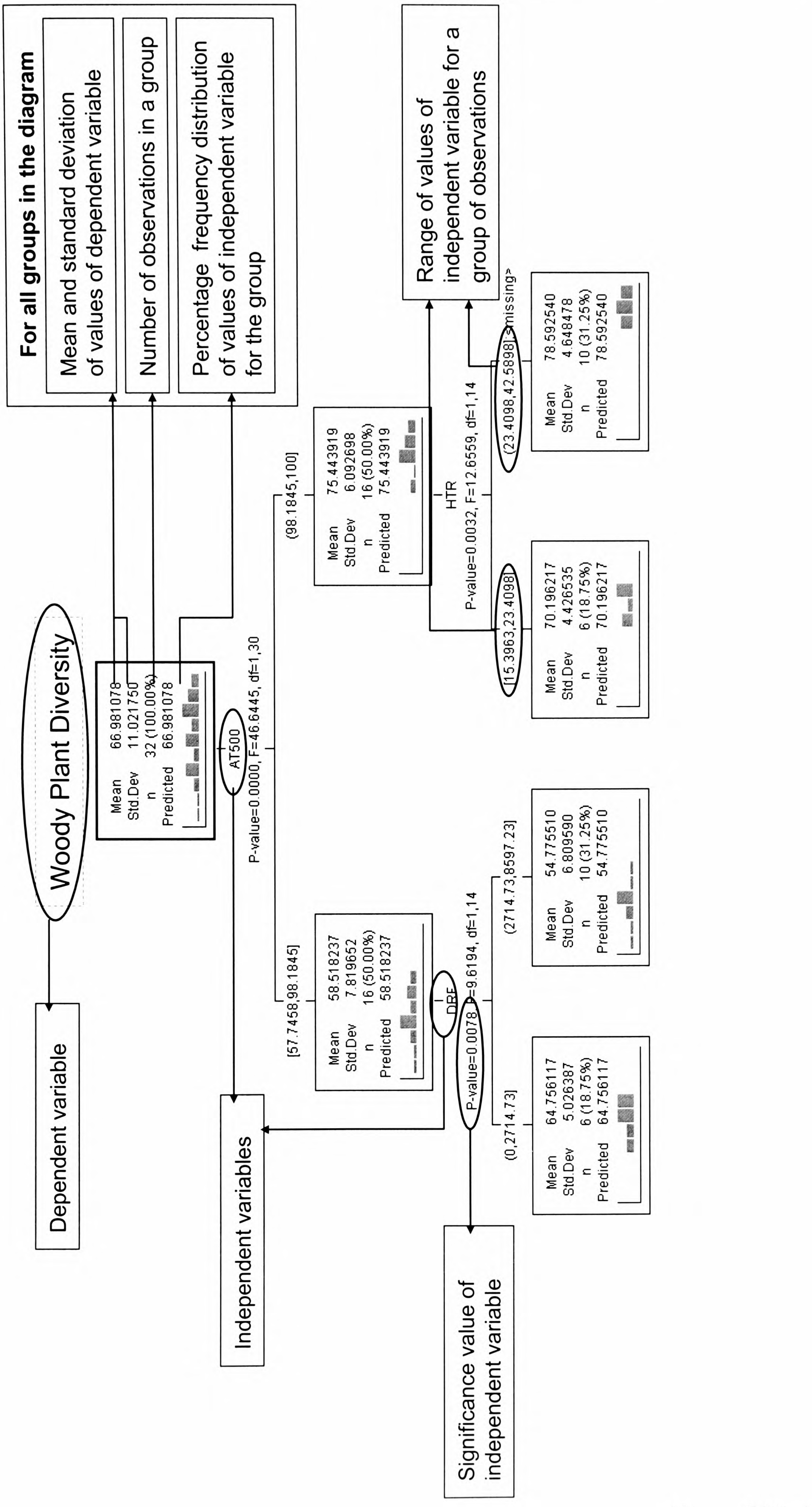
Decision trees are based on the principle of recursive partitioning, which means that the data set is split, based on the values of explanatory variables (Breiman *et al.* 1984). This technique has been in use since the development of advanced computing technology. The decision trees can be used to predict and classify data that are related in complex ways and cannot easily be analysed using traditional statistical methods. They are ideally suited for analysing complex ecological data, which require a flexible and robust method that can deal with non-linear relationships, high order interactions and missing values (*e.g.* Schröder *et al.* 1992; Rejwan *et al.* 1999; Déath & Fabricius 2000).

CHAID (CHi squared Automatic Interaction Detection) modelling is an exploratory data analysis method used to study the relationships between a dependent measure and a large series of possible predictor variables that themselves may interact. Kass (1980) first introduced CHAID as a way for classifying a categorical dependent variable based on a number of predictors. Subsequently, this method was extended to include continuous dependent and predictor variables and a superior method, Exhaustive CHAID, evolved (Biggs *et al.* 1991). CHAID has been extensively used as data-mining technique in business and marketing (*e.g.* Erdilek & Wolf 1997), management (*e.g.* Godley *et al.* 1998), medicine (*e.g.* Rakowski & Clark 1998) and veterinary science (*e.g.* Stärk & Pfeiffer 1999). However, it remains little explored in natural resource-based sciences such as forestry - a search for the term 'CHAID' using the Tree CD bibliography database updated to March 2002 produced only one result: Schröder *et al.* (1992). However, in various other studies the technique was found to be advantageous for analysing complex ecological datasets for the following reasons (Huba 2000):

- The level of measurement for the dependent and predictor variables can be nominal, ordinal or continuous. For example, in the present study, land use type was a nominal variable while rarefied species richness was a continuous variable.
- Not all predictors need to be measured at the same scale. For instance, in the present study, landscape composition and configuration was measured at different scales – within buffers of four different sizes.
- The analysis is not affected by missing values and partial data can be used whenever necessary. In the present study, especially while assessing proportions of tree cover, in many cases the values for small buffer sizes (of 250 and 500 m radius) were 100% due to the high tree cover in the landscape, while the number of patches within the buffer were zero. Thus, the data suffered from missing values at both ends.

According to Huba (2000) CHAID analysis has certain limitations because it is a stepwise model fitting method. The sequential model-fitting algorithm means that the later effects are dependent on the earlier ones unlike in ANOVA or GLM, where all effects are fitted simultaneously. Nonetheless, in areas such as landscape ecology, where there is a lack of strong theory to indicate clearly which variables are or are not predictors of particular independent measures, CHAID is very useful in identifying major data trends.

Figure 2.19 (Please see Page 75) The interpretation of a hypothetical CHAID diagram.


Dependent variable
Woody Plant Diversity
Independent variables
Significance value of independent variable
For all groups in the diagram
Mean and standard deviation of values of dependent variable
Number of observations in a group
Percentage frequency distribution of values of independent variable for the group
Range of values of independent variable for a group of observations
Mean 66.981078
Std.Dev 11.021750
n 32 (100.00%)
Predicted 66.981078
AT500
P-value=0.0000, F=46.6445, df=1,30
[57.7458,98.1845]
(98.1845,100]
Mean 58.518237
Std.Dev 7.819652
n 16 (50.00%)
Predicted 58.518237
Mean 75.443919
Std.Dev 6.092698
n 16 (50.00%)
Predicted 75.443919
DRF
P-value=0.0078, F=9.6194, df=1,14
[0,2714.73]
(2714.73,8597.23]
Mean 64.756117
Std.Dev 5.026387
n 6 (18.75%)
Predicted 64.756117
Mean 54.775510
Std.Dev 6.809590
n 10 (31.25%)
Predicted 54.775510
HTR
P-value=0.0032, F=12.6559, df=1,14
[15.3963,23.4098]
(23.4098,42.5898],<missing>
Mean 70.196217
Std.Dev 4.426535
n 6 (18.75%)
Predicted 70.196217
Mean 78.592540
Std.Dev 4.648478
n 10 (31.25%)
Predicted 78.592540

Interpretation of a CHAID diagram

CHAID analysis produces a tree diagram (*Figure 2.19*), which has a 'tree-trunk' progressively split into smaller and smaller 'branches' that terminate in small groups – 'leaves'. The initial tree-trunk is made of all samples in the pool. A series of predictor variables is assessed in order to determine whether splitting the sample-pool based on the predictors leads to a statistically significant discrimination of the dependent variable. The most significant of the available predictor variables defines the first split of the sample that appears in the diagram as the first branch of the tree-trunk. For each new group formed, the next most significant variable is identified (which may include the predictor variable used earlier) and the respective branch is split further. This process is repeated until significant splits are obtained. The terminal leaves cannot be split further using any significant predictor variable. The groups obtained are maximally different from each other.

SPSS Answer Tree 2.1 software was used to perform tree analyses. The Exhaustive CHAID option was used because, although computationally more intensive, it performs a thorough job of identifying the most significant split. Exhaustive CHAID performs segmentation modelling by using tests based on χ^2 statistics for categorical dependent variables and F statistics for continuous dependent variables in order to divide cases into groups and subgroups that are significantly different. Each time CHAID forms groups and subgroups, the values of the predictor variable being used for the segmentation are combined so as to minimise the within-group variance and maximise the between-group variance. This maximises separation between groups.

The intention was to produce a tree diagram for each group of organisms. Rarefied species richness was used as the dependent variable. In addition to land-use type, all variables listed in *Table 2.2* and *Table 2.4* were entered as predictor variables. Maximum decision tree depth – the number of branches in the tree – was set to 5. The minimum number of cases in the parent node were set to 10 and in the child node to 5. This means that a branch with fewer than 10 observations cannot be split further, and the terminal leaves will have at least 5 observations. This was chosen for the ease of data management and interpretation. Splitting probability was set to 0.05 and

Bonferroni adjustment was used to calculate the *P* value of each predictor. This adjustment takes into account the large number of variables entered into the analysis simultaneously (Bland & Altman 1995) and adjusts the *P* values accordingly, thus correcting the problem of multicollinearity in independent variables.

In order to obtain the best CHAID model, the main decision tree trunk was split into branches and leaves using the best variable out of those listed by the SPSS Answer Tree 2.1 for each node. The best variable was the one with the highest F value or the lowest adjusted *P* value or both. The splitting was continued using such variables until a non-significant split was obtained. The terminal branches with non-significant splits were pruned, thus, resulting in a tree with all statistically significant ($P \leq 0.05$) splits due to predictor variables (*Figure 2.19*).

Stepwise regression

More traditionally, regression analyses are used to examine the relationship between one dependent variable and several independent variables. Regression analysis on a continuous variable is more powerful than logistic regression on presence-absence data (van Dorp & Opdam 1987). In order to compare the results of modern methods such as CHAID with the traditional stepwise regression method, the variation in rarefied species richness across localities and its dependence on land use type, landscape and habitat characteristics was also analysed by stepwise multiple linear regression using SPSS software (SPSS Inc. 1989-1999). Linear Regression estimates the coefficients of the linear equation, involving one or more independent variables, that best predict the value of the dependent variable, while stepwise procedure examines the variables at each step for entry or removal. Regression models are compared and evaluated on the basis of their adjusted R^2, statistical ability to meet assumptions. The evaluation of the results obtained using CHAID and stepwise regression allowed comparison between:

- A modern (CHAID) and a traditional (stepwise regression) multivariate technique
- A non-parametric (CHAID) and a parametric (stepwise regression) multivariate technique.

3 . TREE DIVERSITY

The effects of landscape modification on tree diversity in Kodagu are examined in this chapter. Effects on tree communities and species are presented; conservation approaches are assessed and the implications of the results for conservation practice are discussed.

3.1 Results

3.1.1 Distribution of tree species in the landscape

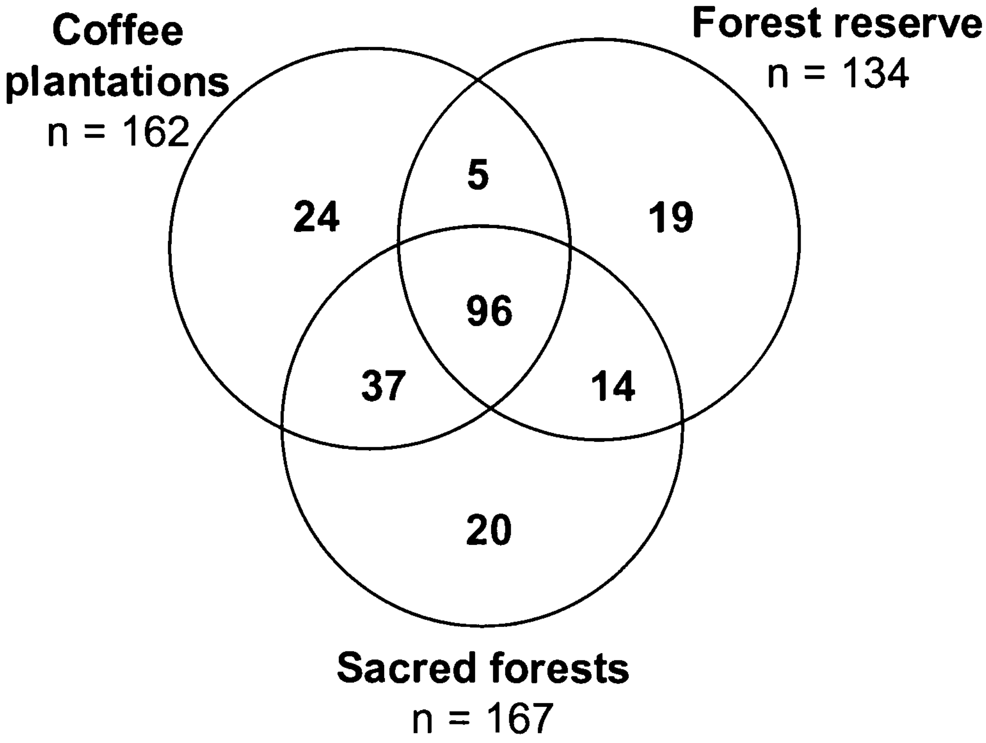


Figure 3.1 Numbers of unique and shared tree (≥ 10 cm dbh) species (Total number of species sample, N = 215). This analysis is based on species samples obtained from 58 sampling localities – 10 from the forest reserve, 25 sacred forests and 23 coffee plantations.

A total of 215 tree species were recorded in the forest reserve, sacred forests and coffee plantations (*APPENDIX 3*). Although more extensive field surveys are necessary to affirm the presence or absence of a species, the distribution of species in the three land use types, as observed in the present study, is shown in *Figure 3.1*. Species that were found only in a single type are referred to as 'unique species', those shared by any two of the three types are 'shared species' and those found in all three types are referred to as 'ubiquitous species'. Forty-five percent of all species are ubiquitous, 26% are shared, and 29% are unique. The expected distribution of unique species was calculated by adjusting the species numbers to the sample sizes in the respective land use types. The observed number of unique species was higher than expected in the forest reserve but lower in the sacred forests (χ^2 test, $\chi^2 = 6.992$, df = 2, $P = 0.0195$). Coffee plantations had nearly the same number of observed and expected unique species.

3.1.2 Effects of landscape modification on tree communities

Diversity of tree species

Rarefied species richness

Species diversity was adjusted according to sample size in order to calculate rarefied species richness at each sampling locality. Hurlbert's (1971) rarefaction method was used. The numbers of sampling localities that permitted calculation of rarefied species richness for a given number of trees ≥ 10 cm dbh are shown in *Table 3.1*. The samples were rarefied to 50 individual trees ≥ 10 cm dbh because this number allowed the inclusion of all sampling localities in the analysis. Rarefaction curves for all sampling localities are displayed in *Figure 3.2*. In more than 90% of the localities, the increase in rarefied species richness was consistent with increasing number of individuals (*Figure 3.2*). Therefore, although the species are rarefied to a smaller number of individuals (50) due to the limitations of the data, this estimate can be reliably used as an index of diversity in the present study.

Table 3.1 Numbers of sampling localities that permitted calculation of rarefied species richness for trees ≥ 10 cm dbh

	Numbers of individuals (trees ≥ 10 cm dbh) to which samples were rarefied					
	50	*55*	*75*	*100*	*125*	*150*
Numbers of sampling localities that permitted calculation of rarefied species richness						
Reserve (n = 10)	10	10	9	5	2	0
Sacred (n = 25)	25	25	22	2	1	0
Coffee (n = 23)	23	23	20	6	4	3
Total (n = 58)	***58***	***58***	***51***	***13***	***7***	***3***

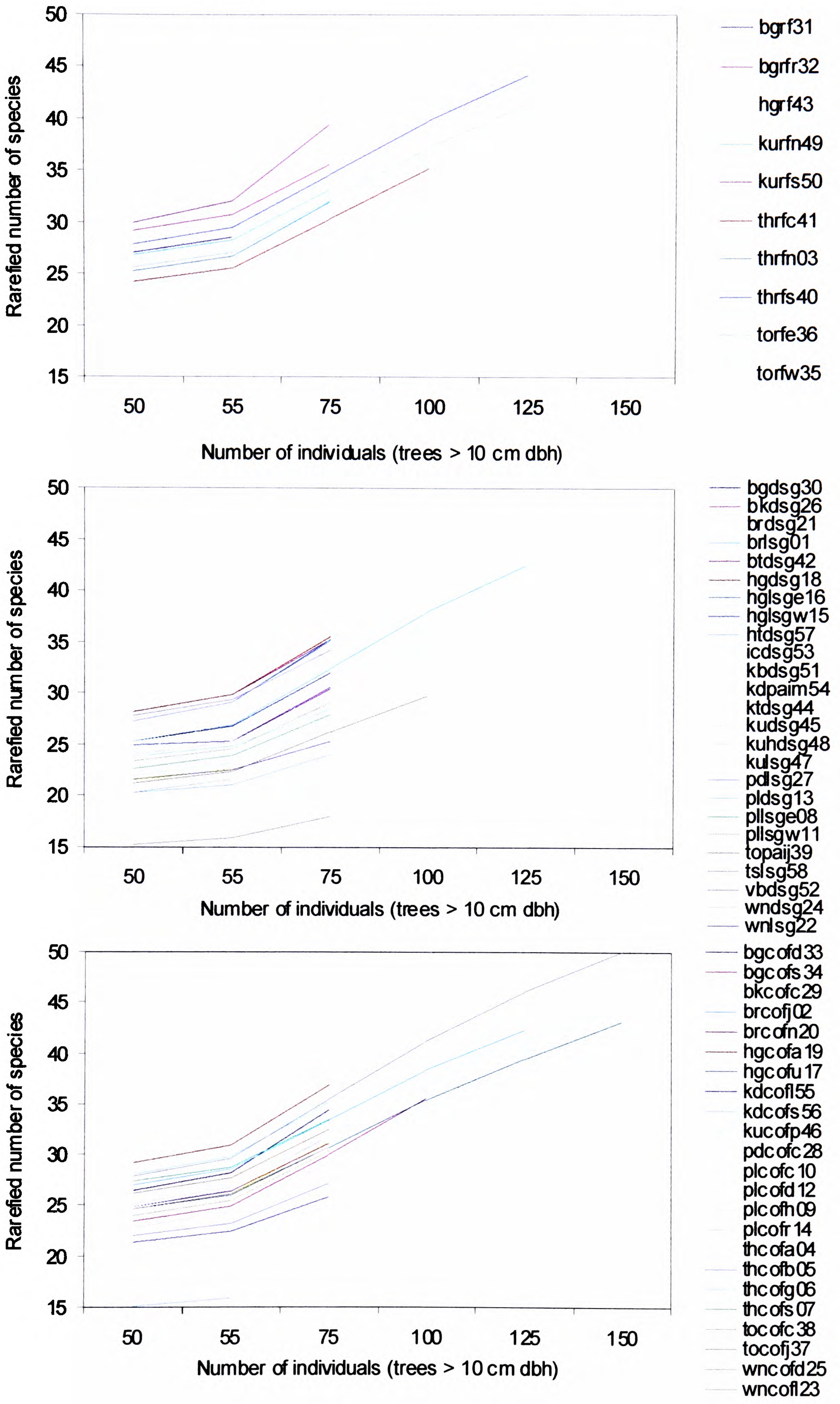


Figure 3.2 Rarefaction curves (trees ≥ 10 cm dbh) for sampling localities; the top panel shows forest reserve localities (n = 10), the middle panel shows sacred forests (n = 25) and the bottom panel shows coffee plantations (n = 23).

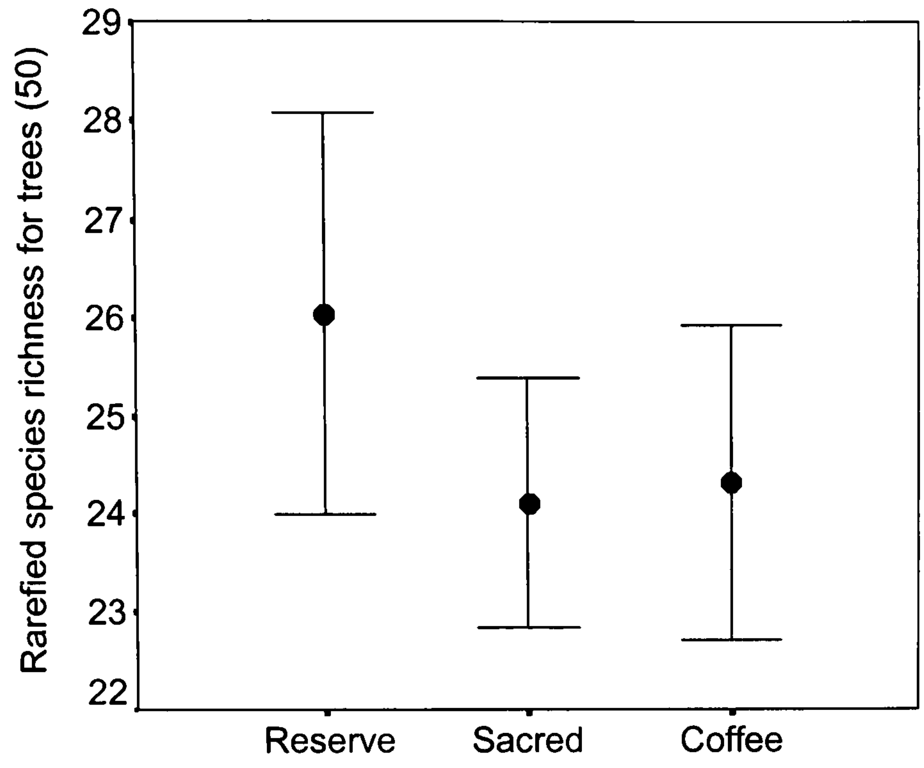


Figure 3.3 Mean rarefied tree (≥ 10 cm dbh) species richness (for 50 individuals) in the three land use types; numbers of samples in the forest reserve, sacred forests and coffee plantations were 10, 25 and 23 respectively. Note: error bars indicate 95% confidence intervals.

There were no significant differences in the mean rarefied species richness (One-way ANOVA, F = 1.271, df = 2, $P = 0.289$) across the three land use types (*Figure 3.3*).

Effect of patch area

Patch size is one of the most obvious parameters that can be measured in fragmented landscapes and is often found to explain patterns of species richness (Arrehnius 1921; Preston 1960; Connor & McCoy 1979; Rosenzweig 1995). Although the documented areas of sacred forest patches (Karnataka State Forest Department 1985) were inconsistent with what they appeared to be on the ground, GIS provided means for measuring areas of patches accurately from digitised maps. In order to examine whether tree diversity was associated with patch size, the rarefied species richness of trees in sacred forests were plotted against the sizes of respective patches. No significant association ($R^2 = 0.001$, $P = 0.958$) was found between patch size and rarefied species richness per 50 individuals ≥ 10 cm dbh (*Figure 3.4*).

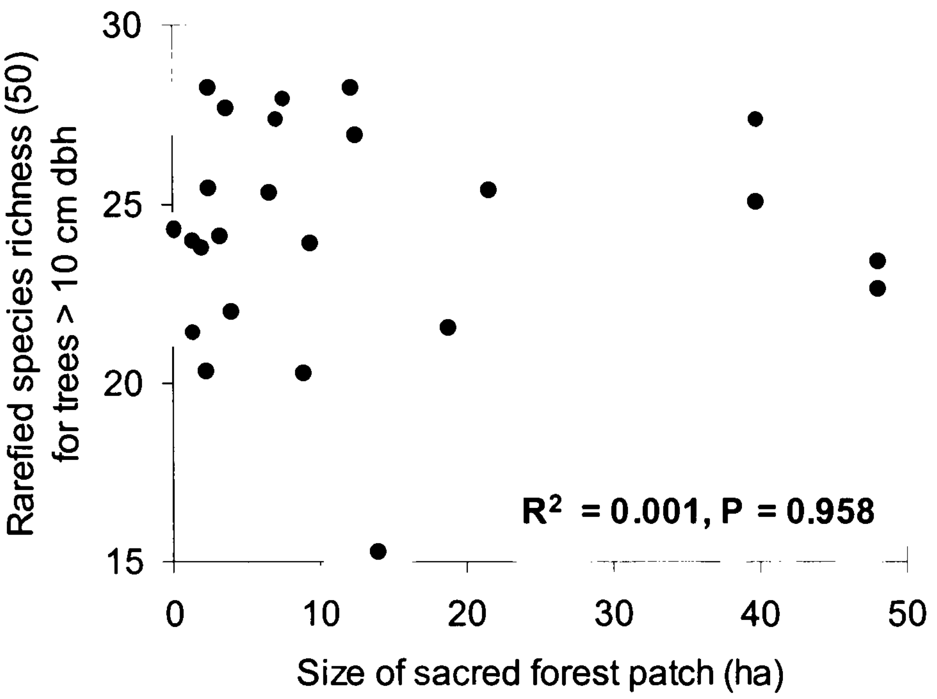


Figure 3.4 The relationship between the size of sacred forest patch (n = 24) and tree (≥ 10 cm dbh) diversity.

Landscape effects

In the highly seasonal evergreen forests of the Western Ghats, the proportion of evergreen species declines along a latitudinal gradient corresponding to the decrease in rainfall and increase in length of the dry season from south to north (Utkarsh *et al.* 1998). Human activity is another important factor in the proportional decline of evergreen trees in the region (Daniels *et al.* 1995). Elsewhere in the tropics, for instance in Australia and Brazilian Amazonia, phenological characteristics of tree communities are known to alter as a result of forest fragmentation (Laurance 1991b), causing proportional increases in deciduous trees.

Based on the published information (Pascal 1988; Keshavamoorthy & Yoganarasimhan 1989), trees were classified into evergreen or semi-evergreen, and deciduous or semi-deciduous. The proportion of evergreen trees declined (One-way ANOVA, F = 5.626, df = 2, *P* = 0.006) and that of deciduous trees increased with increasing human intervention in land management (*Figure 3.5*). In addition to the human intervention, this can be attributed to the slight west-east rainfall gradient in the study area (*Figure 2.4*).

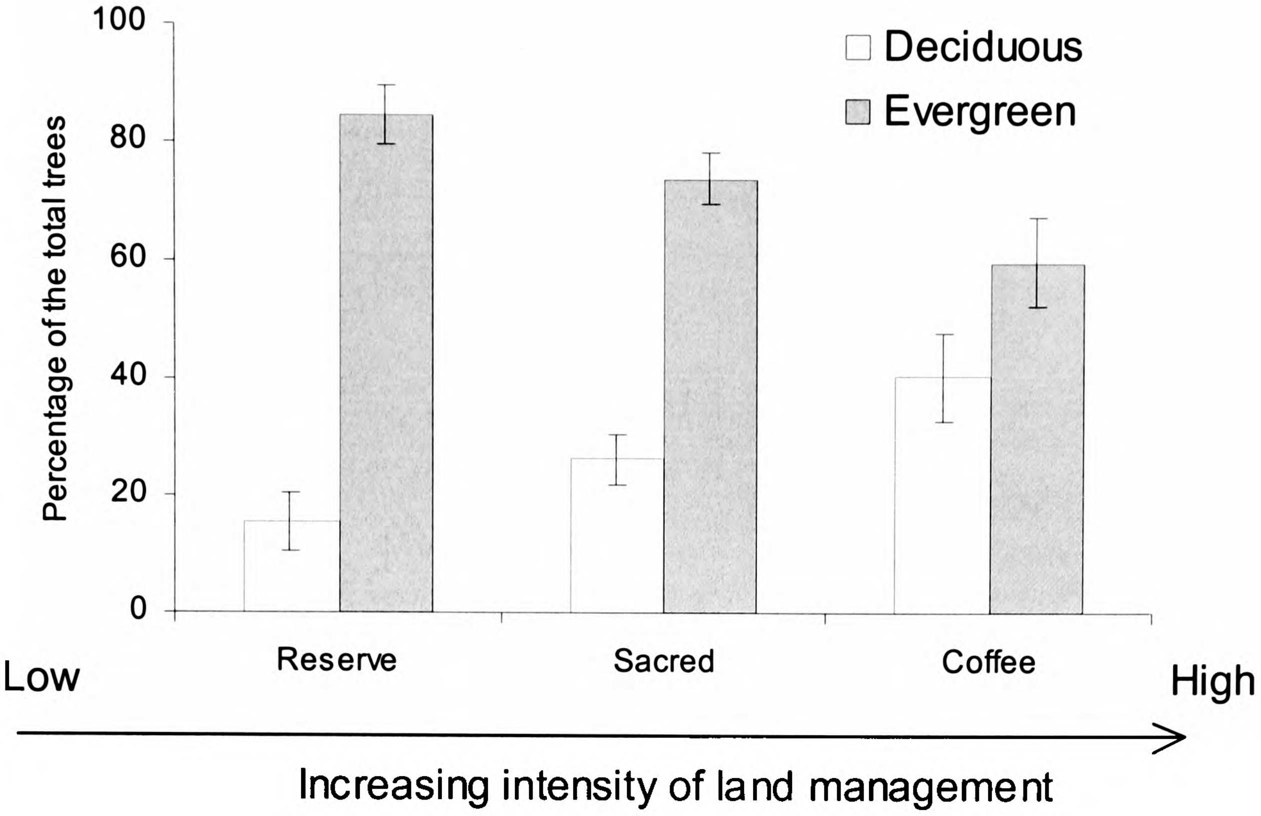


Figure 3.5 The decline in the proportion of evergreen trees (individuals, ≥ 10 cm dbh) and an increase in the proportion of deciduous trees with growing human intervention in land management. The forest reserve (n = 10), sacred forest patches (n = 25) and coffee plantations (n = 23) are on the continuum of human intervention - the forest reserve has the lowest degree of land management and coffee plantations the highest. Note: error bars indicate 95% confidence intervals.

Effect of isolation

Two methods, CHAID and stepwise regression, were used to identify important landscape variables. CHAID identified isolation to be the most significant ($P < 0.001$) variable that explained tree diversity in the landscape. Other important factors identified by CHAID include variability in canopy height ($P < 0.01$) and stem density ($P < 0.05$) (*Figure 3.6*). The results of CHAID were compared with that of forward stepwise regression (*Table 3.2*). Although isolation was the most significant factor in explaining diversity of trees ≥ 10 cm dbh, it explained only 8% of the total variation.

Figure 3.6 (Please see Page 84) CHAID diagram showing variables that are significant in explaining tree (≥ 10 cm dbh) diversity. Full titles of the variables and the ranges of values they take are shown in boxes.

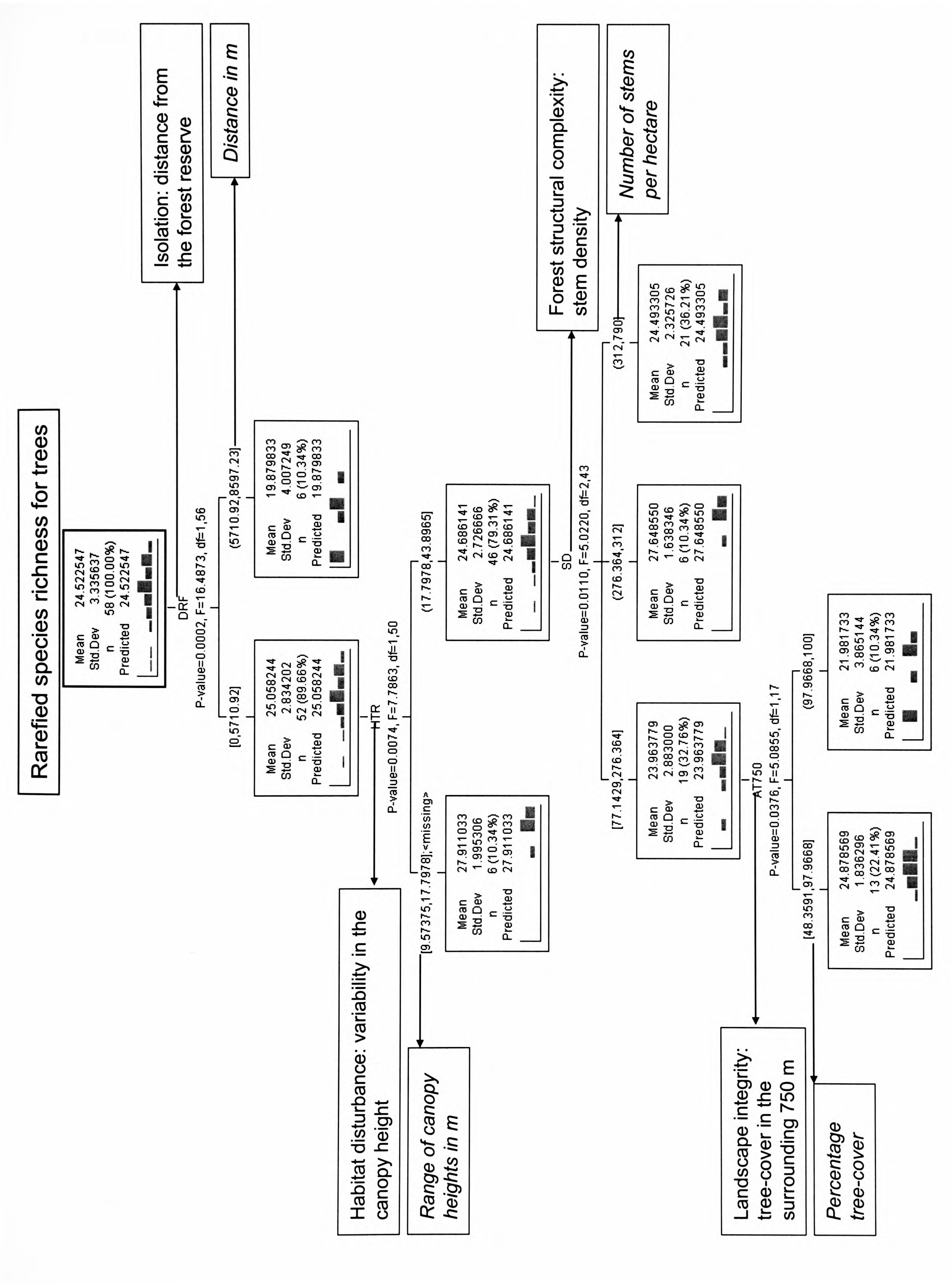

Rarefied species richness for trees
Mean 24.522547
Std.Dev 3.335637
n 58 (100.00%)
Predicted 24.522547
DRF
P-value=0.0002, F=16.4873, df=1,56
Isolation: distance from the forest reserve
Distance in m
[0,5710.92]
Mean 25.058244
Std.Dev 2.834202
n 52 (89.66%)
Predicted 25.058244
(5710.92,8597.23]
Mean 19.879833
Std.Dev 4.007249
n 6 (10.34%)
Predicted 19.879833
HTR
P-value=0.0074, F=7.7863, df=1,50
Habitat disturbance: variability in the canopy height
Range of canopy heights in m
[9.57375,17.7978];<missing>
Mean 27.911033
Std.Dev 1.995306
n 6 (10.34%)
Predicted 27.911033
(17.7978,43.8965]
Mean 24.686141
Std.Dev 2.726666
n 46 (79.31%)
Predicted 24.686141
SD
P-value=0.0110, F=5.0220, df=2,43
Forest structural complexity: stem density
Number of stems per hectare
[77.1429,276.364]
Mean 23.963779
Std.Dev 2.883000
n 19 (32.76%)
Predicted 23.963779
(276.364,312]
Mean 27.648550
Std.Dev 1.638346
n 6 (10.34%)
Predicted 27.648550
(312,790]
Mean 24.493305
Std.Dev 2.325726
n 21 (36.21%)
Predicted 24.493305
AT750
P-value=0.0376, F=5.0855, df=1,17
Landscape integrity: tree-cover in the surrounding 750 m
Percentage tree-cover
[48.3591,97.9668]
Mean 24.878569
Std.Dev 1.836296
n 13 (22.41%)
Predicted 24.878569
(97.9668,100]
Mean 21.981733
Std.Dev 3.865144
n 6 (10.34%)
Predicted 21.981733

Table 3.2 Forward stepwise regression model for tree (≥ 10 cm dbh) diversity.

Step	R^2	Variable	T value	*P*
1	0.0829	Distance from the forest reserve (isolation)	- 2.17	0.00039
2	0.1953	Distance from the forest reserve	- 3.18	0.00060
		Tree-cover in the surrounding 500 m	- 2.67	0.104

There was consistency in the most important variable – isolation of a locality from the forest reserve – identified by both methods. However, the subsidiary variables were inconsistent. For instance, while CHAID identified habitat variables such as stem density and variability in canopy height as significant (*Figure 3.6*), stepwise regression identified tree cover in the surrounding landscape as important but not significant (*Table 3.2*). While isolation may have a linear relationship with tree diversity, variables of habitat quality such as stem density and variability in canopy height may be related in non-linear fashion. Although more detailed analyses of these relationships are necessary in order to assess the ecological meaning of the statistically powerful CHAID analysis, CHAID appears to be a more insightful technique in comparison with stepwise regression for its ability to identify non-linear relationships.

Effects of isolation and stem density

It is well known that abundance in some species increases disproportionately in response to fragmentation, caused by edge effects coupled with the effects of the surrounding landscape matrix (*e.g.* Brown & Hutchings 1997; Laurance 1997a). In response to fragmentation and species loss, some disturbance-adapted species achieve unusually high densities due to reduced inter-specific competition (MacArthur *et al.* 1972). For instance, Laurance *et al.* (1998b) observed that in Amazonian tree communities early successional trees increased in older and smaller fragments because of recurring disturbances and favourable light and microclimatic conditions, which facilitated their germination and growth (Clark 1990). The relationship between tree diversity and stem density in forest patches is examined in the rest of this section. Only 24 forest patches and 8 forest reserve localities (1 sacred forest and 2 forest reserve localities that do not have sufficient data for stems ≥ 1 cm dbh are excluded) are considered in this analysis because stem densities in coffee plantations are artificial – the understorey is removed and the shade trees are thinned in order to plant coffee.

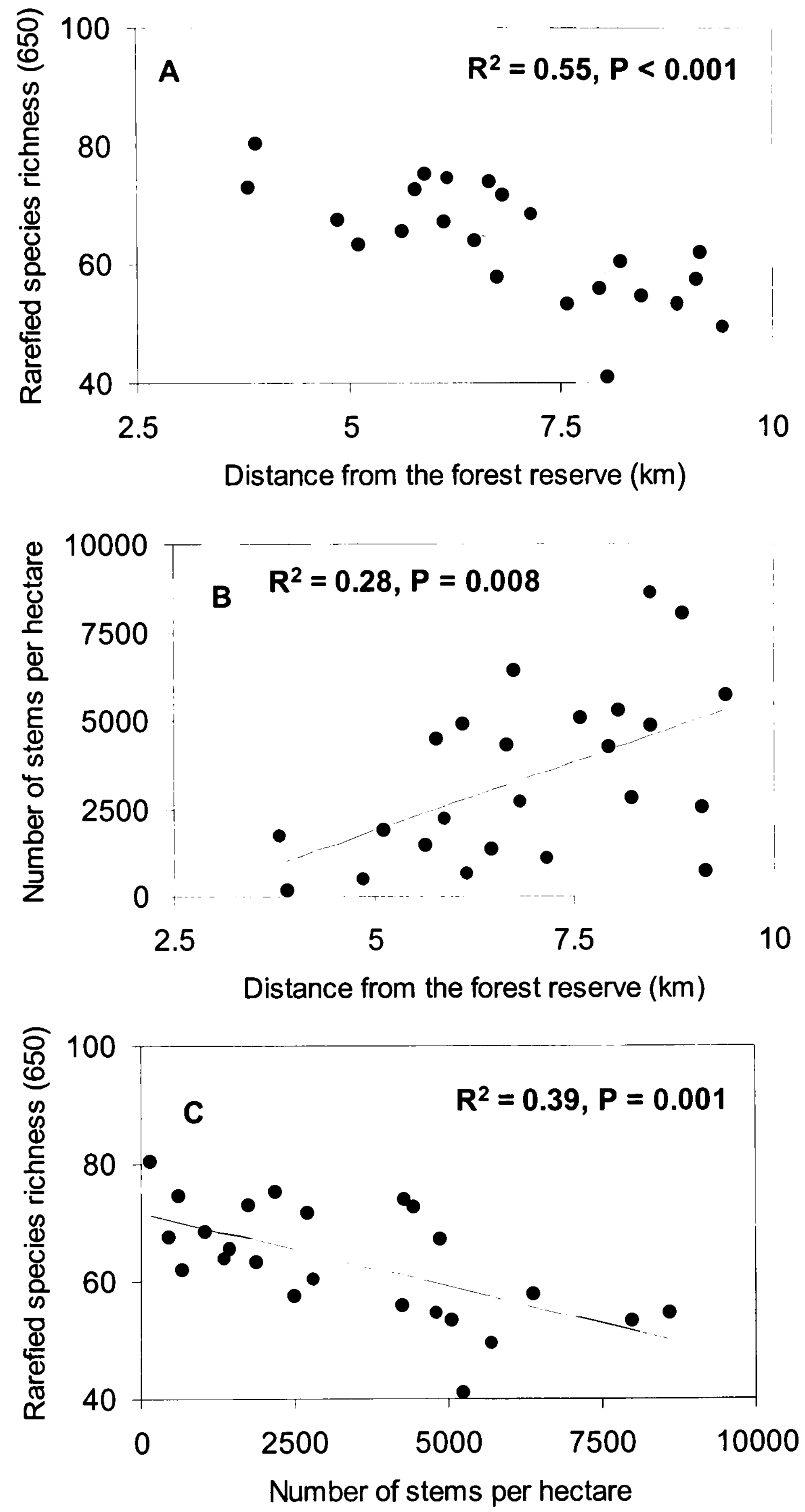


*Figure 3.7 (**A**) – Decrease with isolation in diversity of trees with stems ≥ 1 cm dbh in sacred forest patches (n = 24); (**B**) – Increase with isolation in density of stems ≥ 1 cm dbh in sacred forest patches (n = 24); (**C**) – The relationship between density of stems ≥ 1 cm dbh and tree diversity in sacred forest patches (n = 24).*

There was a strong negative association ($R^2 = 0.55$, $P < 0.001$) between diversity of stems ≥ 1 cm dbh and the distance of a patch from the forest reserve (*Figure 3.7, A*) suggesting steep decline in diversity in more distant forest patches. Stem density increased rather weakly with isolation ($R^2 = 0.28$, $P = 0.008$) (*Figure 3.7, B*). There is a negative relationship ($R^2 = 0.39$, $P = 0.001$) between tree diversity and stem density (*Figure 3.7, C*).

However, apart from the isolation, other factors that were not measured may be at work in the Kodagu landscape. For instance, growing human disturbance in localities farther away from the reserve may contribute to the increase in stem density in isolated patches. In addition, certain dispersal-limited species may not be able to reach patches that are far away from the reserve. In effect, isolated patches are left with fewer species. Under these circumstances, some widely dispersed or disturbance-adapted species become abundant and dominate the tree community in forest patches by forming a dense understorey of saplings.

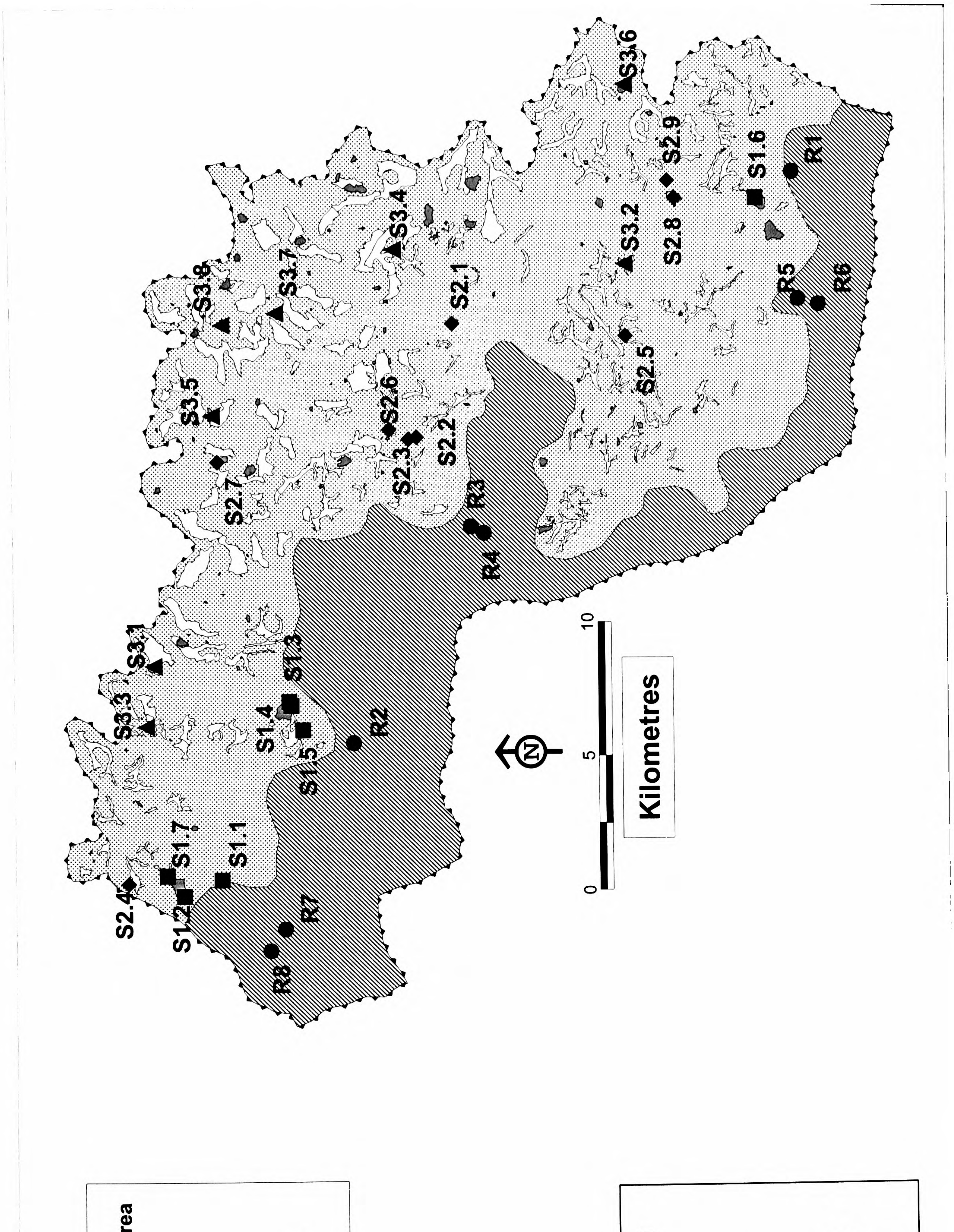
Species-abundance distribution

One of the consequences of the unusually high abundances attained by some species is a change in species-abundance distribution of the community. *Figure 3.9* shows rank-abundance curves, for trees ≥ 1 cm dbh, for sacred forests at varying distances from the reserve. Coffee plantations were excluded from this analysis because the understorey is cleared for planting coffee; as a result there are no individuals between 1 and 10 cm dbh. Only 24 sacred forest patches and 8 forest reserve localities are considered in this analysis, while 1 sacred forest and 2 forest reserve localities that do not have sufficient data for stems ≥ 1 cm dbh are excluded. The 32 sampling localities in the reserve and sacred forest patches were divided into four distance bands. The first band included eight forest reserve localities, all considered to be at 0 km from the reserve. The second, third and the fourth bands included 7, 9 and 8 sacred forest patches that were at < 1.5 km, 1.5–5 km and > 5 km from the edge of the forest reserve respectively (*Figure 3.8*). Tree species were ranked according to their abundances. The means of the relative species abundances in each of the ranks (irrespective of the identity of the species) were plotted against the corresponding ranks on a semi-log plot (*Figure 3.9*). The aim was limited to the investigation of the underlying processes that cause the given species-abundance distribution (Tokeshi 1993); the examination of ecological characteristics of individual species was not intended in this analysis. It was assumed that the identity of a species at the given rank is not important.

The curve for the forest reserve (0 km) had the lowest gradient, and that for the localities more than 5 km away was the steepest. This suggests that a small number of species become dominant with increasing distance from the reserve (*cf.* Laurance 1997a).

Tokeshi (1993) argues that, although routinely practiced by proponents of most species-abundance distribution models, "visual inspection" of rank-abundance curves (*Figure 3.9*) can often be misleading. In order to compare whether the species-abundance distributions are significantly different from one another, statistical methods can be used. The distributions in three distance bands of sacred forests were compared with the distribution in the reserve using the Two-Sample Kolmogorov-Smirnov Test. The differences between the distributions were not significant ($Z = 0.707$, 0.950, 1.166 with corresponding P values of 0.700, 0.328, 0.132 $n = 9$, 7 and 8 respectively, with increasing distance from the reserve). This may be because all four distributions are from the same type of ecological community (*i.e.* medium-altitude evergreen forest) and the *Kolmogorov-Smirnov test* is not sensitive enough to reveal differences.

Figure 3.8 (Please see Page 89) The positions of sampling localities in the forest reserve and sacred forest patches at various distances from the reserve. The 32 sampling localities are divided into four distance bands. Each sampling locality is represented by a point. Those titled R belong to the forest reserve and those titled S belong to sacred forests. In each title, the number following S represents the distance band that the particular locality belongs to (1 corresponds with < 1 km, 2 corresponds with 1.5-5 km, 3 corresponds with > 5 km away from the forest reserve). Numbers following R and those at the end of titles beginning with S indicate ordering of localities within a particular distance band.


Legend
Boundary of the study area
Tree-covered land
Sacred forest patch
Forest reserve
Boundary of a patch
Non tree-covered patch
Distance (km) of Sampling locality (Table 2.1)
● 0 (Forest reserve)
■ < 1.5 (Sacred forest)
◆ 1.5 – 5 (Sacred forest)
▲ >5 (Sacred forest)
N
0
5
10
Kilometres
R8
R7
S1.2
S2.4
S1.7
S1.1
S1.5
R2
S1.4
S3.3
S1.3
S3.1
R4
R3
S2.7
S2.3
S2.2
S2.6
S3.5
S2.5
S3.8
S3.7
S2.1
S3.4
R5
R6
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S2.8
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S1.6
R1
S3.6

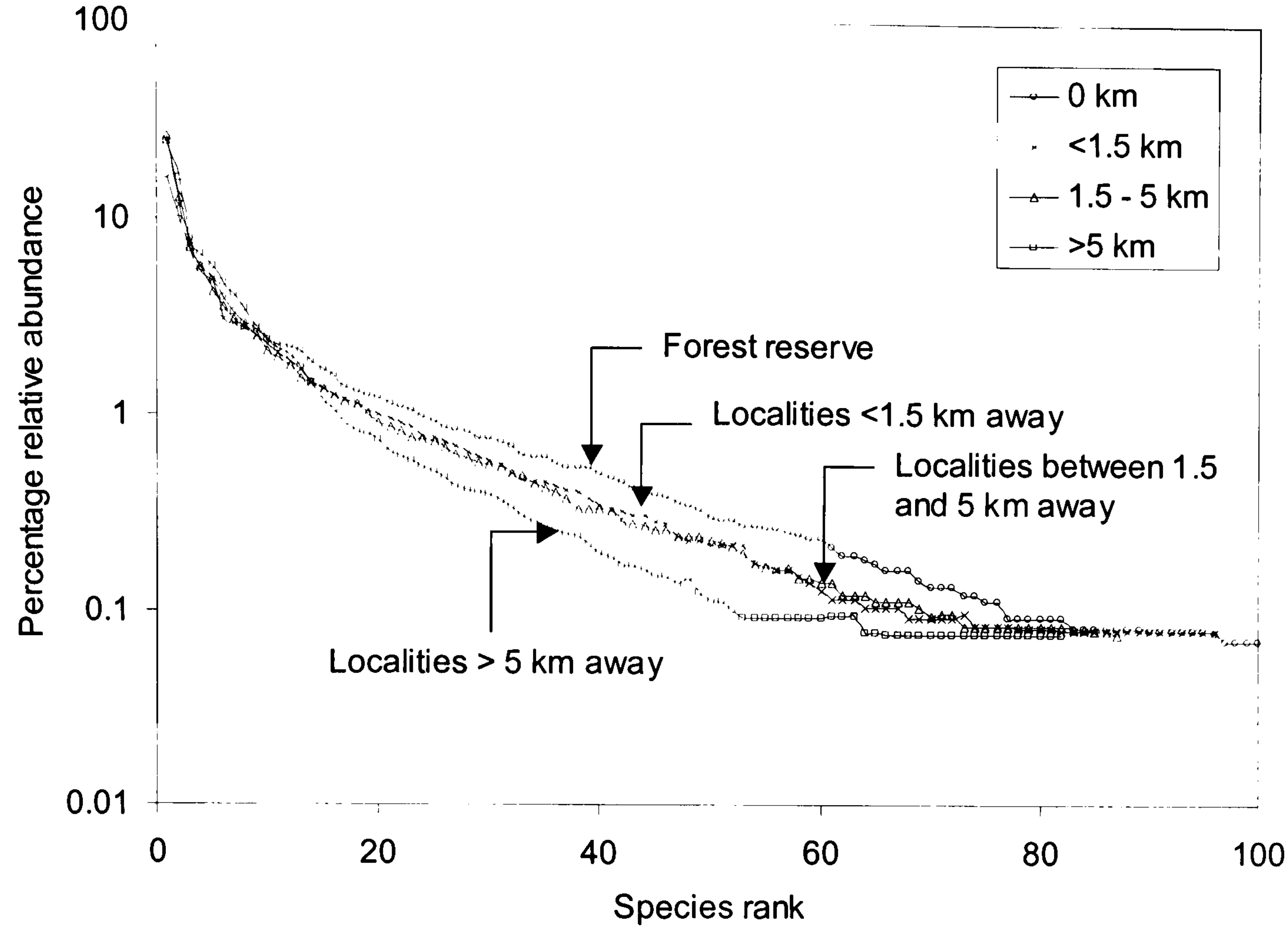


Figure 3.9 Rank-abundance curves for the forest reserve (0 km) and sacred forest patches (< 1.5, 1.5-5, > 5 km) in four distance bands along an increasing degree of isolation. Total number of individuals (trees ≥ 1 cm dbh), 'N' = 39,271; total number of sites, 'n' = 32. Number of individuals, 'N' and number of sampling localities, 'n' in each distance band are 9827, 8641, 10814, 9989 and 8, 9, 7, 8 respectively along the increasing distance gradient.

Similarity in tree diversity

Due to the disproportionately high abundances of some disturbance-adapted species, a few species in a community dominate the species-abundance distributions (*Figure 3.9, see the curve for localities > 5 km away from the forest reserve*). As a result, isolated patches may become dissimilar to the unbroken forest. The similarity between the forest reserve and the sacred forest patches at various distances from the reserve is examined in the rest of this section. Only 24 forest patches and 8 forest reserve localities are considered in this analysis, while 1 sacred forest and 2 forest reserve localities that do not have sufficient data for stems ≥ 1 cm dbh are excluded.

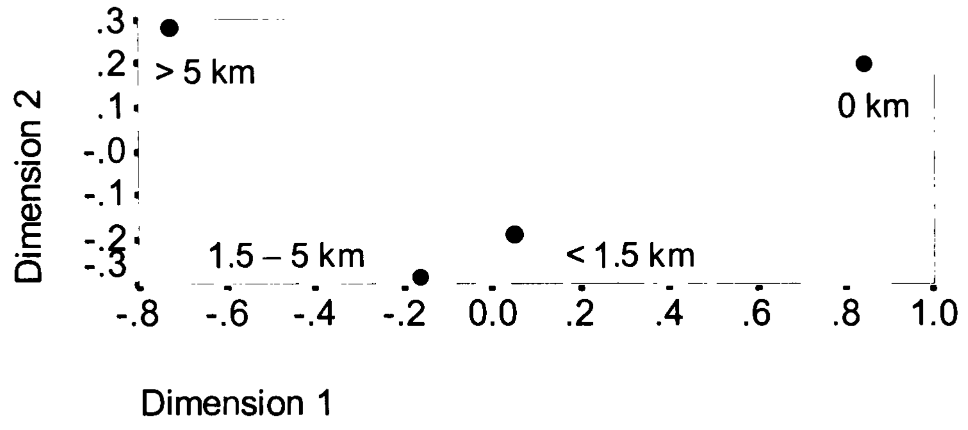
Table 3.3 Mean Sorenson's quantitative index showing a decline in similarity in tree (≥ 1 cm dbh) diversity between the reserve (0 km) and the sacred forest patches (at < 1.5, 1.5-5, > 5 km away from the reserve).

	Sorenson's quantitative index			
Distance (km)	**0**	**< 1.5**	**1.5 - 5**	**> 5**
% Similarity	100	89	87	81
n	8	9	7	8

Sorenson's quantitative similarity index (Magurran 1988) gives an estimate of the percentage similarity between rank-abundance distributions. Similarity in tree diversity between sacred forests and the forest reserve decreased with the distance of patches from the reserve (*Table 3.3*).

Non-metric multidimensional scaling (Kenkel & Orloci 1986) assigns observations of community similarities (represented by points) to specific locations in a conceptual low-dimensional space such that the distances between points in the space match the given similarities as closely as possible. The results of non-metric multidimensional scaling are presented in *Figure 3.10.*

The results suggest that the sacred forests more than 5 km away from the forest reserve are most dissimilar to the reserve. Furthermore, dissimilarity increases progressively with distance. The sacred forests that are 1.5-5 km away and those that are less than 1.5 km away are more similar to each other in comparison with those that are more than 5 km away from the reserve. This also corresponds with the rank-abundance curves (*Figure 3.9*), where the two curves (those for the localities 1.5-5 km away and less than 1.5 km away) are virtually indistinguishable.



Euclidean distances

Dimensionality: 2

	0 km	< 1.5 km	1.5-5 km	> 5 km
0 km	.000			
< 1.5 km	1.153	.000		
1.5-5 km	1.273	0.179	.000	
> 5 km	1.443	0.758	0.592	.000

Figure 3.10 Non-metric multidimensional scaling of similarity in tree (≥ 1 cm dbh) diversity between the reserve (0 km) and sacred forest patches (at < 1.5, 1.5-5, > 5 km from the reserve). The χ^2 distances progressively increased along the gradient of physical distances (in km) of forest patches from the reserve.

In addition to the changes in rank-abundance distribution with isolation, a slight gradient of rainfall (*Figure 3.11*) may be important in explaining the pattern of decline in similarity. However, the comparison of rainfall data in *Figure 3.11* is based only on the records in a single year. Therefore, the result should be interpreted with caution. Although FRLHT (2000a) suggest that there is no significant rainfall gradient across the study area from west to east (*Figure 2.4*), this claim is based on a very ‘coarse-grained’ map. The difference in rainfall at forest reserve localities and those > 5 km away from it is significant (*Figure 3.11*). Although there is a significant difference in rainfall between the forest reserve and sacred forests > 5 km away from the reserve, the differences between sacred forests at various distances from the reserve do not appear to be significant (*Figure 3.11*). This variable could not be included in the multivariate model used to explain tree diversity (*Figure 3.6*) due to the lack of sufficient data. Nevertheless, rainfall appears to be an important variable.

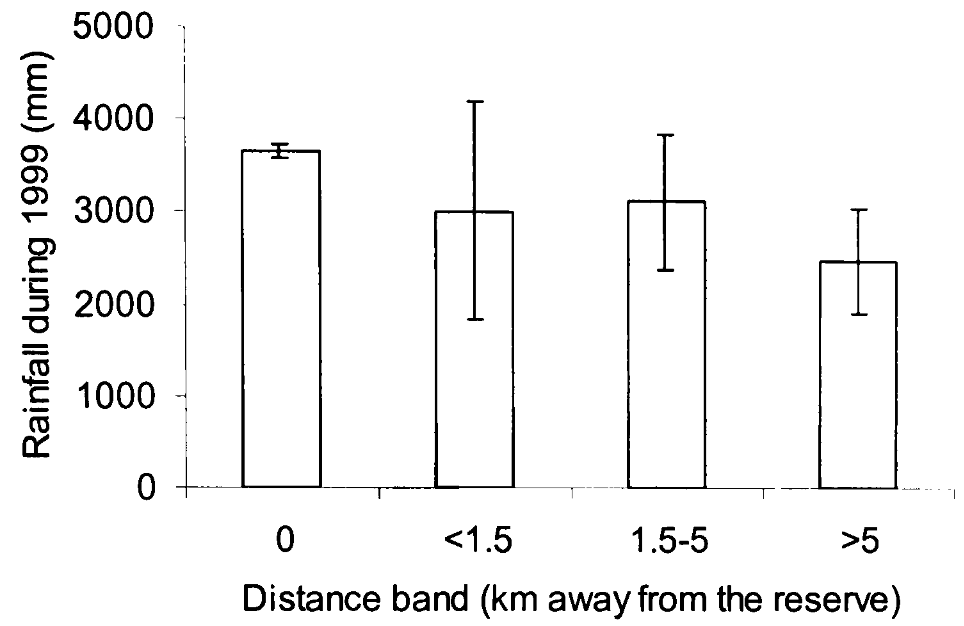


Figure 3.11 The rainfall information collected during 1999 in four distance bands. The information was available from 2 sites located in the forest reserve (0 km), 2 from the sites < 1.5 km away, 3 from the sites between 1.5-5 km away and 3 from those > 5 km away from the reserve.

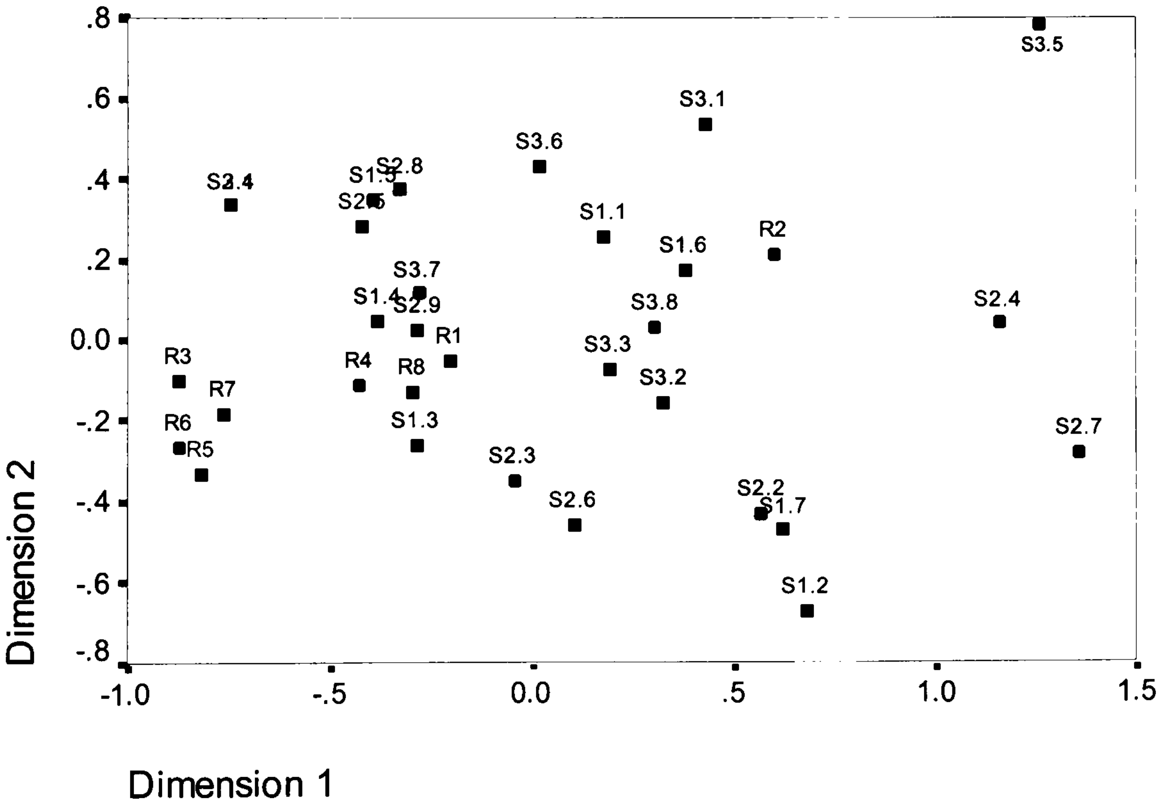


Figure 3.12 Non-metric multidimensional scaling of similarity in tree (≥ 1 cm dbh) diversity between the localities in the reserve and sacred forest patches; each sampling

locality is represented by a point. Those titled R belong to the forest reserve and those titled S belong to sacred forests. In each title, the number following S represents the distance band that the particular locality belongs to (1 corresponds with < 1 km, 2 corresponds with 1.5-5 km, 3 corresponds with > 5 km away from the forest reserve). Numbers following R and those at the end of titles beginning with S indicate ordering of localities within a particular distance band (e.g. S 2.2 indicates the locality in sacred forest that is 1.5-5 km away from the reserve and is the second nearest locality to the reserve in that distance band).

In order to examine how similar individual sacred forest patches are to each other and to the localities in the reserve, rank-abundance distributions of trees ≥ 1 cm dbh were subjected to non-metric multidimensional scaling (*Figure 3.12*). Coffee plantations were excluded from this analysis because they do not contain individuals ≥ 1 and < 10 cm dbh. They were excluded also to avoid over-crowding of points in the two-dimensional representation shown in *Figure 3.12*. Most forest reserve localities clustered together, but this may be because of their clumped distribution (*Figure 3.8*). There was no ordering in sacred forest localities (*Figure 3.12*) (S-STRESS = 0.00602).

Table 3.4 Pair-wise comparisons of dissimilarity in tree diversity between forest reserve localities and sacred forest patches in the Kodagu landscape; the values show Euclidean distances; larger the distance value, higher is the dissimilarity. Locations of sampling localities within the study area are mapped in Figure 3.8.

Locality	**R1**	**R2**	**R3**	**R4**	**R5**	**R6**	**R7**	**R8**
S1.1	109.115	78.759	191.238	126.024	195.556	197.712	174.786	108.120
S1.2	207.451	165.158	306.901	227.548	287.844	298.010	288.753	221.630
S1.3	45.343	181.942	120.050	55.534	116.696	119.046	107.842	54.498
S1.4	61.782	179.967	113.442	41.605	119.235	126.258	97.093	80.579
S1.5	82.662	191.692	124.133	84.220	142.888	140.595	116.168	67.164
S1.6	114.013	78.677	229.689	156.381	236.163	237.691	218.497	132.977
S1.7	174.940	137.117	283.072	205.529	271.271	277.063	267.776	186.682
S2.1	114.232	251.201	110.458	99.373	127.260	133.712	115.486	126.693
S2.2	165.674	131.700	274.915	194.926	263.348	270.167	258.913	177.048
S2.3	83.522	140.211	155.300	94.647	144.672	148.822	141.492	85.884
S2.4	257.088	117.563	369.749	288.482	367.083	372.457	351.898	270.991
S2.5	78.994	195.154	132.227	99.710	152.033	145.705	131.217	69.613
S2.6	104.274	151.888	192.844	118.832	175.929	185.486	177.947	115.823
S2.7	298.776	164.091	409.569	330.283	403.313	409.264	392.702	313.109
S2.8	83.024	178.880	115.451	79.063	138.127	137.880	106.184	86.250
S2.9	59.178	160.415	106.141	55.009	113.851	115.681	94.942	52.650
S3.1	161.536	80.542	256.176	189.288	268.276	270.839	243.606	173.632
S3.2	110.959	90.017	219.171	133.836	212.038	219.905	200.150	128.312
S3.3	84.658	123.183	204.639	125.591	206.395	211.341	193.041	106.597
S3.4	114.232	251.201	110.458	99.373	127.260	133.712	115.486	126.693
S3.5	316.051	172.073	419.252	347.301	426.711	430.232	404.421	331.955
S3.6	100.000	124.439	185.359	119.139	198.348	202.569	172.017	121.293
S3.7	46.379	177.736	122.536	62.538	130.755	136.803	118.731	77.285
S3.8	105.527	80.218	216.171	132.008	213.967	221.946	199.379	127.570

Pair-wise comparisons of dissimilarity between forest reserve localities and sacred forests in three distance bands did not reveal any distinct pattern (*Table 3.4*).

3.1.3 Effects of landscape modification on tree species

Landscape modification may have negative effects on species that are rare and have restricted or patchy distributions (Wilcox 1980; Kellman *et al.* 1996; Laurance *et al.* 1998a). Also, the populations of habitat-specialist species are negatively affected while there is a positive effect on the populations of disturbance-adapted species (Saunders *et al.* 1991; Brown & Hutchings 1997; Laurance *et al.* 1998b). In the present study, while most species were ubiquitous (*Figure 3.1*), a few were restricted to the forest reserve, many of them endemic to the Western Ghats (*Table 3.5*). These species may be affected by landscape modification and are not found in the samples from patchy landscape in the present study.

Out of the total 215 tree species, 14 were selected based on the adequacy of their representation in tree diversity samples from the reserve ($n = 8$) and the sacred forests ($n = 24$) (including all those that occurred at more than 15 sampling localities out of 32; and for which a minimum of 100 trees ≥ 10 cm dbh were encountered in all samples together). Only 24 sacred forest patches and 8 forest reserve localities are considered in this analysis, while 1 sacred forest and 2 forest reserve localities that do not have sufficient data for stems ≥ 1 cm dbh are excluded. The effects of isolation on stem densities of each of the 14 species were examined by regression (*Table 3.6*). *Cinnamomum macrocarpum*, *Dimocarpus longan*, *Nothopegia beddomei* and *Olea dioica* occurred at very high stem densities, suggesting that they may be adapted to disturbance in the Kodagu landscape. There was positive association between their densities and the isolation of localities where they occurred (*Table 3.6*). The stem densities of *Actinodaphne malabarica*, *Aglaia jainii*, *Antidesma menasu*, *Calophyllum polyanthum*, and *Flacourtia montana* also showed positive association. The associations were not significant in the remaining species (*Table 3.6*).

Table 3.5 Evergreen habitat-specialist tree (≥ 10 cm dbh) species unique to the forest reserve (An asterisk indicates species endemic to the Western Ghats).

* *Aglaia elaeagnoidea* - Meliaceae
Agrostistachys meeboldii - Euphorbiaceae
* *Baccouria courtallensis* - Euphorbiaceae
* *Blachia denudata* - Euphorbiaceae
* *Diospyros pruriens* - Ebenaceae
* *Drypetes oblongifolia* - Euphorbiaceae
* *Elaeocarpus munronii* - Elaeocarpaceae
* *Garcinia indica* - Clusiaceae
Garcinia pictorius - Clusiaceae
* *Heritiera papilio* - Sterculiaceae
* *Humboldtia brunonis* – Fabaceae (Caesalpinioideae)
* *Litsea glabrata* - Lauraceae
Litsea insignis - Lauraceae
* *Mallotus stenanthes* - Euphorbiaceae
Memecylon wightii - Melastomataceae
Mitrephora heyneana - Annonaceae
Polyalthia coffeoides - Annonaceae
* *Schefflera capitata* - Araliaceae
Syzygium lanceolatum - Myrtaceae

In order to compare population trends in species in contiguous and patchy landscapes, size-class distributions of two contrasting species, *Nothopegia beddomei*, which occurs at a high stem density (*Table 3.6*), and *Toona ciliata*, which occurs at a low stem density (*Table 3.6*) in the forest reserve and sacred forests were examined. *Nothopegia beddomei* regenerates much more profusely (Kolmogorov-Smirnov test, Z = 1.604, P = 0.012) in sacred forests than in the forest reserve (*Figure 3.13*). *Cinnamomum macrocarpum*, *Dimocarpus longan* and *Olea dioica* (*Table 3.6*), three other species that occurred at high densities, showed similar trends. By contrast, *Toona ciliata* occurs at a low density in sacred forests. There was no difference in its size-class distribution (Kolmogorov-Smirnov test, Z = 0.802, P = 0.541) in sacred forest patches and the reserve (*Figure 3.14*). *Evodia lunu-ankenda, Lagerstroemia lanceolata* and *Syzygium gardnerii* (*Table 3.6*) also belong to this group. However, based on this information it is not possible to compare or comment on the population growth in these two types of species – Condit *et al.* (1998) have shown that fast-growing tree species have fewer juveniles in the understorey than slow-growing species even when population growth is equal. The results are limited to a 'snap-shot' picture of the effects of landscape modification on species that are representative of the contrasting trends in stem density and size-class distributions.

Table 3.6 Effect of isolation (distance from the forest reserve) on stem density of tree (≥ 1 cm dbh) species in sacred forests (n = 24) and the forest reserve (n = 8) (The numbers of sampling localities where species occur are in brackets; data on stem density from a total of 32 localities were used). Species shown in the table occurred at more than 15 sampling localities out of 32. A minimum of 100 trees ≥ 10 cm dbh of these species were encountered in all samples together.

Species name	**Average number of stems ≥ 1cm dbh per ha**	**Nature of effect**	**R^2**	***P***
Actinodaphne malabarica - Lauraceae	20 (28)	+ ve	.138	.036
Aglaia jainii - Meliaceae	30 (25)	+ ve	.163	.022
Antidesma menasu - Euphorbiaceae	59 (31)	+ ve	.308	.001
Calophyllum polyanthum - Clusiaceae	48 (15)	+ ve	.143	.033
Cinnamomum macrocarpum - Lauraceae	159 (24) ‡	+ ve	.370	.0001
Dimocarpus longan - Sapindaceae	1046 (30) ‡	+ ve	.166	.021
Evodia lunu-ankenda - Rutaceae	23 (13)	No	.059	.181
Flacouortia montana - Flacourtiaceae	63 (16)	+ ve	.299	.001
Lagerstroemia lanceolata - Lythraceae	10 (15)	No	.011	.571
Mangifera indica - Anacardiaceae	18 (26)	No	.021	.434
Nothopegia beddomei - Anacardiaceae	428 (28) ‡	+ ve	.149	.029
Olea dioica - Oleaceae	105 (29) ‡	+ ve	.177	.017
Syzygium gardnerii - Myrtaceae	21 (22)	No	.028	.356
Toona ciliata - Meliaceae	21 (21)	No	.020	.438

Note: ‡ indicates species with high stem densities.

3.1.4 Conservation of tree diversity

Two prominent conservation interest groups can be identified, whose aspirations are to conserve different elements of tree diversity.

- *Local communities* are interested in maintaining diversity and numbers of those tree species that are of some use. Tropical forests are known to provide a large range of wood and non-wood products that are important for the local economy (Myers 1988). In a study in the northeast of Peru (Pinedo-Vasquez *et al.* 1990), 60% of tree species were found to be useful to the local people for food, construction, craft, medicine *etc.* In the Western Ghats, as elsewhere in the tropics, rural livelihoods depend on the trees found in the neighbouring forest.

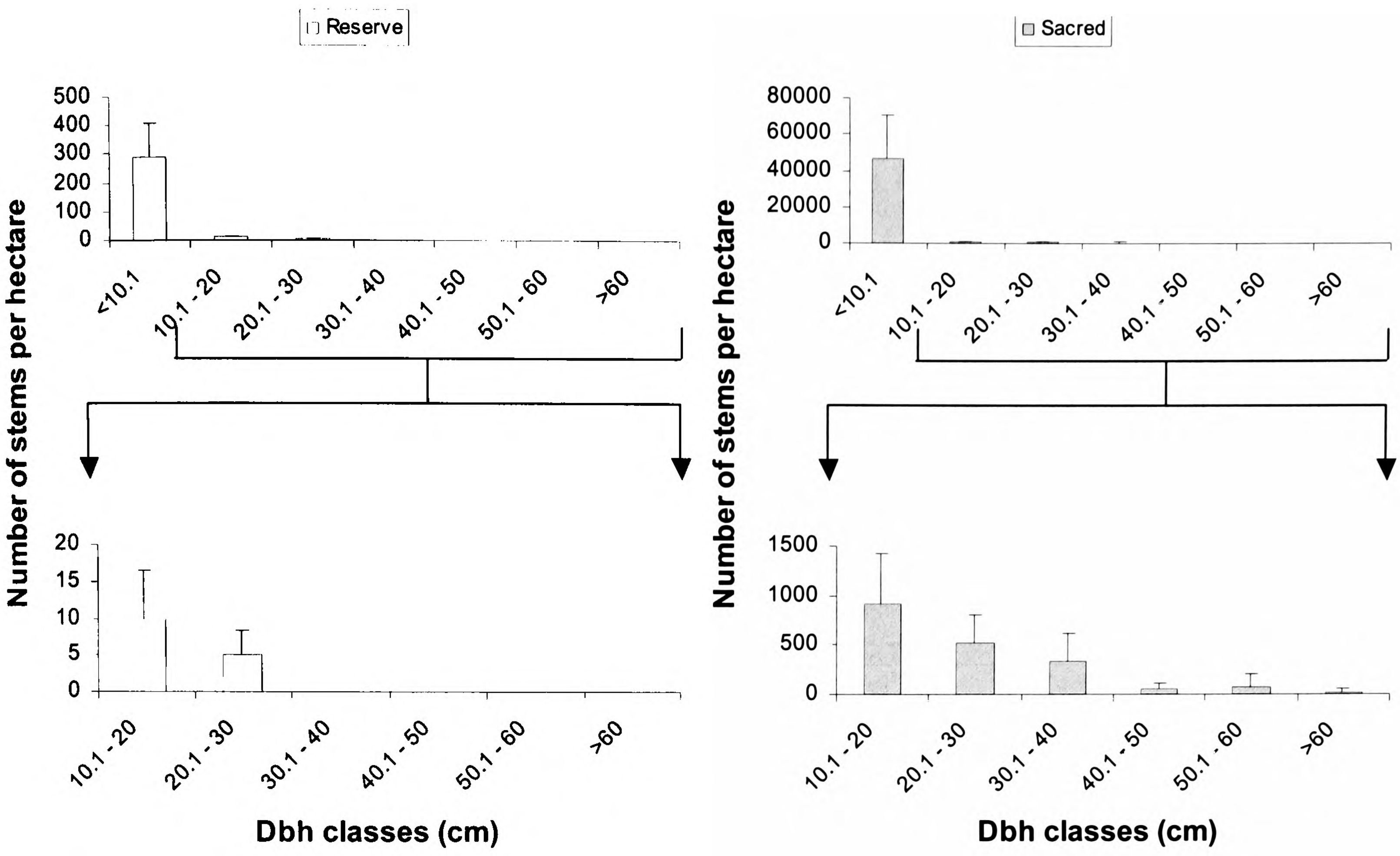


Figure 3.13 Size-class distributions of Nothopegia beddomei, *an evergreen tree, in the forest reserve localities (n=6) and sacred forests (n=22); note the difference in the scales on Y axis for the reserve and sacred forests - the numbers of stems of* Nothopegia beddomei *in the lowest size-class are more by a magnitude in comparison with that in the reserve. Note: error bars indicate 95% confidence intervals.*

- *The International community* and NGOs such as IUCN and WWF are interested in species that are known to be facing extinction. Thus, tree species that are threatened by activities such as land-clearance, as well as those restricted in their distributions to certain small biogeographical regions are currently of international conservation concern (IUCN 2000). The Western Ghats has a large number of endemic plants and animals that are under threat due to rapid deforestation and habitat destruction (Myers *et al.* 2000).

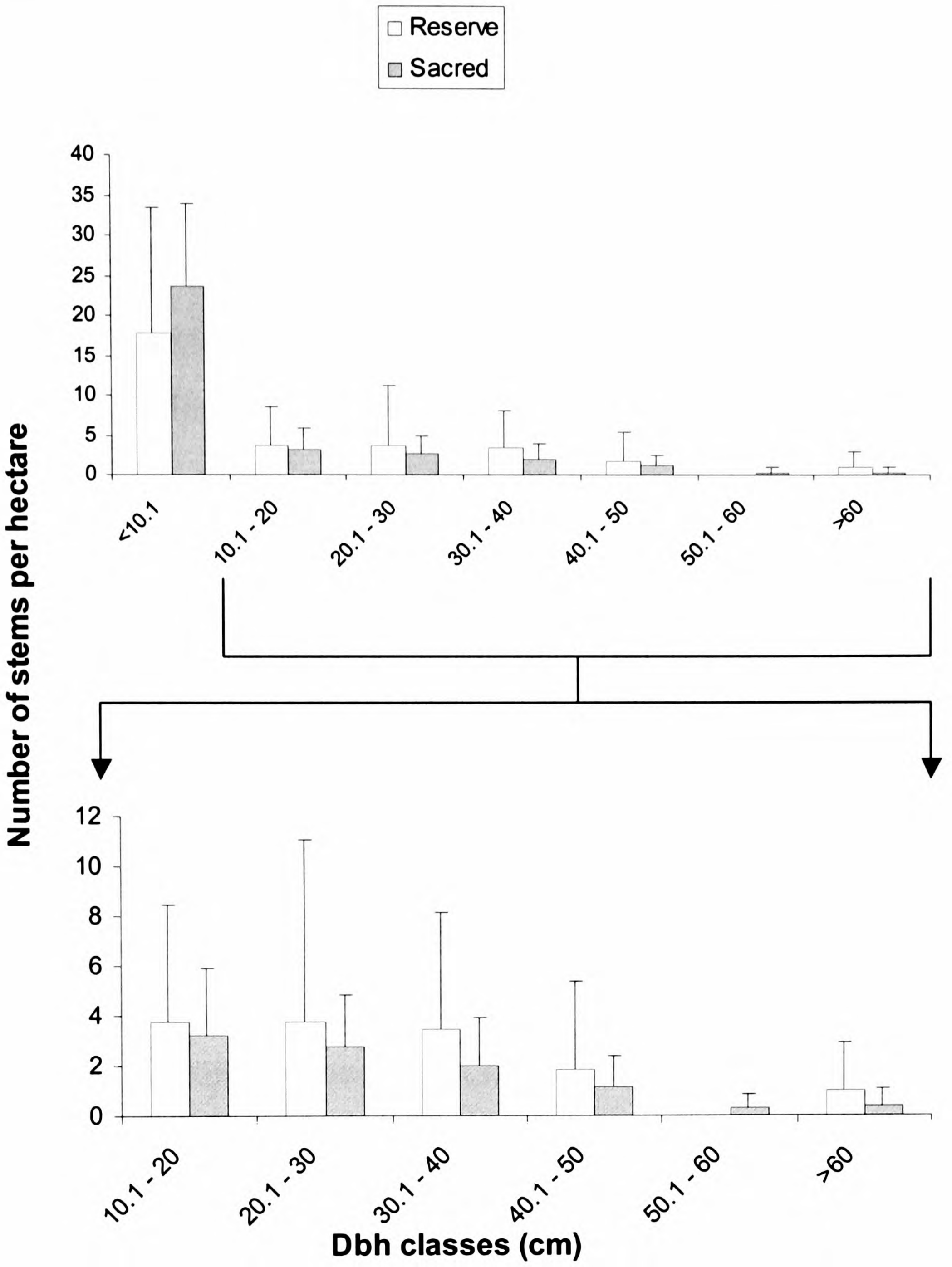


Figure 3.14 Size-class distributions of Toona ciliata, *a deciduous tree, in the forest reserve localities (n=4) and sacred forests (n=17); the numbers of stems of* Toona ciliata *in all size-classes are not significantly different in sacred forests and the reserve. Note: error bars indicate 95% confidence intervals.*

Trees useful to the local communities

Species were classified either as useful to local people or not useful, in order to examine in which part of the landscape useful species persisted best. Two categorisations were used:

1. The Wealth of India (Council for Scientific and Industrial Research 1989) database was used to determine the information about the usefulness of a species. A total of 70% of species in the Kodagu area yield useful non-timber forest products, 30% of

those are also used as timber. However, the uses of such species are limited to cottage industries that make carts, other agricultural implements and craftworks (Council for Scientific and Industrial Research 1989). Although such species have timber value, in Kodagu they are not exploited by large-scale timber industries unlike in many other tropical regions, where use of trees as timber is important. Thus, as far as the utility of trees is concerned, non-timber uses of trees are more prominent in Kodagu than uses as timber.

2. The medicinal plants database prepared by the Bangalore-based Foundation for Revitalisation of Local Health Traditions (FRLHT 1999) was used to categorise species into those with known medicinal properties and those without, in order to determine where most are found. Herbal medicines have been used in India for numerous generations. The local herbal medicine system is extensive – India has about 7000 well-documented medicinal plant species (FRLHT 2000b). Just as elsewhere in India, medicinal trees in Kodagu are of great value to the local people.

Useful trees were significantly more abundant in coffee plantations (Kruskal-Wallis test, $\chi^2 = 7.553$, df = 2, $P = 0.023$) while those that had no known use were significantly more abundant in the forest reserve (Kruskal-Wallis test, $\chi^2 = 7.224$, df = 2, $P = 0.027$). Medicinal trees were also significantly more abundant in coffee plantations (Kruskal-Wallis test, $\chi^2 = 6.992$, df = 2, $P = 0.030$) in comparison with the forest reserve and sacred forests.

Cinnamomum macrocarpum is a tree that yields valuable NTFPs such as the bark, which is used in spices. It is also used in medicinal preparations and is listed by FRLHT (1999) as a species that has priority for conservation.

Table 3.7 The distribution of useful trees in the Kodagu area (Figures in brackets indicate percentages in the respective land use type).

Land Use Type	(A) DISTRIBUTION OF SPECIES Useful trees ≥ 10 cm dbh			
	Utility value		Medicinal value	
	Useful	No known use	Medicinal	Not medicinal
Forest reserve	83 (62)	50 (38)	32 (24)	101 (76)
Sacred forests	113 (67)	55 (33)	42 (25)	126 (75)
Coffee plantations	118 (73)	44 (27)	48 (30)	114 (70)

Land Use Type	(B) DISTRIBUTION OF INDIVIDUALS Useful trees ≥ 10 cm dbh			
	Utility value		Medicinal value	
	Useful	No known use	Medicinal	Not medicinal
Forest reserve	706 (58)	519 (42)	282 (23)	943 (77)
Sacred forests	1680 (63)	992 (37)	626 (23)	2046 (77)
Coffee plantations	1900 (70)	829 (30)	732 (27)	1997 (73)

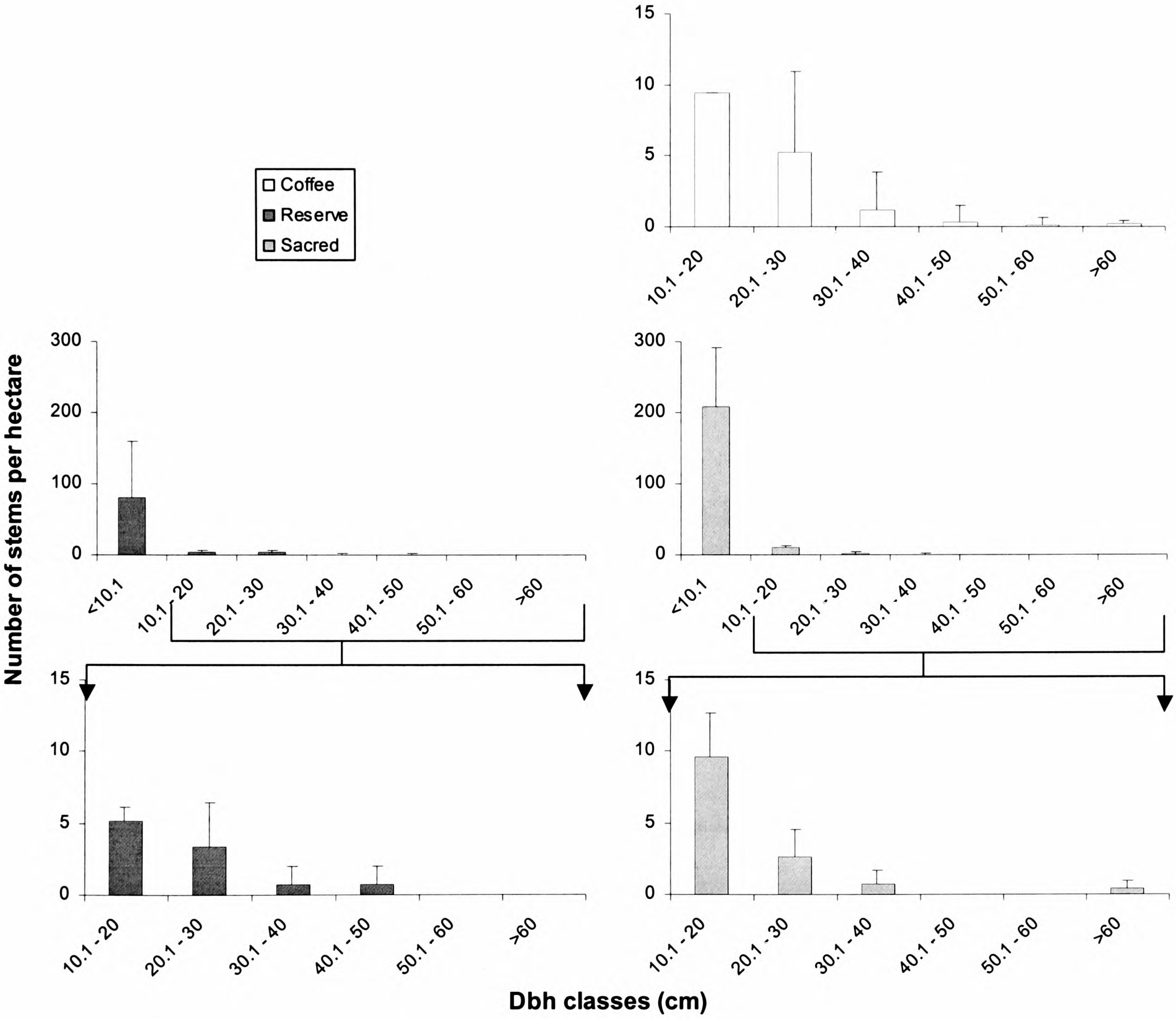


Figure 3.15 Cinnamomum macrocarpum, *which is medicinal and also yields other valuable non-timber forest products, regenerates better in sacred forests (n=19). The trees in the largest size-classes are retained in coffee plantations (n=18) while they are absent from the forest reserve (n=5). Note: error bars indicate 95% confidence intervals.*

Although the tree is well distributed over the entire landscape, the differences in its selective use and retention are reflected in the existing size-class distributions. Large size-classes are found only in coffee plantations, presumably because they are retained by the landowners for periodic harvesting of the bark, which fetches good market prices. The species generally occurs at a high density (*Table 3.6*) and regenerates better in sacred forests than in the reserve (*Figure 3.15*).

Trees of international conservation concern

There are high levels of endemism in the Western Ghats forests, reaching 63% of species in the southern part of the Ghats (Pascal & Pelissier 1996). The proportion of species endemic to the Western Ghats was examined in different land use types, based on the Atlas of Endemics prepared by Ramesh *et al.* (1997). Trees were also classified as non-threatened and locally or globally threatened species based on the IUCN threat categories (IUCN 2000) (*APPENDIX 4*). Their occurrence in different land use types was examined.

Table 3.8 Distribution of trees of international conservation importance in Kodagu (Figures in brackets indicate percentages in the respective land use type).

Land Use Type	**(A) DISTRIBUTION OF SPECIES Trees (≥ 10 cm dbh) of conservation importance**			
	Endemicity		**Threat status**	
	Endemic	**Non-endemic**	**Threatened**	**No threat**
Forest reserve	49 (37)	84 (63)	25 (19)	108 (81)
Sacred forests	47 (28)	121 (72)	32 (19)	136 (81)
Coffee plantations	39 (24)	123 (76)	27 (17)	135 (83)

Land Use Type	**(B) DISTRIBUTION OF INDIVIDUALS Trees (≥ 10 cm dbh) of conservation importance**			
	Endemicity		**Threat status**	
	Endemic	**Non-endemic**	**Threatened**	**No threat**
Forest reserve	487 (40)	738 (60)	289 (24)	936 (76)
Sacred forests	799 (30)	1873 (70)	890 (33)	1782 (67)
Coffee plantations	621 (23)	2108 (77)	631 (23)	2098 (77)

Endemic trees were significantly more abundant in the forest reserve (Kruskal-Wallis test, $\chi^2 = 12.754$, df = 2, $P = 0.002$) while non-endemic trees were significantly more abundant in coffee plantations (Kruskal-Wallis test, $\chi^2 = 8.306$, df = 2, $P = 0.016$). Threatened trees were significantly more abundant in sacred forests and non-threatened ones were significantly less abundant (Kruskal-Wallis test, $\chi^2 = 11.465$, df = 2, $P = 0.003$) in comparison with the forest reserve and coffee plantations. *Nothopegia beddomei* is an example of a Western Ghats' endemic tree that is also listed as threatened in its home range (IUCN 2000). The tree regenerates better in sacred forest patches than in the forest reserve (*Figure 3.13*).

3.2 Discussion

3.2.1 Distribution of tree diversity in the landscape

Despite the long-term landscape modification in Kodagu, there are no significant differences in tree species diversity (calculated using rarefied species richness for 50 individuals $\geq$ 10 cm dbh) across land use types (*Figure 3.3*) and a large proportion of tree species are ubiquitous (*Figure 3.1*). Turner *et al.* (1994, 1996) and Corlett & Turner (1997) have speculated that trees are very slow at responding to habitat fragmentation due to their long generation times. In their studies, many fewer tree species (29%) were found to have become extinct from the 4 ha fragment of the Botanic Garden in Singapore than the predicted extinction of 76% species by the species-area relationship, despite isolation for more than century. The landscape in Kodagu seems to have retained much of the original tree diversity. As seen from *Figure 3.1*, a large proportion of species (45%) have been able to maintain their populations in both coffee plantations, sacred forests and the reserve. Only 9% are restricted to the contiguous forest and another 9% to the forest patches. Such habitat-specialists are absent from coffee plantations.

A look at the unique species shows the nature of the tree species that may have been lost due to centuries of modification in the Kodagu landscape. All 19 species unique to the forest reserve are evergreen and 63% of them are endemic to the Western Ghats (*Table 3.5*). In sacred forests, 45% of the 20 unique species are deciduous, the rest are evergreen and 30% of the unique species are endemic. In coffee plantations, 83% of the unique species are deciduous; three species, *Cytheroxylon subserratum, Glyricidia maculata* and *Psidium guajava* out of the total 24 are exotic (planted by owners) while two (8%) are endemic to the Western Ghats. Thus, the long-term landscape modification has caused a disproportionate decline in evergreen and endemic species in Kodagu.

A χ^2 test reveals that the unbroken forest reserve has more unique species than expected while the sacred forests have fewer. This could be because of the effects of management by man causing loss of species that have restricted or patchy distributions (Wilcox 1980). Such species might have occurred in the landscape before its modification started.

3.2.2 Effects of landscape modification on tree diversity

Tree species diversity was not significantly different across the three land use types as seen from *Figure 3.3*. Therefore, apparently, land management has had no effect on overall diversity of trees in the Kodagu landscape.

Effect of patch area

Hill & Curran (2001) report that patch area accounted for 92% of the variation in tree species richness in fragmented forests of Ghana. However, patch area had no effect on tree diversity in the present study (*Figure 3.4*). This can be explained by the nature of the vegetation surrounding sacred forest patches. In Ghana, the fragmented forests were surrounded by predominantly agricultural land whereas in Kodagu the sacred forest patches were surrounded by coffee plantations, where tree-cover is retained to provide shade for coffee plants. The highly tree-covered landscape means that patches are not well defined, since the boundaries between patches and the surrounding plantations are often indistinguishable. The lack of an association between tree diversity and patch area may be because of the high tree-cover in Kodagu. Consequently, the area of forest that can host forest-dependent species is not limited by patch edges. Non-forest species cannot establish themselves due to the lack of edge habitat. As a result, patch area explains little of the variation in species diversity. It is likely that the factors influencing tree diversity operate at scales larger than that of a patch.

Landscape effects

It has been shown in the wet tropics that deciduous plants increase proportionally as a result of fragmentation (*e.g.* Bierregaard *et al.* 1992; Kellman *et al.* 1996; Laurance *et al.* 1998b) and the consequent changes in the local micro-climate (Kapos *et al.* 1997; Condit & Markham 1998; Laurance *et al.* 1998a; Condit *et al.* 2000). Hill & Curran (2001) found in fragmented forests of Ghana that, as the area of forest fragments increased, the relative proportion of evergreen and shade-tolerant species increased with respect to deciduous pioneers. In other words, small patches were left with comparatively large edges and a high proportion of edge-dwelling deciduous species.

In response to microclimatic changes (such as an increase in photosynthetically active radiation and temperature, a decrease in relative humidity and soil moisture) associated with land modification in Kodagu, disturbance-adapted deciduous species increase in relation to evergreen species (*cf.* Daniels *et al.* 1995). A similar phenomenon was observed by Laurance (1997a) in the isolated rainforest reserves in Australia, where disturbance-adapted rattans and lianas were found to respond positively to edge effects of fragmentation. Also, in another study in Amazonia (Laurance *et al.* 1998b), successional trees were found to have increased in older and smaller fragments and in plots near forest edges due to:

- Recurring disturbances providing light and microclimatic conditions favouring their growth (Clark 1990); and
- Heavy seed rain from pioneer trees in the surrounding landscape matrix (Janzen 1983).

While landscape fragmentation results in the creation of gaps and edge effects, land management results in habitat modification. Both processes favour disturbance-adapted species. Because of high tree-cover in the Kodagu landscape, edge effects were not prominent. However, land modification appears to have effects on microclimate similar to those in fragmented landscapes (Kapos *et al.* 1997; Freidenberg 1998). As a consequence of several centuries of land management in Kodagu, the phenological characteristics of the tree community have altered over time; the intensity of land management determines the proportions of deciduous and evergreen trees (*Figure 3.5*).

A decline in diversity was observed in isolated patches (*Figure 3.7, A*), while the density of stems ≥ 1 cm dbh increased significantly with isolation (*Figure 3.7, B*). In addition to the distance of the patches from the reserve (*Figure 3.8*), other factors such as a significant rainfall gradient, particularly in the region more than 5 km away from the forest (*Figure 3.11*), a slight east-west gradient in topography, soils and climate (*Figure 2.4*), as well as human disturbance at localities farther away from the reserve may cause a decline in diversity. Such decline means reduced number of species in isolated patches. This favours species that are adapted to disturbance such as *Nothopegia beddomei* (*Figure 3.13*). It is a common observation in fragmented landscapes that the loss of species from fragments allows persisting species to achieve unusually high densities (Laurance *et al.* 1997). Turner *et al.* (1996) found in a 4 ha tropical forest fragment of Singapore Botanic Garden that species such as *Calophyllum ferrugeineum* were the commonest and contributed 23.9% of the total number of stems compared with only 2.3% and 3.7% in mixed dipterocarp forests at Pasoh in Peninsular Malaysia respectively. Although this species is common in old secondary forests of Singapore, it does not require gaps in the forest canopy for seedling establishment; dense carpets of *Calophyllum ferrugeineum* seedlings were found in deep shade in the botanic garden's jungle (Turner *et al.* 1996). *Nothopegia beddomei* shows a similar pattern in Kodagu.

The unusually high abundances of certain species in the community are reflected in the steep species-abundance curve for forest patches that were isolated by five or more kilometres from the forest reserve (*Figure 3.9*, *Figure 3.8*). In addition to the adaptation of some species to disturbance, certain others are limited by dispersal and are unable to reach isolated patches. As a result, the composition of species changes in isolated communities, making them dissimilar to the community in unbroken forest. In the present study, community similarity was found to decline with distance (*Table 3.3*, *Figure 3.10*) as revealed by Sorenson's similarity index and non-metric multidimensional scaling of species-abundance distributions in sacred forest patches at various distances from the reserve. Non-metric multidimensional scaling of individual sampling localities in the reserve and sacred forests showed that there was no ordering in sacred forests, while most localities from the forest reserve clustered together (*Figure 3.12*). However, this may be because the distribution of forest reserve sites was more clumped than sacred forests (*Figure 3.8*). Sacred forests, that do not show any clustering, may be dissimilar to the forest reserve and to each other – a greater heterogeneity of tree species assemblages may protected in sacred forests than in the reserve. Coffee plantations in the patchy landscape together with sacred forests protect a higher number of species than in the forest reserve alone (*Figure 3.1*). This suggests that the patchy landscape in Kodagu may be important for maintaining tree diversity and valuable for conservation.

3.2.3 Approaches to conservation of tree diversity

This study provides a clue as to the most effective ways of maintaining tree diversity in Kodagu. Those elements of the landscape that are important for the maintenance of useful species and of species that are of international conservation concern, can be identified and strategies for meeting the conservation interests of the local people and the international community in Kodagu, defined.

The forests of Kodagu have been used by the local people for many centuries and they all show a strong human influence. Although useful trees are extracted from the landscape, the low-impact extraction appears to have been one of the factors responsible for maintaining a large proportion of trees of conservation importance. Daniels *et al.* (1995) found that populations of habitat-specialist evergreen trees that are useful to the local people were not affected while the populations of commercially exploited species had suffered severely in the Western Ghats of Uttara Kannada district, 500 km north of Kodagu. In Kodagu, species such as *Cinnamomum macrocarpum* that yield valuable non-timber forest products have been maintained in coffee plantations (*Figure 3.15*).

Endemics are believed to be vulnerable to extinction (Bierregaard *et al.* 1997) because of their restricted distributions. A large number of endemic trees in the Western Ghats are also evergreen (Ramesh & Pascal 1997). For instance, all 19 tree species unique to the forest reserve are evergreen, 12 (63%) of which are also endemic to the Western Ghats (*Table 3.5*). In addition, evergreen trees are relatively more abundant in the forest reserve than in the patchy landscape (*Figure 3.5*). The forest reserve provides an uninterrupted habitat for evergreen and endemic trees in comparison with the patches. Thus, strict protection of large contiguous tracts of forest is desirable for such species.

A study by Colding & Folke (1997) attributes the occurrence of species threatened in their native ranges to the traditional systems of conservation and social restraints on their utilisation. Such systems are believed to occur in several modern societies (Gadgil *et al.* 1993). Although there is no evidence of social restraint or conscious effort for maintaining threatened tree species in Kodagu, the sacred forests protected on the basis of religious beliefs contain assemblages of species of conservation importance.

In summary, while species are lost from isolated forest patches in the managed landscape of Kodagu, modern land uses harbour populations of useful trees. Traditional systems of conservation are still valuable for offering protection to certain elements of tree diversity including threatened trees. On the other hand, trees such as those endemic to the Western Ghats require large contiguous areas in the forest reserve. The resource availability in the already modified landscape in the form of useful tree species may be important in removing the pressure of resource use from the forest reserve to the traditionally managed land, thus achieving a better conservation of a strictly protected area. A combination of approaches is therefore necessary for conservation of tree diversity in Kodagu.

3.3 Conclusions

3.3.1 Implications for conservation practice

The species-rich communities of trees in Kodagu are best maintained in the less isolated, least disturbed forest patches that are surrounded by a continuous tree-cover (*Figure 3.6*). However, a closer look at species groups and individual species suggests that some species are persisting in the landscape, largely in isolated patches. For instance, *Nothopegia beddomei*, which is also of international conservation concern, regenerates profusely in isolated sacred forests (*Figure 3.13*). Sacred forest patches together with coffee plantations in the surroundings host a greater variety of species assemblages than in the forest reserve (*Figure 3.1*) contributing to the landscape-level heterogeneity (Lapin & Barnes 1995), which may be of value to other forms of wildlife, such as butterflies. For example, according to a study by Soubadra Devy & Ganeshaiah (2001 *pers. comm.*), about 46% of the butterfly species in the Western Ghats were in low abundance in a few patchily distributed locations.

Considering the complex land tenure system in Kodagu (Vijaya 2000) (*APPENDIX 2*), it is difficult to recommend one single measure for protecting tree diversity. A suitable conservation strategy has to be decided and planned through the initiatives of the local people. The preferred strategy could be different for different villages. General principles that should be followed in the choice of any strategy include the following:

- It is essential to recognise the importance of the contiguous forest as well as the forest patches. The findings of this study suggest that both these elements of the Kodagu landscape are important for conservation of tree diversity for different reasons. The forest reserve is important for maintaining species-rich communities of habitat-specialist trees while sacred forest patches are important for maintaining landscape level heterogeneity that may be valuable to other forms of wildlife.

- The maintenance of tree cover of native species in coffee plantations is vital for ensuring long-term protection of forest tree diversity in Kodagu. In the absence of tree cover, species may be lost at a faster rate than at present, as experienced elsewhere in the tropics (Laurance *et al.* 1997). Maintenance of native trees in the landscape can facilitate pollination and dispersal of certain species, even in isolated patches. In addition, a high tree-cover will ensure less prominent edge effects thus avoiding the decline in diversity from patches (Kapos *et al.* 1997).

- Human disturbance in sacred forests and selective preference in coffee plantations is necessary for the regeneration and persistence, respectively of certain useful native tree species such as *Cinnamomum macrocarpum* (*Figure 3.15*). Such maintenance of useful species can be a key to diverting resource-use pressures from strictly protected areas to the already modified landscape, ensuring a better protection of the forest reserve.

3.3.2 Ideas for future research

The focus on the study of trees in the present research has been more on the maintenance of tree diversity than on management of individual tree species. This study can provide an empirical example for future research aimed at conserving tree diversity. The species-specific problems that are important in investigating tree communities in fragmented landscapes and are not addressed in this study are listed below. These are limitations of the present investigation, and it is recommended that future research in Kodagu address them:

- Available data on biologically rare or habitat-specialist trees that are known to be vulnerable to fragmentation were not sufficient for analysing quantitatively their minimum viable populations in the present study. For instance, Hawthorne (1996) recognised those tree species in Ghana that were rare and restricted to riverside or uplands. More intensive data collection, specifically on such species is necessary in order to estimate population sizes and to determine which conservation strategy is most effective for their survival. Future research on tree diversity in Kodagu should focus on the distribution of such species in the landscape and approaches that can conserve them the best.

- Pollination and dispersal are two important processes in fragmented landscapes that can affect the ability of a tree to survive. In the present study, information about pollination strategies and dispersal abilities of tree species (*e.g.* Corlett 1998) was not available. In future, it will be interesting to examine tree diversity patterns in this context. For instance, Hammond *et al.* (1996) compiled seed sizes and dispersal strategies in Guianan timber trees as a basis for recommending appropriate management for sustainable timber harvest. In Kodagu, such information will be useful for scientific management of sacred forest patches. In addition, the sexual system (hermaphrodite, monoecious or dioecious) of the particular tree species, is important in determining the ways in which it will be affected in a fragmented landscape (*e.g.* Bawa & Ashton 1991). Future research on tree diversity in Kodagu should consider such species-specific problems and investigate the consequences of landscape modification for their relative success.

- Local people in Kodagu are undoubtedly dependent on native tree species. For instance, *Cinnamomum macrocarpum* is a tree that yields important non-timber forest products and regenerates well in sacred forests (*Figure 3.15*). It may be possible for local people to collect non-wood products from *Cinnamomum* in sacred forests without causing harm to its natural populations. While this study presents the distribution of diversity of useful trees and those of conservation importance, it does not comment on their sustainable harvest levels (*e.g.* Ganesan 1993). Conservation management of trees in Kodagu landscape will require such information.

4 . BIRD DIVERSITY

The effects of landscape modification on bird diversity in Kodagu are examined in this chapter.

4.1 Results

4.1.1 Distribution of bird species in the landscape

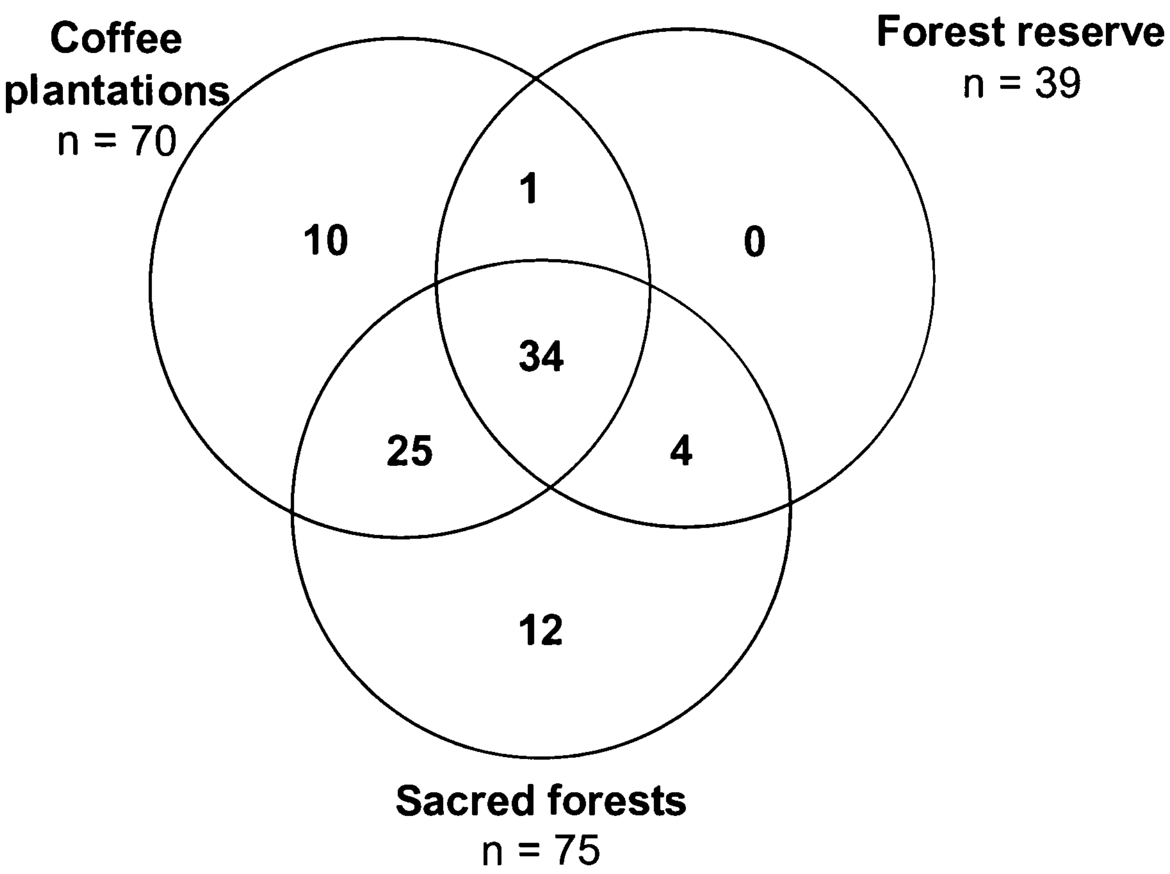


Figure 4.1 Numbers of unique and shared bird species (Total number of species sample, N = 86). This analysis is based on species samples obtained from 58 sampling localities – 10 from the forest reserve, 25 sacred forests and 23 coffee plantations.

A total of 86 bird species were recorded (*APPENDIX 5*). Although more extensive field surveys are necessary to affirm the presence or absence of a species, the distribution of species in the three land use types, as observed in the present study, is shown in *Figure 4.1*. Species that were found only in a single land use type are referred to as 'unique species', those shared by any two of the three types are 'shared species' and those found in all three types are referred to as 'ubiquitous species'. Nearly 40% of all species are ubiquitous, 35% are shared, and 25% are unique. The forest reserve does not have any unique species. The expected distribution of unique species was calculated by adjusting the species numbers to the sample sizes in the respective land use types. The observed numbers of unique species were not significantly different to the expected numbers (χ^2 test, $\chi^2 = 2.631$, df = 2, $P = 0.0977$) in any of the three land use types.

4.1.2 Effects of landscape modification on bird communities

Diversity of bird species

Rarefied species richness

Species diversity was adjusted according to sample size in order to calculate rarefied species richness at each sampling locality. Hurlbert's (1971) rarefaction method was used. The numbers of sampling localities that permitted calculation of rarefied species richness for a given observations of birds are shown in *Table 4.1*. The samples were rarefied to 15 observations because this number allowed the inclusion of all sampling localities in the analysis. Rarefaction curves for all sampling localities are displayed in *Figure 4.2*. In more than 90% of the localities, the increase in rarefied species richness was consistent with increasing number of observations (*Figure 4.2*). Therefore, although the species are rarefied to a smaller number of observations (15) due to the limitations of the data, this estimate can be reliably used as an index of diversity in the present study.

Table 4.1 Numbers of sampling localities that permitted calculation of rarefied species richness for birds

	Numbers of observations of birds to which samples were rarefied					
	10	*15*	*25*	*35*	*50*	*100*
Numbers of sampling localities that permitted calculation of rarefied species richness						
Reserve (n = 10)	10	10	10	10	1	0
Sacred (n = 25)	25	25	25	25	13	1
Coffee (n = 23)	23	23	22	19	7	1
Total (n = 58)	***58***	***58***	***57***	***54***	***21***	***2***

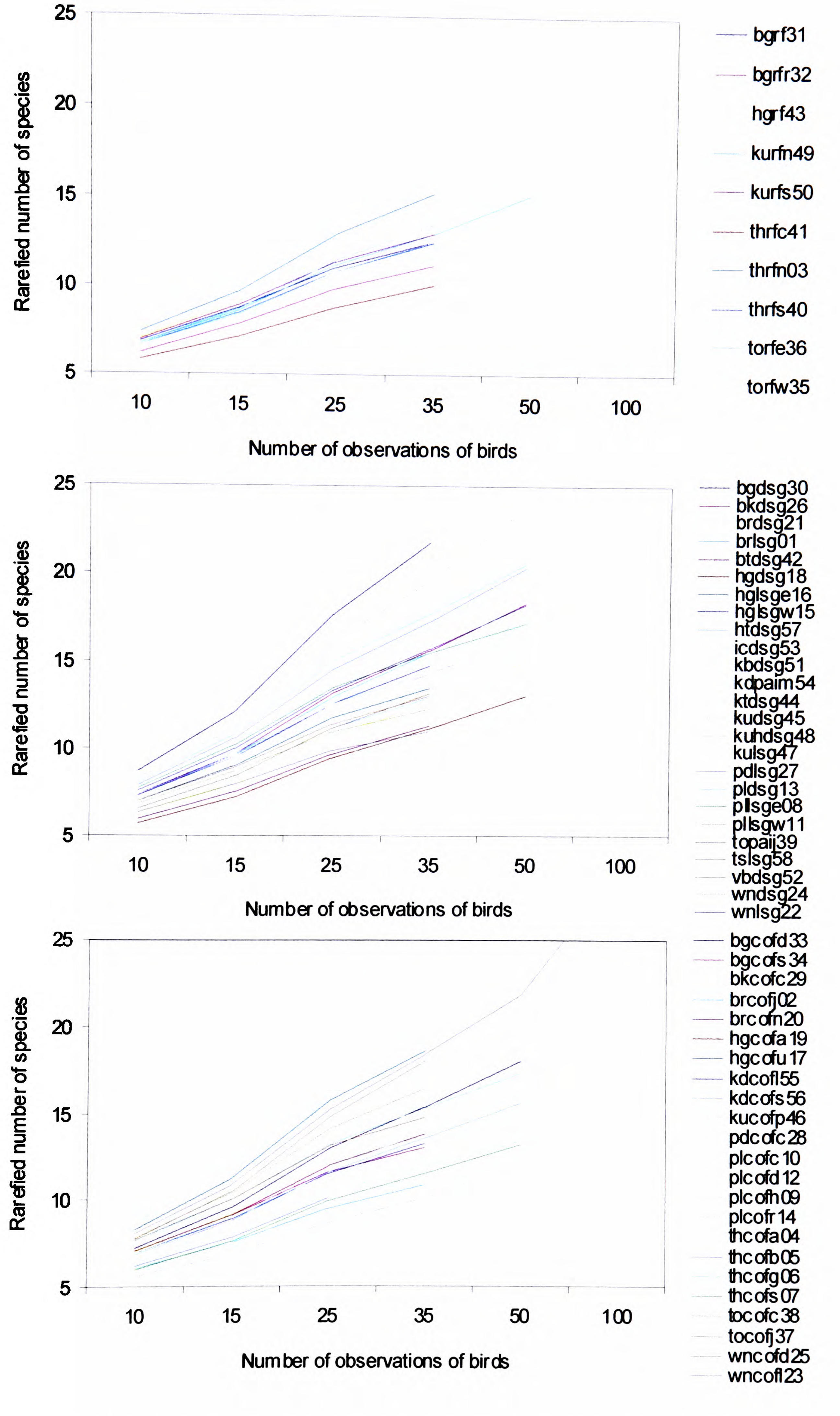


Figure 4.2 Rarefaction curves (birds) for sampling localities; the top panel shows forest reserve localities (n = 10), the middle panel shows sacred forests (n = 25) and the bottom panel shows coffee plantations (n = 23).

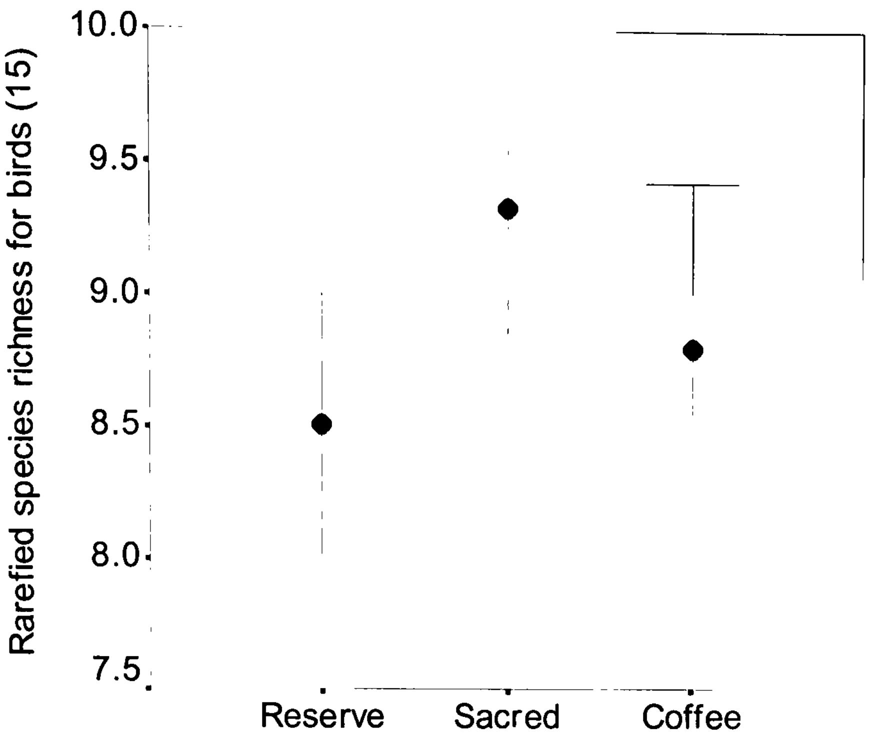


Figure 4.3 Mean rarefied bird species richness (for 15 observations) in three land use types; numbers of samples in the forest reserve, sacred forests and coffee plantations were 10, 25 and 23 respectively. Note: error bars indicate 95% confidence intervals.

There was no significant difference in the mean rarefied species richness (One-way ANOVA, F = 2.037, df = 2, $P = 0.140$) across the three land use types (*Figure 4.3*).

Effect of patch area

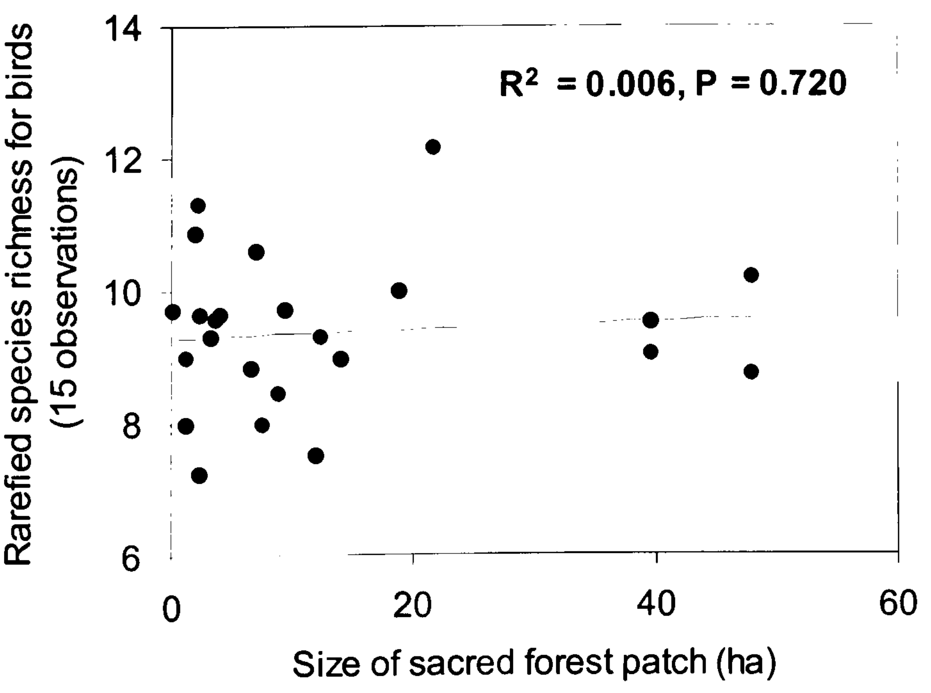


Figure 4.4 The relationship between the size of sacred forest patch (n = 24) and bird diversity.

Opdam *et al.* (1985), van Dorp & Opdam (1987), Warburton (1997) and Miller & Cale (2000) have found patch area a key factor in explaining bird diversity. In order to examine whether bird diversity in Kodagu is associated with patch size, rarefied species richness for birds in sacred forests were plotted against the sizes of different patches. No significant association was found between patch size and rarefied species richness per 15 observations of birds ($R^2 = 0.006$, $P = 0.720$) (*Figure 4.4*).

Landscape effects

Effect of isolation

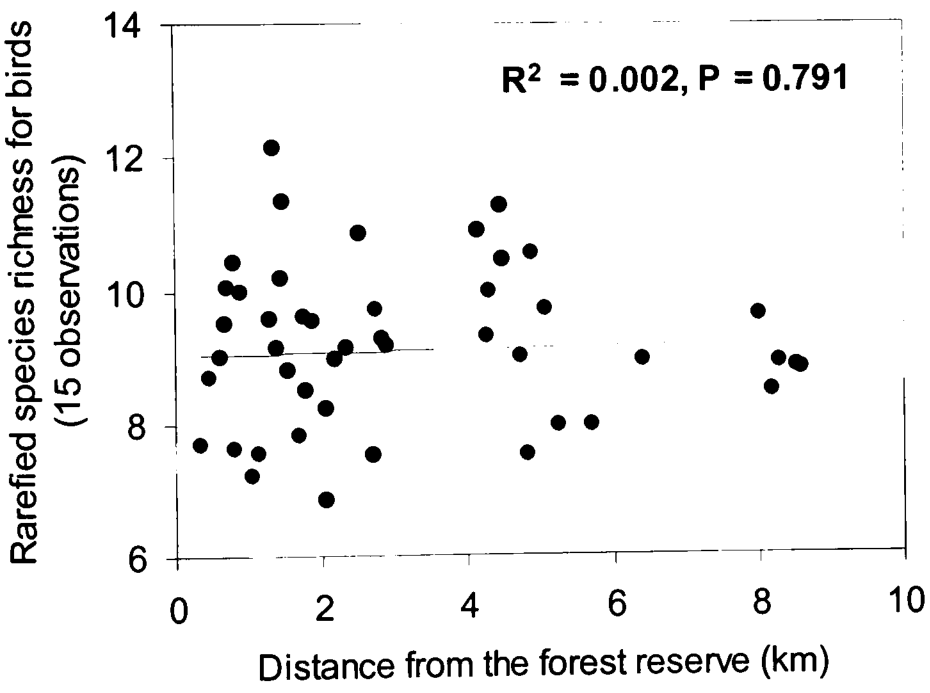


Figure 4.5 The relationship between the isolation of sampling localities (sacred forests, n = 25 and coffee plantations, n = 23) and bird diversity.

In addition to patch size, isolation is also believed to be important in explaining bird diversity in fragmented landscapes. Increasingly isolated patches have lower bird diversity (Opdam *et al.* 1985; van Dorp & Opdam 1987) because the distances of patches from the large forest may affect the frequency of movement of birds (Whitcomb *et al.* 1981; Lynch & Whigham 1984) and certain species may not be able to reach the isolated patches. In order to examine whether bird diversity in Kodagu is associated with isolation, the rarefied species richness for birds observed in sacred forests and coffee plantations were plotted against the distances of the respective sampling localities from the forest reserve. No significant association was found ($R^2 = 0.002$, $P = 0.791$) (*Figure 4.5*).

However, isolation may have an effect on the abundances of individual species depending on their ability to use the available habitat. For instance, nectarivorous birds are known for their ability to use a wider range of resources in fragmented landscapes than frugivorous birds (Thiollay 1994; Renjifo 1999). It is likely that isolation might have an effect on such species. Effects on bird species with certain ecological characteristics are examined later in the Chapter (*pp. 122-130*).

Effect of the intervening landscape matrix

CHAID identified tree-cover in the immediate surroundings of a sampling locality as the most significant ($P = 0.0038$) variable in determining bird diversity in the landscape (*Figure 4.6*). In addition, CHAID explained the residual variation in bird diversity by various factors related to forest structure, such as the variability in canopy height ($P = 0.002$), canopy closure ($P = 0.0437$) and basal area ($P = 0.0095$).

The results of forward stepwise regression were similar to that of CHAID for the most important variable – tree cover in the surrounding 250 m, although it explained only 9% of the variation (*Table 4.2*).

Figure 4.6 (Please see Page 118) CHAID diagram showing variables that are significant in explaining bird diversity. Full titles of the variables and the ranges of values they take are shown in boxes.

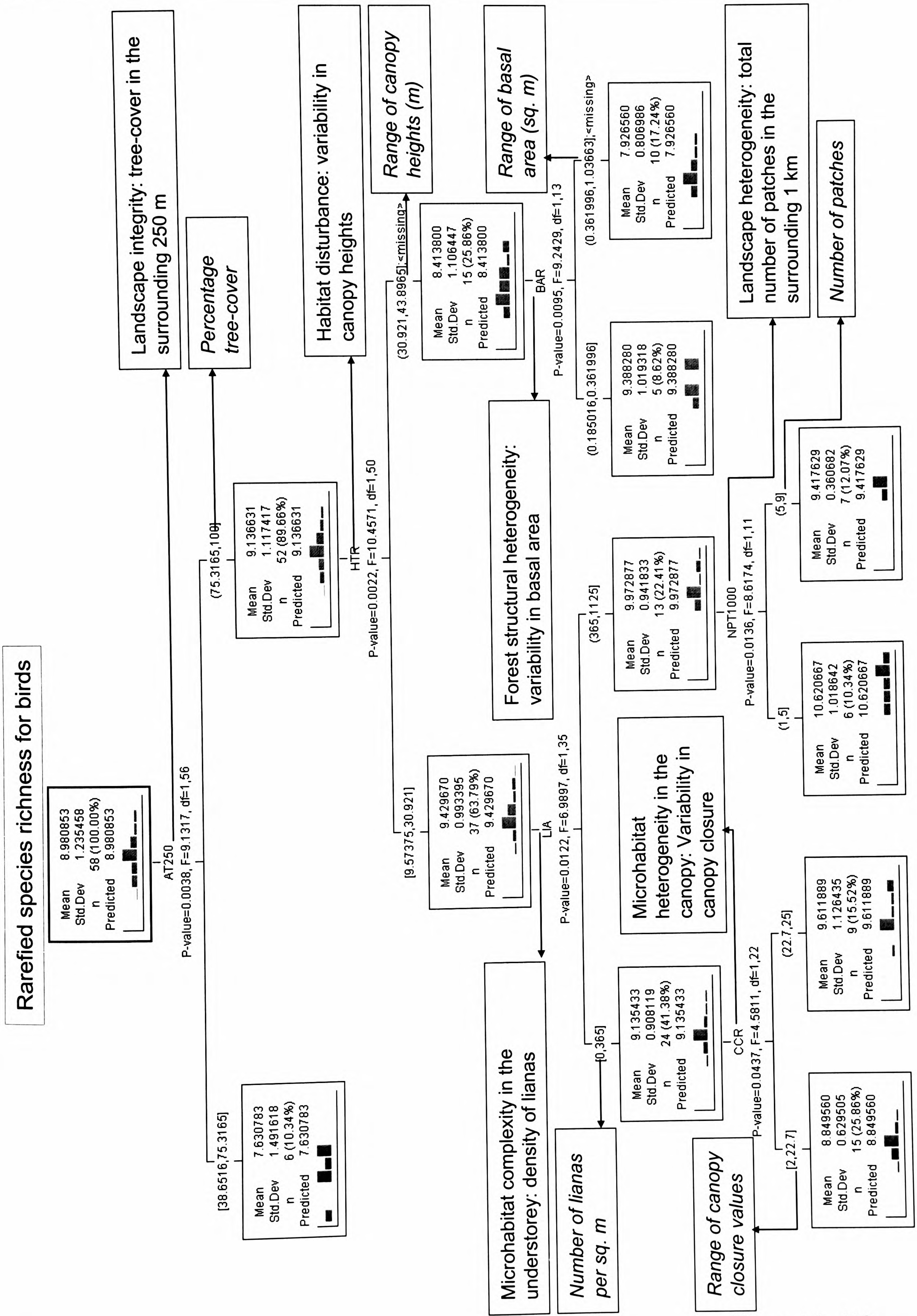

Rarefied species richness for birds
Mean 8.980853
Std.Dev 1.235458
n 58 (100.00%)
Predicted 8.980853
AT250
P-value=0.0038, F=9.1317, df=1,56
Landscape integrity: tree-cover in the surrounding 250 m
Percentage tree-cover
[38.6516,75.3165]
Mean 7.630783
Std.Dev 1.491618
n 6 (10.34%)
Predicted 7.630783
(75.3165,100]
Mean 9.136631
Std.Dev 1.117417
n 52 (89.66%)
Predicted 9.136631
HTR
P-value=0.0022, F=10.4571, df=1,50
Habitat disturbance: variability in canopy heights
Range of canopy heights (m)
[9.57375,30.921]
Mean 9.429670
Std.Dev 0.993395
n 37 (63.79%)
Predicted 9.429670
(30.921,43.8965];<missing>
Mean 8.413800
Std.Dev 1.106447
n 15 (25.86%)
Predicted 8.413800
LIA
P-value=0.0122, F=6.9897, df=1,35
Microhabitat complexity in the understorey: density of lianas
Number of lianas per sq. m
BAR
P-value=0.0095, F=9.2429, df=1,13
Forest structural heterogeneity: variability in basal area
Range of basal area (sq. m)
[0.185016,0.361996]
Mean 9.388280
Std.Dev 1.019318
n 5 (8.62%)
Predicted 9.388280
(0.361996,1.03663];<missing>
Mean 7.926560
Std.Dev 0.806986
n 10 (17.24%)
Predicted 7.926560
[0,365]
Mean 9.135433
Std.Dev 0.908119
n 24 (41.38%)
Predicted 9.135433
(365,1125]
Mean 9.972877
Std.Dev 0.941833
n 13 (22.41%)
Predicted 9.972877
CCR
P-value=0.0437, F=4.5811, df=1,22
Microhabitat heterogeneity in the canopy: Variability in canopy closure
Range of canopy closure values
[2,22.7]
Mean 8.849560
Std.Dev 0.629505
n 15 (25.86%)
Predicted 8.849560
(22.7,25]
Mean 9.611889
Std.Dev 1.126435
n 9 (15.52%)
Predicted 9.611889
NPT1000
P-value=0.0136, F=8.6174, df=1,11
Landscape heterogeneity: total number of patches in the surrounding 1 km
Number of patches
[1,5]
Mean 10.620667
Std.Dev 1.018642
n 6 (10.34%)
Predicted 10.620667
(5,9]
Mean 9.417629
Std.Dev 0.360682
n 7 (12.07%)
Predicted 9.417629

Figure 4.6

The subsidiary factors identified by the two analyses seem very different. It is possible that there is a linear association between diversity of birds and tree cover in the surrounding 250 m (as suggested by the stepwise regression), while the association with habitat variables, such as variability in canopy height (identified by CHAID) is non-linear. Although more detailed analyses of these associations are necessary in order to assess the ecological meaning of the statistically powerful CHAID analysis, CHAID appears to be a more insightful technique in comparison with stepwise regression for its ability to identify non-linear relationships.

Table 4.2 Forward stepwise regression model for bird diversity.

Step	**R^2**	**Variable**	**T value**	***P***
1	0.0877	Tree cover in the surrounding 250 m	2.24	0.031
2	0.2091	Tree cover in the surrounding 250 m	3.52	0.055
		Total number of patches in the surrounding 250 m	2.80	0.42

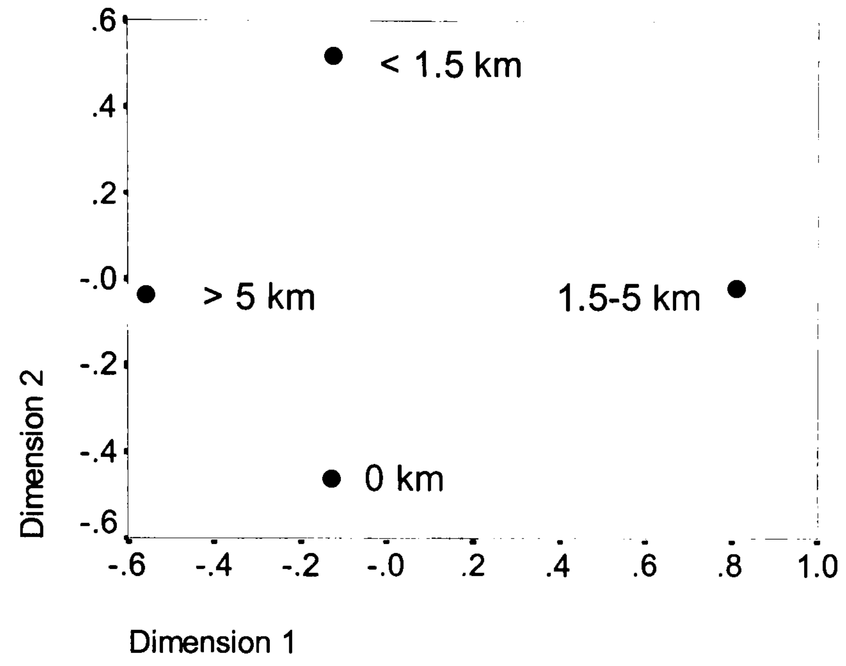
Similarity in bird diversity

Because of disproportionately high abundances of some disturbance-adapted species, a few bird species in a community can dominate species-abundance distribution (*e.g.* Bierregaard & Stouffer 1997). As a result, isolated fragments can become more and more dissimilar to the large forest with increasing distance.

In the present study, similarity in tree diversity was found to decrease with distance. In order to examine whether bird diversity followed the same pattern, Sorenson's quantitative similarity index (Magurran 1988), which gives an estimate of percentage similarity between rank-abundance distributions, was compared across groups of localities. Only 24 sacred forest patches and 8 forest reserve localities are considered in this analysis, while 1 sacred forest and 2 forest reserve localities (that do not have sufficient data for stems ≥ 1 cm dbh) are excluded so that the analysis is comparable to that for trees. The 32 sampling localities in the reserve and sacred forest patches were divided into four distance bands. Similarity in bird species composition between sacred forests and the forest reserve decreased with the distance of patches from the reserve (*Table 4.3*).

Table 4.3 Mean Sorenson's quantitative index showing a decline in similarity in bird diversity between the reserve (0 km) and the sacred forest patches (at < 1.5, 1.5-5, > 5 km away from the reserve).

	Sorenson's quantitative index			
Distance (km)	**0**	**< 1.5**	**1.5 - 5**	**> 5**
% Similarity	100	96	89	85
n	8	9	7	8



Euclidean Distances

Dimensionality: 2

	0 km	< 1.5 km	1.5-5 km	> 5 km
0 km	.000			
< 1.5 km	0.311	.000		
1.5-5 km	0.666	0.852	.000	
> 5 km	1.376	1.274	1.099	.000

Figure 4.7 Non-metric multidimensional scaling of similarity in bird diversity between the reserve (0 km) and sacred forest patches (at < 1.5, 1.5-5, > 5 km from the reserve).

Non-metric multidimensional scaling (Kenkel & Orloci 1986) assigns observations of community similarities (represented by points) to specific locations in a conceptual low-dimensional space such that the distances between points in the space match the given similarities as closely as possible. The results of non-metric multidimensional scaling are presented in *Figure 4.7*. The results did not correspond with the decrease in Sorenson's similarity along the distance gradient. According to the latter, sacred forests that are > 5 km away from the reserve were more similar to the reserve than those 1.5 - 5 km away and those < 1.5 km away.

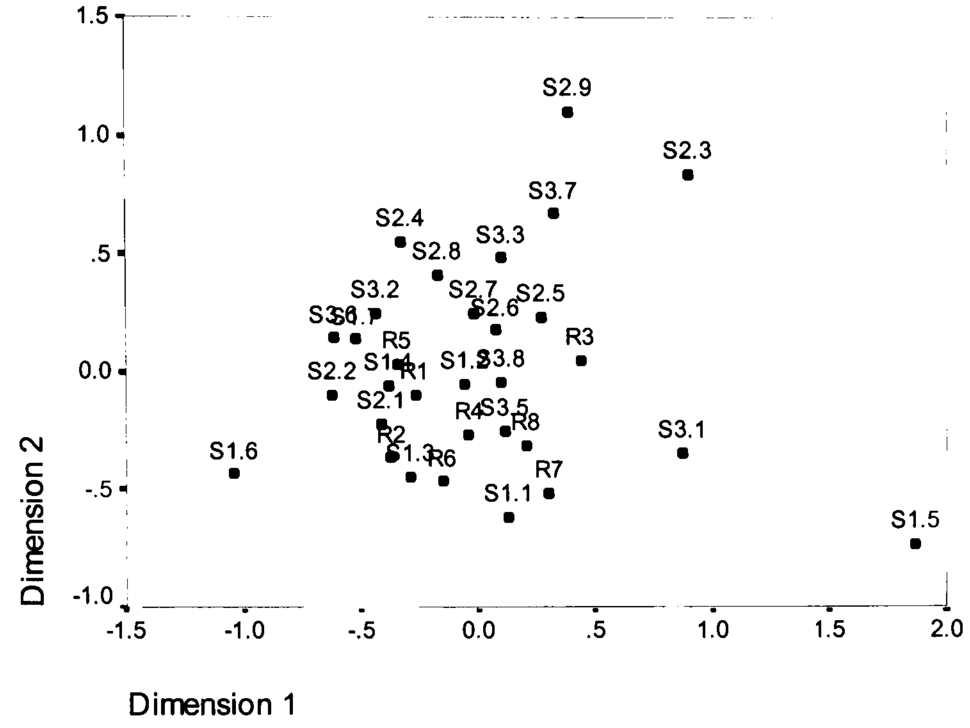


Figure 4.8 Non-metric multidimensional scaling of similarity in bird diversity between the localities in the reserve and sacred forest patches; each sampling locality is represented by a point. Those titled R belong to the forest reserve and those titled S belong to sacred forests. In each title, the number following S represents the distance band that the particular locality belongs to (1 corresponds with < 1 km, 2 corresponds with 1.5-5 km, 3 corresponds with > 5 km away from the forest reserve). Numbers following R and those at the end of titles beginning with S indicate ordering of localities within a particular distance band (e.g. S 2.2 indicates the locality in sacred forest that is 1.5-5 km away from the reserve and is the second nearest locality to the reserve in that distance band).

In order to examine how similar individual sacred forest patches are to each other and to the localities in the reserve, rank-abundance distributions of birds were subjected to non-metric multidimensional scaling (*Figure 4.8*). Coffee plantations were excluded from this analysis in order to avoid over-crowding of points in the two-dimensional representation shown in *Figure 4.8*. The forest reserve localities did not form a distinct cluster, and there was no ordering in the localities sampled (*Figure 4.8*) (S-STRESS = 0.02332). Locality S 3.4 was removed because it is very dissimilar to others as a result of sampling artefact (a large flock of Hill Mynas (*Gracula religiosa*) feeding in the sacred forest at the time of sampling).

Pair-wise comparisons of dissimilarity between forest reserve localities and sacred forests in three distance bands did not reveal any distinct pattern (*Table 4.4*).

Table 4.4 Pair-wise comparisons of dissimilarity in bird diversity between forest reserve localities and sacred forest patches in the Kodagu landscape; the values show Euclidean distances; larger the distance value, higher is the dissimilarity. Locations of sampling localities within the study area are mapped in Figure 3.8.

Locality	R1	R2	R3	R4	R5	R6	R7	R8
S1.1	6.708	6.325	6.325	4.899	6.928	6.164	4.123	5.196
S1.2	4.243	6.557	6.403	5.196	5.196	5.916	6.633	4.243
S1.3	4.359	3.742	7.211	4.243	4.243	5.477	5.916	6.083
S1.4	2.828	3.873	6.856	4.583	2.646	4.796	7.071	5.292
S1.5	20.174	19.494	14.697	18.111	20.396	19.596	16.462	18.358
S1.6	8.660	8.000	13.266	10.000	7.874	9.592	11.619	11.000
S1.7	4.583	5.292	8.246	5.477	3.464	6.481	8.426	7.280
S2.1	3.873	4.690	7.348	5.292	3.742	4.899	6.708	4.796
S2.2	3.742	3.873	8.775	5.745	3.873	5.385	8.485	7.211
S2.3	12.923	14.491	9.055	12.806	13.038	13.638	13.748	12.610
S2.4	6.856	7.616	7.874	7.071	5.831	8.246	9.000	8.426
S2.5	5.916	6.481	4.000	5.657	6.164	6.782	7.000	6.403
S2.6	4.359	5.657	4.000	4.690	4.690	6.164	6.856	5.745
S2.7	4.359	5.831	4.472	5.292	4.243	5.831	7.141	5.745
S2.8	4.899	6.856	6.403	6.403	4.796	6.557	9.055	7.211
S2.9	12.767	13.784	10.000	12.000	12.166	13.638	12.689	12.369
S3.1	10.536	10.488	6.782	9.274	11.136	9.899	7.141	7.280
S3.2	4.472	5.196	7.810	5.385	3.873	6.403	8.485	7.348
S3.3	6.481	7.937	5.385	7.000	6.245	7.810	8.944	7.348
S3.4				*Outlier removed*				
S3.5	4.583	5.831	5.657	4.690	5.477	5.657	5.196	3.873
S3.6	4.359	6.481	8.485	6.000	4.472	6.633	9.110	7.000
S3.7	9.055	10.630	7.000	8.185	8.660	10.440	10.198	9.487
S3.8	3.873	4.472	4.000	4.243	4.243	5.099	5.196	4.123

Ecological groups of bird species

Worldwide, birds form the most well studied group of organisms in terms of the effects of fragmentation on diversity (Turner 1996). Bird communities in the Western Ghats are no exception. Information about the natural history of most bird species is readily available through illustrated field guides (*e.g.* Ali 1996; Grimmett *et al.* 1998). This made the task of defining ecological species groups in birds much easier than in the case of trees, where they are often imprecisely defined (Swaine & Whitmore 1988). Ecological groups in birds were defined on the basis of three criteria:

- *Habitat preference*: It is useful in determining the ability of the species to survive in a disturbed landscape (*e.g.* Johns 1991; Greenberg 1996; Miller & Cale 2000).

- *Feeding guild*: It is important because the availability of food resources for a certain bird species, and the loss of such resources due to habitat modification can dictate the response of the species to the landscape scale processes (*e.g.* Thiollay 1994; Warburton 1997; Renjifo 1999).
- *Body size and fecundity*: They are often associated with the vulnerability of a bird species to extinction (Pimm *et al.* 1988; Gaston & Blackburn 1995; Greenwood *et al.* 1996) because habitat loss, fragmentation, natural predation and hunting can have different effects on birds having different body sizes and fecundities.

These criteria are important because landscape modification can also alter the available habitat, its quality and feeding resources for birds. Consequently, habitat-specialist birds can suffer from fragmentation while habitat generalist birds could be favoured (*e.g.* Bierregaard & Stouffer 1997). Thus, birds with specific feeding habits or those with large bodies and hence requiring more feeding resources may be vulnerable to landscape management by humans while those with omnivorous diets and small bodies may be more resilient.

Habitat preference

Declines in the proportions of forest-dwelling birds and increases in non-forest dwelling birds with increasing intensities of land management are commonly observed (Thiollay 1994; Warburton 1997; Renjifo 1999). Thiollay (1994) attributed this phenomenon in agroforests of Sumatra to the change in forest structure with increasing land management.

In the present study, habitat preferences of bird species were determined by prior knowledge or personal field observations and compared with the information in the field guides by Ali (1996) and Grimmett *et al.* (1998). Almost 80% of bird species (68 out of 86) in Kodagu prefer tree-covered to open habitats. This is also reflected in the proportions of birds in the three land use types (*Table 4.6*, *Figure 4.9*), where non-forest dwellers are present in very small numbers. Coffee plantations, with the most open canopies, have the highest proportion of non-forest dwellers, while the forest reserve with the most closed canopy has no non-forest dwelling birds. The proportion of forest-dwelling birds decreased with the intensity of land management while that of non-forest dwellers increased (One-way ANOVA, F = 5.183, df = 2, *P* = 0.009) (*Figure 4.9*).

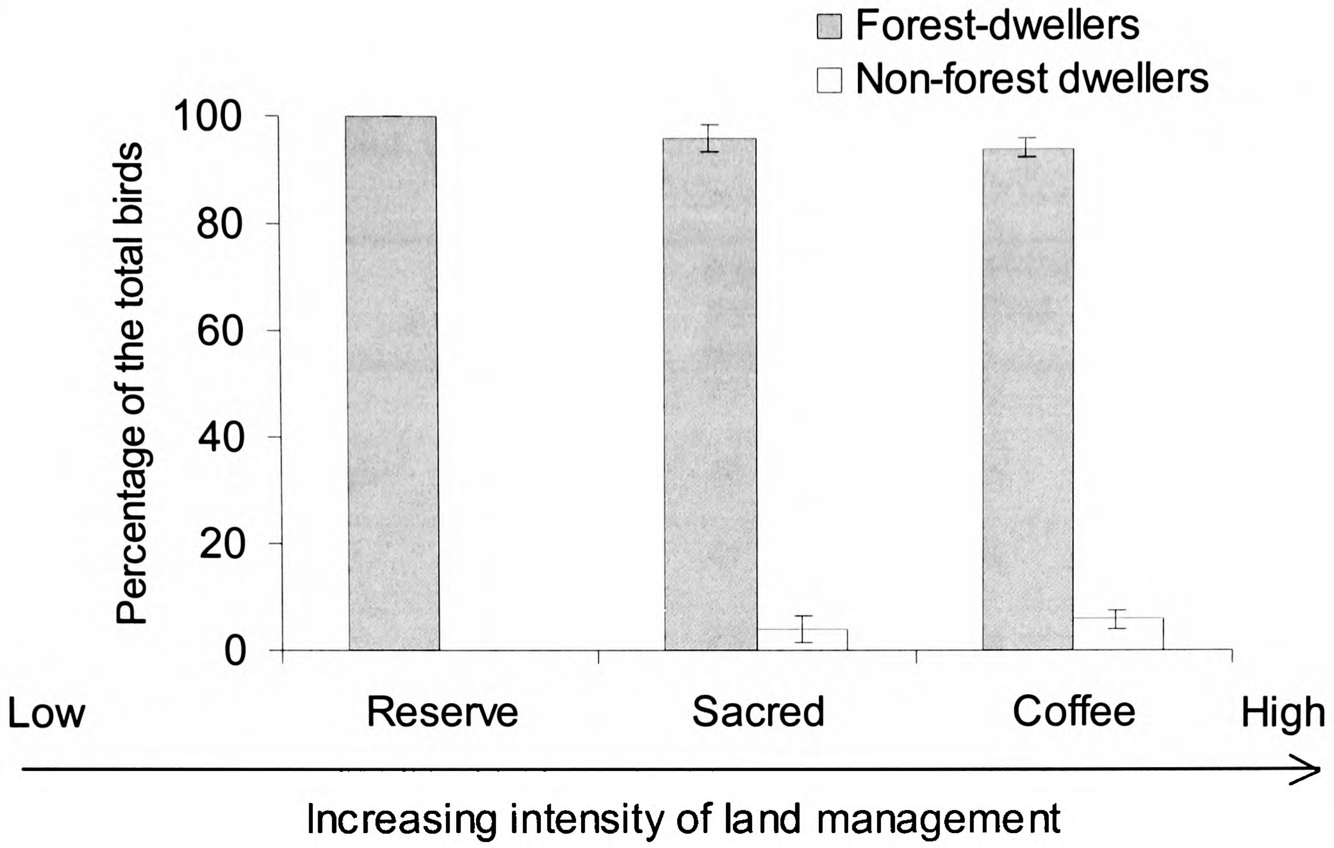


Figure 4.9 The mean proportion of forest-dwelling and non-forest dwelling birds (observations) in the three land use types showing a decline and an increase respectively, with growing human intervention in land management. The forest reserve (n = 10), sacred forest patches (n = 25) and coffee plantations (n = 23) are on the continuum of human intervention - the forest reserve has the lowest degree of land management and coffee plantations the highest. Note: error bars indicate 95% confidence intervals.

Out of the total of 86 species, 10 representative species, which were sampled adequately (including those, which occurred at more than 25 sampling localities out of 48; and were observed on at least 100 occasions in all samples together), were selected for a detailed analysis of their abundances in sacred forests and coffee plantations. The effects of the proportion (%) of tree-cover in the surroundings on the percentage relative abundances of bird species were examined. The relative abundances of most forest-dwelling birds were unaffected (*Table 4.5*). However, the effect on the abundance of the Yellow-Browed Bulbul (*Iole indica*) was positive (R^2 = 0.102, *P* = 0.045), while the abundance of the Red-whiskered Bulbul (*Pycnonotus jocosus*) was affected negatively (R^2 = 0.217, *P* = 0.011) (*Table 4.5*). The abundance of the Large-billed Crow (*Corvus macrorhyncus*), a non-forest dweller was unaffected ($R^2 < 0.0001$, *P* = 0.977) while the effect on the abundance of the Yellow-footed Green Pigeon (*Treron phoenicoptera*), another non-forest dweller was non-significant negative ($R^2 < 0.220$, $P = 0.091$) (*Table 4.5*).

Table 4.5 Effect of the proportion (%) of tree-cover in the surrounding 250 m on % relative abundances of forest-dwelling and non-forest dwelling bird species from sacred forests (n = 25) and coffee plantations (n = 23). The species shown in the table occurred at more than 25 sampling localities out of 48 and were observed on at least 100 occasions in all samples together.

Name of bird species	**Habitat preference**	**Number of sampling localities**	**Nature of effect**	**R^2**	***P***
Crimson-backed Sunbird (*Nectarinia minima*)	Forest	41	No	.046	.171
Greater Racket-tailed Drongo (*Dicrurus paradiseus*)	Forest	40	No	.001	.821
Large-billed Crow (*Corvus macrorhyncus*)	Non-forest	41	No	.000	.977
Malabar Grey Hornbill (*Ocyceros griseus*)	Forest	25	No	.004	.753
Malabar Parakeet (*Psittacula columboides*)	Forest	38	No	.042	.208
Pale-billed flowerpecker (*Dicaeum erythrorhyncus*)	Forest	42	No	.004	.676
Red-whiskered Bulbul (*Pycnonotus jocosus*)	Non-forest	28	- ve	.217	.011
White-cheeked Barbet (*Megalaima viridis*)	Forest	41	No	.047	.169
Yellow-browed Bulbul (*Iole indica*)	Forest	39	+ ve	.102	.045
Yellow-footed Green Pigeon (*Treron phoenicoptera*)	Non-forest	13	- ve	.220	.091

Feedings guilds

It is commonly observed in fragmented tropical landscapes that fruit-eating birds are generally affected more by fragmentation than those feeding on flower nectar (Feinsinger 1976; Thiollay 1994; Renjifo 1999). This is attributed to the restriction in food available to frugivores that depend on specific type of fruits while the nectarivores are able to feed on a wide range of floral resources. In order to examine whether landscape modification in Kodagu has affected the proportions of feeding guilds of birds, specific diets of bird species were assessed by prior knowledge or personal field observation and confirmed by information from Ali (1996) and Grimmett *et al.* (1998). Bird species were classified into: diurnal raptors, which include eagles, buzzards and hawks; frugivores, which include both pulp and fruit-seed eaters; granivores, which are grass-seed feeding birds such as sparrows; insectivores, which include aerial insectivores (*e.g.* swifts), bark insectivores (*e.g.* wood-peckers), canopy insectivores (*e.g.* bee-eaters and fly catchers), terrestrial insectivores (*e.g.* thrushes), understorey insectivores (*e.g.* prinias); and nectarivores, which include sunbirds.

Table 4.6 Distribution of birds with different habitat preferences and feeding guilds in the Kodagu area showing numbers of birds observed and percentages (in brackets) in the respective land use type.

Land use type	**(A) DISTRIBUTION OF SPECIES Ecological groups of birds**										
	Habitat preference		**Feeding guild**								
	F	**NF**	**DR**	**FR**	**GR**	**IA**	**IB**	**IC**	**IT**	**IU**	**NT**
Forest reserve	39 (100)	0 (0)	3 (8)	14 (35)	0 (0)	0 (0)	3 (8)	12 (31)	2 (5)	4 (10)	1 (3)
Sacred forests	65 (86)	11 (14)	5 (7)	24 (31)	3 (4)	0 (0)	7 (9)	24 (32)	2 (3)	7 (9)	4 (5)
Coffee plantations	60 (82)	13 (18)	3 (4)	24 (32)	1 (1)	1 (1)	7 (10)	26 (36)	2 (3)	7 (10)	2 (3)

Land use type	**(B) DISTRIBUTION OF INDIVIDUALS (OBSERVATIONS) Ecological groups of birds**										
	Habitat preference		**Feeding guild**								
	F	**NF**	**DR**	**FR**	**GR**	**IA**	**IB**	**IC**	**IT**	**IU**	**NT**
Forest reserve	554 (100)	0 (0)	6 (1.1)	321 (58)	0 (0)	0 (0)	24 (4.3)	134 (24.2)	2 (0.4)	31 (5.5)	36 (6.5)
Sacred forests	1676 (96)	77 (4)	14 (0.8)	876 (50)	5 (0.3)	0 (0)	68 (3.9)	492 (28)	5 (0.3)	113 (6.4)	180 (10.3)
Coffee plantations	1430 (92)	123 (8)	17 (1.1)	917 (59)	2 (0.1)	39 (2.5)	86 (5.6)	367 (23.7)	6 (0.4)	36 (2.3)	83 (5.3)

Key to the acronyms: Habitat preference: F – forest-dwellers, NF – non-forest dwellers; Feeding guild: DR – diurnal raptors, FR – frugivorous, GR – granivorous, IA – aerial insectivorous, IB – bark insectivorous, IC – canopy insectivorous, IT – terrestrial insectivorous, IU – understorey insectivorous, NT – nectarivorous birds.

Generally, frugivorous birds were most prominent and formed 50 – 59% of the bird observations in Kodagu (*Table 4.6*). The understorey insectivorous birds were significantly less abundant in coffee plantations (Kruskal-Wallis test, $\chi^2 = 11.262$, df = 2, $P = 0.004$) while nectarivorous birds were significantly more abundant in sacred forests (Kruskal-Wallis test, $\chi^2 = 12.671$, df = 2, $P = 0.002$). The distribution of birds belonging to the remaining feeding guilds was not significantly different across the three land use types (*Table 4.6*).

Body size and fecundity

Body size has been a major consideration in assessing a bird's response to fragmentation – the theoretical predictions (*e.g.* Gaston & Lawton 1988; Blackburn *et al.* 1990; Gaston & Blackburn 1995; Greenwood *et al.* 1996) and empirical observation imply that large-bodied birds are more extinction-prone in fragmented landscapes than small-bodied birds (Turner 1996; Laurance *et al.* 1997). This has been attributed amongst other things to hunting (Peres & Terborgh 1995). Similarly, birds with large body sizes have low fecundities and may be more prone to extinction in forest patches possibly because of high nest predation (Sieving & Karr 1997).

Body mass in grams is commonly used to assess body size in birds (*e.g.* Terborgh *et al.* 1990; Warburton 1997). Information on the body weights of the Western Ghats' birds was not easily available, since this usually has to be obtained by mist-netting and collecting birds of each species or by examining records of museum specimens. Body-lengths[17], obtained from the Field Guide for the Birds of the Indian Subcontinent by Grimmett *et al.* (1998) were used as a source for body sizes. Birds were divided into five body size classes. Similarly, based on the information on the average number of eggs in a clutch, obtained from Ali (1996), birds were categorised into three fecundity classes[18] (*Table 4.7*).

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[17] Body length, including bill and tail was measured in centimetres; figures taken from the most reliable published source. The length of tail feathers in species such as the Greater Racket-tailed Drongo (*Dicrurus paradiseus*) or the Paradise Flycatcher (*Terpsiphone paradisi*) was excluded from the measurement. The use of body mass is more appropriate for the present analysis; body length may only give a crude idea of body size. Nevertheless, in the absence of any other reliable source, the use of body length documented by Grimmett *et al.* (1998) was the most pragmatic solution to the problem of finding precise information.

[18] Some birds such as the House sparrow (*Passer domesticus*) breed throughout the year and produce many clutches. On the other hand, birds such as the Black Eagle (*Ictinaetus malayensis*) produce only one egg, once in a year (Ali 1996). Thus, clutch size in highly fecund birds is likely to be a poor estimate of reproductive rate. However, the main purpose of the analysis is to identify the groups of birds that may be vulnerable to landscape modification (*i.e.* birds with low fecundities), the information for which is based on the field observation by Ali (1996) and therefore reliably accurate.

Table 4.7 Distribution of birds belonging to different body size and fecundity classes in the Kodagu area showing numbers of birds observed and percentages (in brackets) in the respective land use type.

Land use type	(A) DISTRIBUTION OF SPECIES Ecological groups of birds							
	Body size (Length in cm)					Fecundity (Clutch size)		
	< 10	11-30	31-50	51-70	> 70	1	2-4	>5
Forest reserve	2 (5)	23 (59)	10 (26)	4 (10)	0 (0)	3 (8)	29 (74)	7 (18)
Sacred forests	6 (8)	49 (65)	16 (22)	4 (5)	0 (0)	3 (4)	53 (70)	20 (26)
Coffee plantations	4 (5)	52 (72)	12 (16)	4 (6)	1 (1)	4 (6)	50 (68)	19 (26)

Land use type	(B) DISTRIBUTION OF INDIVIDUALS (OBSERVATIONS) Ecological groups of birds							
	Body size (Length in cm)					Fecundity (Clutch size)		
	< 10	11-30	31-50	51-70	> 70	1	2-4	>5
Forest reserve	85 (15)	248 (45)	195 (35)	26 (5)	0 (0)	16 (3)	506 (91)	32 (6)
Sacred forests	320 (18)	907 (52)	377 (22)	149 (8)	0 (0)	12 (1)	1420 (81)	321 (18)
Coffee plantations	201 (12.9)	774 (50)	403 (26)	173 (11)	2 (0.1)	14 (1)	1243 (80)	296 (19)

Small-bodied birds (< 10 cm in length) were significantly more abundant in sacred forests (Kruskal-Wallis test, $\chi^2 = 6.941$, df = 2, $P = 0.031$) compared to coffee plantations and the reserve. Large bodied birds (> 50 cm in length) were significantly more abundant in coffee plantations (Kruskal-Wallis test, $\chi^2 = 13.449$, df = 2, $P = 0.001$), than in sacred forests. They were more abundant in sacred forests than in the forest reserve. This observation is contradictory to what would be expected in a modified landscape. The birds with small clutch sizes (number of egg: 1) were significantly more abundant in the forest reserve (Kruskal-Wallis test, $\chi^2 = 11.471$, df = 2, $P = 0.003$) while those with large clutch sizes (number of eggs: > 5) were significantly less abundant (Kruskal-Wallis test, $\chi^2 = 17.433$, df = 2, $P < 0.0001$).

In Kodagu, abundances of most bird species were unaffected by isolation from the forest reserve (*Table 4.8*). However, the relative abundances of two frugivorous birds, the White-cheeked Barbet (*Megalaima viridis*) ($R^2 = 0.113$, $P = 0.029$) and the Yellow-footed Green Pigeon (*Treron phoenicoptera*) ($R^2 = 0.311$, $P = 0.038$) increased in more isolated localities. The abundance of the Crimson-backed Sunbird (*Nectarinia minima*), a nectarivore, also increased in isolated localities ($R^2 = 0.158$, $P = 0.009$). The Crimson-backed Sunbird (*Nectarinia minima*) is about 8 cm in length, the White-cheeked Barbet (*Megalaima viridis*) is about 23 cm, while the Yellow-footed Green Pigeon (*Treron phoenicoptera*) is 33 cm long. It appears that body-size is not a good predictor of the effects of isolation from the reserve on the bird species in Kodagu. Feeding guild may be more important. For instance, the White-cheeked Barbet (*Megalaima viridis*) and the Yellow-footed Green Pigeon (*Treron phoenicoptera*), both are known to relish figs (Ali 1996) and often found feeding on fig trees (*Ficus* spp.), which are significantly more common in sacred forests and coffee plantations than in the forest reserve (*Figure 4.10*). The Crimson-backed Sunbird is more abundant in isolated localities because of its nectarivorous habit (*e.g.* Feinsinger 1976; Greenberg 1981; Johns 1991; Restrepo *et al.* 1997) and the ability to use a wide range of floral resources in the patchy landscape away from the contiguous forest.

Table 4.8 Effect of isolation (distance from the forest reserve in km) on % relative abundances of bird species with different feeding guilds and body sizes from sacred forests (n = 25) and coffee plantations (n = 23). The species shown in the table occurred at more than 25 sampling localities out of 48 and were observed on at least 100 occasions in all samples together.

Name of bird species	**Feeding guild**	**Body length (cm)**	**Number of sampling localities**	**Nature of effect**	**R^2**	***P***
Crimson-backed Sunbird (*Nectarinia minima*)	Nectari-vorous	8	41	+ ve	.158	.009
Greater Racket-tailed Drongo (*Dicrurus paradiseus*)	Canopy insectivore	32	40	No	.061	.120
Large-billed Crow (*Corvus macrorhyncus*)	Frugivore, Omnivore	53	41	No	.000	.977
Malabar Grey Hornbill (*Ocyceros griseus*)	Frugivore	45	25	No	.023	.456
Malabar Parakeet (*Psittacula columboides*)	Frugivore	38	38	No	.061	.129
Pale-billed flowerpecker (*Dicaeum erythrorhyncus*)	Frugivore	8	42	No	.004	.680
Red-whiskered Bulbul (*Pycnonotus jocosus*)	Frugivore, Insectivore	20	28	No	.010	.602
White-cheeked Barbet (*Megalaima viridis*)	Frugivore	23	41	+ ve	.113	.029
Yellow-browed Bulbul (*Iole indica*)	Frugivore, Insectivore	20	39	No	.036	.242
Yellow-footed Green Pigeon (*Treron phoenicoptera*)	Frugivore	33	13	+ ve	.311	.038

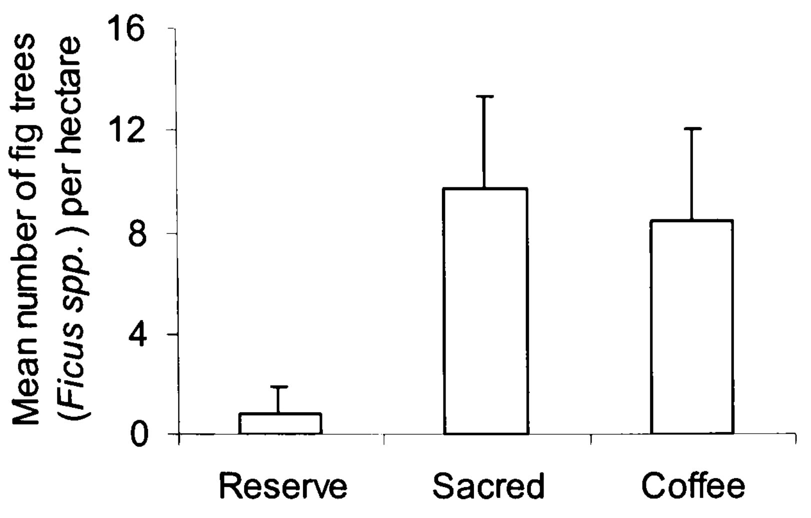


Figure 4.10 Numbers of fig trees (<u>Ficus</u> spp.) per hectare in the forest reserve (n = 10), sacred forests (n = 25) and coffee plantations (n = 23), error bars indicate 95% confidence intervals. The trees include 13 species of figs, namely, <u>Ficus amplissima</u>, <u>F. asperrima</u>, <u>F. beddomei</u>, <u>F. benghalensis</u>, <u>F. callosa</u>, <u>F. glomerata</u>, <u>F. hispida</u>, <u>F. microcarpa</u>, <u>F. mysorensis</u>, <u>F. racemosa</u>, <u>F. tsjahela</u>, <u>F. virens</u> and an unidentified species of <u>Ficus</u>.

4.1.3 Conservation of bird diversity

Birds have a great public appeal and the international community is interested in their conservation. Species that are known to have restricted distribution range or those that are facing the danger of extinction from their natural habitat are currently of international conservation concern (*e.g.* IUCN 2000).

The level of endemicity in the birds of the Western Ghats is lower than that in trees (WCMC 1992) – 16% of the birds in the study area are endemic in contrast with 63% of trees. Also, only 2% of the birds in the study area were listed as threatened by IUCN (2000).

There was no significant difference in the distribution of endemic birds (Kruskal-Wallis test, $\chi^2 = 2.631$, df = 2, $P = 0.268$). Very few birds in the landscape were threatened; their numbers were not sufficient in order to analyse whether the differences in their occurrence across land use types were significant.

Table 4.9 Distribution of endemic and threatened birds in the Kodagu area showing numbers of birds observed and percentages (in brackets) in the respective land use type.

Land use type	(A) DISTRIBUTION OF SPECIES Birds of conservation importance			
	Endemicity		Threat status	
	Endemic	Non-endemic	Threatened	Non-threatened
Forest reserve	12 (31)	27 (69)	1 (3)	38 (97)
Sacred forests	14 (18)	62 (82)	2 (3)	74 (97)
Coffee planta^{n}s	13 (18)	60 (82)	2 (3)	71 (97)

Land use type	(B) DISTRIBUTION OF INDIVIDUALS (OBSERVATIONS) Birds of conservation importance			
	Endemicity		Threat status	
	Endemic	Non-endemic	Threatened	Non-threatened
Forest reserve	259 (47)	295 (53)	2 (0.4)	552 (99.6)
Sacred forests	608 (35)	1145 (65)	6 (0.3)	1747 (99.7)
Coffee planta^{n}s	504 (32)	1049 (68)	4 (0.3)	1549 (99.7)

4.2 Discussion

4.2.1 Distribution of bird diversity in the landscape

There were no significant differences in the expected and the observed numbers of species unique to any land use type, as revealed by a χ^2 test. A large proportion of species were either ubiquitous (40%) or shared between the land use types (35%). This can be attributed to the mobility of birds – the intervening matrix of coffee plantations is not a strong barrier. Species such as the Malabar Trogon (*Harpactes fasciatus*) that are known to be very sensitive to the changes in habitat quality in the forest reserve (Ali 1996) can be seen in forest patches near to the reserve. Thus, unlike trees, where a few species were restricted to the contiguous forest, many birds are able to use the patches adjacent to the reserve.

Owiunji & Plumptre (1998) observed in Budongo forest, Uganda, that logging can lead to a dramatic decline in bird diversity due to the negative effects on several habitat-sensitive species. The negative effects of fragmentation on birds are also well documented by other researchers (*e.g.* Turner 1996; Laurance *et al.* 1997). Thus, intensive management coupled with fragmentation can have adverse effects on bird diversity. However, despite long-term landscape modification and land management in Kodagu, species richness was not significantly different across land use types (*Figure 4.3*), possibly because the landscape in Kodagu is not as insularised as are many other tropical landscapes.

4.2.2 Effects of landscape modification on bird diversity

Patch area

In fragmented landscapes, patch area has often been found to explain a high percentage of variation in bird diversity (Opdam *et al.* 1985; Freemark & Merriam 1986; Warburton 1997; Miller & Cale 2000). Furthermore, there is evidence suggesting that area is a surrogate variable that represents a number of factors affecting species' persistence in a given patch of limited area – insufficient resources, isolation from the source habitat or elevated levels of nest predation – resulting in a decrease in species diversity (Askins 1995; Robinson *et al.* 1995). However, it is important to look at the types of landscapes where the above studies have taken place. For instance, the study by Miller & Cale (2000) took place in the central wheat belt in the western Australia, where 93% of the vegetation was cleared for crop production. It can be argued that patch size assumes greater importance in highly fragmented systems, while there is empirical support to the assertion that in the landscapes where a large proportion of the original habitat is still intact, patch area may not be as important (Andrén 1994).

Landscape effects

Isolation

Lynch & Whigham (1984) presented an excellent analysis of avian communities of 185 forest patches ranging in size from 5 to more than 1000 ha in the Maryland State, USA. They demonstrated the impact of isolation on the richness and abundances of bird species. Opdam *et al.* (1985) also showed the importance of isolation in explaining bird diversity using multivariate models. They speculated that because isolated patches are reached less often by species with specific habitat requirements or with certain ecological traits that limit their dispersal, such species will either be absent from remote patches or be present in low numbers. In contrast, the habitat-generalists and those with high dispersal capabilities, are not affected or are favoured by fragmentation-induced isolation, as they benefit from reduced competition in more isolated patches.

In the previous chapter, tree diversity was found to be strongly negatively associated with isolation (*Figure 3.6 A*). As a consequence, the available variety of resources for birds will decline with distance of a patch from the forest reserve. In response to this, birds that are able to use a wide range of resources will increase in abundance while those that are selective are likely to decline. As for the individual species, although it was not possible to examine the effects on habitat-specialist species because of limited occurrence of such birds in sampling, isolation was found to favour an increase in abundance of a habitat-generalist, nectarivorous bird, the Crimson-Backed Sunbird (*Nectarinia minima*) (*Table 4.8*). Similarly, the abundances of two frugivores, the White-cheeked Barbet (*Megalaima viridis*) and the Yellow-footed Green Pigeon (*Treron phoenicoptera*) also showed positive effects of isolation, possibly because of their habit of feeding on a variety of fruit trees such as figs, that are more common in sacred forests and coffee plantations than in the reserve (*Figure 4.10*).

Intervening landscape matrix

Burkey (1989) observed that species, which move between fragments are less likely to become locally extinct because their populations could be supplemented through immigration and recolonisation. Laurance (1991a; 1997b) found the inability of a species in using the intervening landscape matrix to be the key predictor of local vulnerability in mammals and suggested that species detected in the matrix are most effective in dispersing between large forest and patches. Warburton (1997) observed similar traits in birds, in terms of vulnerability of certain species to fragmentation. Thus, the ability of a species to use the intervening landscape matrix is important in determining how a species will respond to fragmentation.

Owiunji & Plumptre (1998) found in logged forest of Budongo, Uganda, that the abundances of certain bird species were positively correlated with the amount of mature forest type in the surroundings. In the present study, the amount of tree-covered land in the surroundings was found to be an important variable that explained bird diversity

(*Figure 4.6*, *Table 4.2*). While most bird species studied preferred tree-covered habitats, their abundances were not associated with the tree-cover in the surrounding 250 m. However, patterns were found for some species. The abundance of the Yellow-Browed Bulbul (*Iole indica*), a forest-dwelling bird, declined with the decrease in tree-cover in the surrounding landscape; and that of the Red-Whiskered Bulbul (*Pycnonotus jocosus*), a non-forest dwelling bird, increased (*Table 4.5*). According to Grimmett *et al.* (1998), the Yellow-Browed Bulbul is predominantly a forest dweller while the Red-Whiskered Bulbul prefers thinly wooded areas around human habitations. Both birds feed on fruits and casually on insects; both are documented to be about 20 cm in length. Thus, there is no difference in their feeding guild and body sizes. It is most likely that the responses of the two species to the surrounding tree cover are because of their habitat preferences. The Yellow-footed Green Pigeon (*Treron phoenicoptera*) is another bird that prefers open habitats; and was less abundant in tree-covered surroundings, although at a very low significance, $P = 0.091$ (*Table 4.5*). It is likely that the remaining species shown in *Table 4.5* do not have as strict habitat preference as the three examples above. As a result, they do not show significant increase or decrease in abundances with the tree cover in surroundings. In addition to the habitat preference, the scale at which the tree-cover in the landscape affects species abundances may also be important. For instance, the Malabar Parakeet (*Psittacula columboides*) or the Large-billed Crow (*Corvus macrorhyncus*) are highly mobile birds. Their abundances may not be affected by the tree cover in the immediate surroundings because they can move over large distances. For the Pale-Billed Flowerpecker (*Dicaeum erythrorhyncus*), which is small in size, but versatile in using food resources (*e.g.* Feinsinger 1976), factors other than tree cover (*e.g.* the variety of trees producing floral nectar) may be important.

Ecological groups

Species richness of birds is not significantly affected by land management in Kodagu (*Figure 4.3*). However, Miller & Cale (2000) cautioned that a focus on richness alone might mask the unique responses of different species to fragmentation. The responses of birds belonging to various ecological groups are discussed in this section.

Forest dwelling species face severe competition with non-forest dwelling species in fragmented landscapes and they may be preyed upon by invading predator species (*e.g.*

Ambuel & Temple 1983). In Kodagu, 80% of all bird species are forest-dwellers, possibly due to the high tree-cover in the landscape that reduces the adverse effects of fragmentation, including edge effects, and favour species that prefer tree-covered habitat. The intensity of land management caused a small but significant decline in the proportion of forest-dwelling birds in the landscape, along the gradient of increasing disturbance (*Figure 4.9*). A one-way ANOVA also revealed that the proportions of forest dwelling birds significantly decrease while those of non-forest dwellers significantly increase with the intensity of land management. This can be attributed to the change in forest structure related to land management (*e.g.* Thiollay 1994; Owiunji & Plumptre 1998).

Thiollay (1994) found that forest-dwelling species that decreased in numbers or disappeared from fragmented agroforests in Sumatra were large frugivores (pigeons, hornbills), large insectivores (trogons, broadbills), or interior forest specialists, either terrestrial (pheasants, pittas) or associated with low understories (many babblers, flycatchers) or dead trees (woodpeckers). On the other hand, birds that survived or increased in agroforests were mostly smaller, omnivorous species usually found in gaps, upper canopies, open woodlands or high grasses. They were predominantly small frugivores, foliage insectivores and nectarivores (Thiollay 1994). Renjifo (1999) also found in the subandean avifauna that forest raptors, terrestrial insectivores and large frugivores were very extinction-prone, while nectarivores, small frugivores and aerial insectivores were highly resilient. Several other studies have shown that nectarivores, have an ability to use a wide range of resources in fragmented landscapes while frugivores lack such ability (Feinsinger 1976; Greenberg 1981; Johns 1991; Stouffer & Bierregaard 1995; Restrepo *et al.* 1997; Sieving & Karr 1997; Warburton 1997). In the present study, while body size does not appear to be a major factor in the ability of birds to use the modified landscape, the findings about the feeding guilds are in accordance with the above studies.

Theoretical work has often suggested a strong positive association between body size in birds and extinction-proneness (*e.g.* Gaston & Lawton 1988; Blackburn *et al.* 1990; Gaston & Blackburn 1995; Greenwood *et al.* 1996). While some empirical studies support this prediction (*e.g.* Thiollay 1994; Gillepsie 2000), others fail to find such a pattern (*e.g.* Warburton 1997; Renjifo 1999). In the present study, body size was not important in determining vulnerability to local extinction either because all large-bodied birds have already become extinct or because bird diversity is so unharmed in the tree-covered Kodagu landscape that no birds suffer any serious threat – either of the conclusions may be extreme. It is true that some birds such as the Great Hornbill (*Buceros bicornis*) did inhabit the forest reserve in the nineteenth century (Richter 1870) but are not present now. Hunting was primarily responsible for local extinction (*e.g.* Redford 1992, cited in Brandon 1997), although the species is reported to occur in various other parts of the Western Ghats mountain range (Grimmett *et al.* 1998). In the present study, the intervening matrix of coffee plantations in the fragmented landscape had significantly more large-bodied birds than other land use types, possibly because birds such as the Malabar Grey Hornbill (*Ocyceros griseus*) come to coffee plantation to feed, but go back to their preferred habitat for roosting and breeding – while coffee plantations offer a variety of fruits all year round, they may not be a secure habitat for nesting because of human disturbance. Such behaviour was reported by Kalina (1988, cited in Owiunji & Plumptre 1998), for Black and White Casqued Hornbill (*Ceratogymna cylindricus*) in Africa, that preferred to breed in unlogged forests, although logged forests attracted many hornbills because of mass fruiting trees.

Miller & Cale (2000) found that shrub understorey is an important habitat feature for the survival of a number of woodland birds in a fragmented Australian landscape. In the present study, sacred forests had significantly more small-bodied birds than expected, as revealed by the Kruskal-Wallis test, first, because of the shrub understorey that offers a secure habitat and second, because all nectarivorous birds (sunbirds and flowerpeckers) are small-bodied and are more abundant in sacred forests (*Table 4.6*, *Table 4.7*). Thus, unlike other studies (Thiollay 1994; Turner 1996; Laurance *et al.* 1997; Gillepsie 2000), while large-bodied birds are not at a great risk in Kodagu due to the high tree-cover, the structurally rich forest patches provide ideal habitats for small-bodied birds – understorey insectivores and nectarivores.

Forest patches face elevated nest predation rates (Sieving & Karr 1997), possibly because they often support high densities of nest predators such as rodents (Laurance 1994). Less fecund birds were significantly more abundant in the forest reserve (where nest predation by rodents may not be as severe as in the patchy landscape) as revealed by the Kruskal-Wallis test. Thus, the present study suggests that the contiguous forest offers a secure habitat to species such as the Mountain Imperial Pigeon (*Ducula badia*) and the Nilgiri Wood Pigeon (*Columba elphinstonii*), both of which produce one or two eggs in a clutch and are slow breeders, while species such as the Asian Koel (*Eudynamic scolopacea*) and the White-throated Kingfisher (*Halcyon smyrnensis*) that produce on average 13 and 7 eggs respectively, are abundant in the patchy landscape – sacred forests and coffee plantations.

4.3 Conclusions

4.3.1 Implications for conservation practice

Renjifo (1999) found that small patches of forest in the Andes, especially if they contain native tree-cover, were able to support a diverse avifauna despite long term isolation. In Kodagu, the patchy but tree-covered landscape has a potential for protecting bird diversity, predominantly because of the high proportion of native tree-cover that is retained and provides a matrix, which does not inhibit movement of birds. For instance, Johns (1991) observed that a 35 ha forest fragment in Amazonia, which lost many bird species was exposed to microclimatic changes induced by edge effects, however, its proximity to large forest blocks enabled continued visitation by large mobile canopy species – similar to the 'rescue effect' described by Brown & Kondric-Brown (1977). A study by Thiollay (1994) on the role of traditional agroforests in maintaining bird diversity in Sumatra, found that the one agroforest that was richer in native forest trees and less intensively managed in comparison with the others, had a higher bird species richness. On the other hand those poorer in species tended to adopt modern agroforestry, which aims at developing large, uniform blocks of intensively managed, monospecific, even aged plantations often using exotic trees for timber, fruit, spice, oil or rubber production. From this background, the recent trend of replacing native trees with *Grevillea robusta* in Kodagu (Reddy 1990) is alarming. In order to protect the present bird diversity, it is crucial to retain as many native shade trees in coffee plantations as possible. This can be brought about only with the involvement of the local landowners. For instance, Miller & Cale (2000) recognise that most vegetation in their study area is on private land and the future of avian diversity will depend on the cooperation of private landowners.

Thus, conservation of bird diversity in Kodagu will require the following three measures:

- Maintenance of native (rather than exotic) tree-cover in coffee plantations for providing a less hostile intervening matrix for the movement of large birds;
- Conservation of sacred forest patches in order to provide structurally rich habitats for small birds;
- Protection of the forest reserve for the species of birds vulnerable to extinction, *e.g.* the ones with low fecundity, such as the Mountain Imperial Pigeon (*Ducula badia*) and the Nilgiri Wood Pigeon (*Columba elphinstonii*).

4.3.2 Ideas for future research

Although the results presented in this chapter are entirely based on a 'snap-shot' picture of bird diversity, they provide an empirical example, which can direct the future research aimed at conserving bird diversity. However, this study has some limitations, and it is recommended that future research should address those:

- A study of this kind does not provide information concerning dynamic aspects of species abundances in various disturbed habitats. Long term monitoring of population is considerably more valuable than sampling populations at a point in time, especially where human disturbance exists as a part of the landscape processes (Johns 1991). Therefore, discerning relationships between landscape features and population dynamics has become an increasingly important aspect of conservation science and management (Thogmartin & Schaeffer 2000). While the present study has provided a glimpse of the distribution of populations of some bird species in the landscape and has related them to the landscape structure, it has admittedly not been able to link population dynamics with the landscape structure because this will require long time-series data. A careful investigation of species responses to the landscape modification in the long term is necessary.

- Rarity is an important determinant of vulnerability of a bird species to habitat fragmentation. Rare species include those with small geographical distribution, narrow habitat specificity or small population sizes (Renjifo 1999). Due to the limitations of the present data set, it was not possible to analyse the patterns of distribution of rare species in the patchy landscape. The future studies on bird diversity in Kodagu should focus on such species.

5 . MACROFUNGAL DIVERSITY

The effects of landscape modification on macrofungal diversity in the Kodagu landscape are examined in this chapter. The consequences of microclimatic factors and human intervention for sporocarp production are discussed.

5.1 Results

5.1.1 Macrofungi and sporocarp production

Fungi of various taxonomic groups that produce conspicuous sporocarps (fruiting bodies) visible to the unaided human eye are collectively called macrofungi (Arnolds 1992). It is an artificial and somewhat arbitrarily defined polyphyletic group that includes most Basidiomycetes (excluding rusts, smuts and yeasts), some Ascomycetes (*e.g. Peziza*) and Myxomycetes (*e.g. Fuligo*) (Watling 1995). However, due to their conspicuous nature, it is the most suitable group for rapid assessment of fungal diversity (*e.g.* Balmford *et al.* 2000).

Sporocarp production in macrofungi depends on several environmental as well as human-induced factors. Atmospheric temperature and moisture are considered to be the most important environmental factors, although authors differ in their assessment of the effects of these factors. For instance, Hering (1966, cited in Vogt *et al.* 1992), found no clear relationship between these environmental factors and sporocarp production whereas Fogel (1976, cited in Vogt *et al.* 1992), found a significant but weak correlation between the two. Fungal communities in the tropics show seasonal changes in sporocarp production (Watling 1995). Kodagu district has a 4-5 months dry season between December and April, when atmospheric moisture decreases rapidly; relative humidity can be as low as 60% and mean temperatures rise as high as 30°C (Pascal & Meher-Homji 1986). Consequently, soil moisture decreases and temperature rises rapidly. Sporocarp production in Kodagu is largely restricted to the rainy season, between May and October when the relative humidity is high and the temperature is low. Many fruiting bodies persist until January of the following year. Human disturbance is another important factor in sporocarp production (Watling 1995). Macrofungal fruiting is generally more extensive along the forest margins or on forest trails and banks than in the interior (Watling 1995). Watling (1995) attributes this phenomenon to disturbance. It is possible therefore that the various degrees of human intervention in coffee plantations, sacred forests and the forest reserve in Kodagu have strong influences on sporocarp production.

The effect of land management on sporocarp production in forest landscapes has been relatively well studied in temperate regions – *e.g.* Wasterlund (1989, cited in Vogt *et al.* 1992) in Scandinavian boreal forests; Waters *et al.* (1997) in northeastern California; Senn-Irlet & Bieri (1999) in the sub-alpine region of Switzerland. The findings of these studies suggest that diversity of sporocarps is higher in mature stands of natural forest than in younger ones, while sporocarp production is more profuse in managed forests than in unmanaged ones. The effects of landscape modification on sporocarp production in tropical regions are unknown and there are no systematic studies on sporocarp diversity of macrofungi in relation to disturbance. This study attempts to examine such effects in Kodagu.

In the present study, it was possible to identify most sporocarps to the family level, many to the genus level and a few to the species level. Their different identities were established by comparison of the morphological characters such as colour, size and shape of the fruiting bodies. The utility of this approach is argued for in biodiversity studies of highly diverse groups such as macrofungi (Balmford *et al.* 2000).

5.1.2 Distribution of macrofungal morpho-species in the landscape

Sporocarps of one type, are referred to as a 'morpho-species'. A total of 163 morpho-species were recorded (*APPENDIX 6*). Although more extensive field surveys are necessary to affirm the presence or absence of a morpho-species, the distribution of morpho-species in the three land use types, as observed in the present study, is shown in *Figure 5.1*. Morpho-species that were found only in a single land use type are referred to as 'unique', those shared by any two of the three types are 'shared' and those found in all three types are referred to as 'ubiquitous'. About 39% of all morpho-species are ubiquitous, 21% are shared, and 40% are unique – 75% of which were unique only to sacred forests. The expected distribution of unique morpho-species was calculated by adjusting the morpho-species number to the sample size in the respective land use type. Morpho-species distribution between the three land use types was significantly different from expected (χ^2 test, $\chi^2 = 26.262$, df = 2, $P < 0.0001$) due to the high numbers found in sacred forests (*Figure 5.1*).

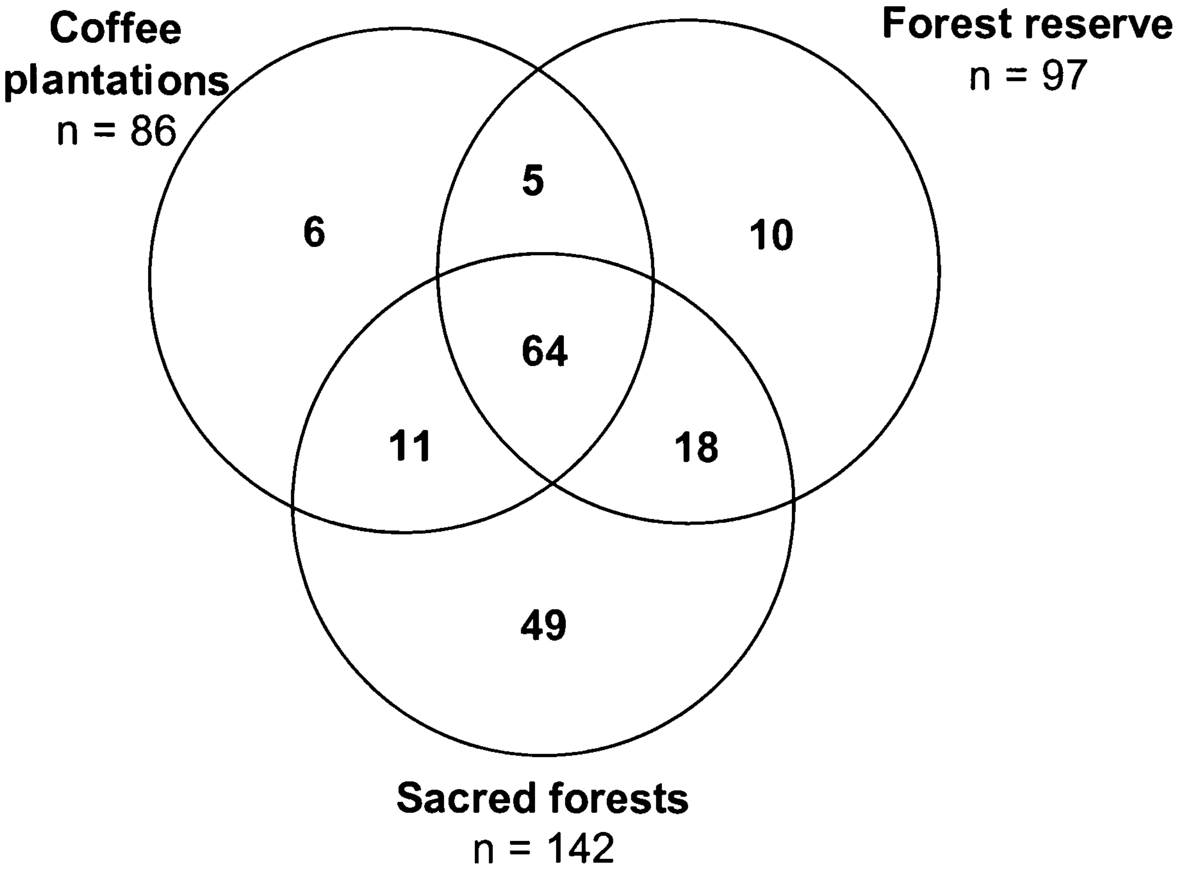


Figure 5.1 Numbers of unique and shared macrofungal morpho-species (Total number of morpho-species sample, N = 163). This analysis is based on morpho-species samples obtained from 58 sampling localities – 10 from the forest reserve, 25 sacred forests and 23 coffee plantations.

5.1.3 Effects of landscape modification on macrofungal communities

Ecological groups of macrofungi

Although little is known about the ecology of macrofungi, based on the morphology of their sporocarps they can be divided into two artificial groups – fleshy fungi having soft sporocarps (*e.g.* morpho-species from genus *Agaricus*), generally associated with leaf litter; and fungi having tough, flaccid sporocarps (*e.g.* morpho-species from genus *Phellinus*) that are normally associated with dead and decaying wood. While sporocarps associated with litter may belong to mycorrhizal macrofungi – *i.e.* those forming symbiotic associations with the host trees – a large proportion of known mycorrhizal macrofungi do not produce sporocarps (Dahlberg 2001). The following analyses pertain to the occurrences of macrofungal sporocarps irrespective of whether they belong to mycorrhizal morpho-species or not; the distribution of sporocarps of putative mycorrhizal morpho-species is presented later in the chapter (*p. 158*).

Habitat preference

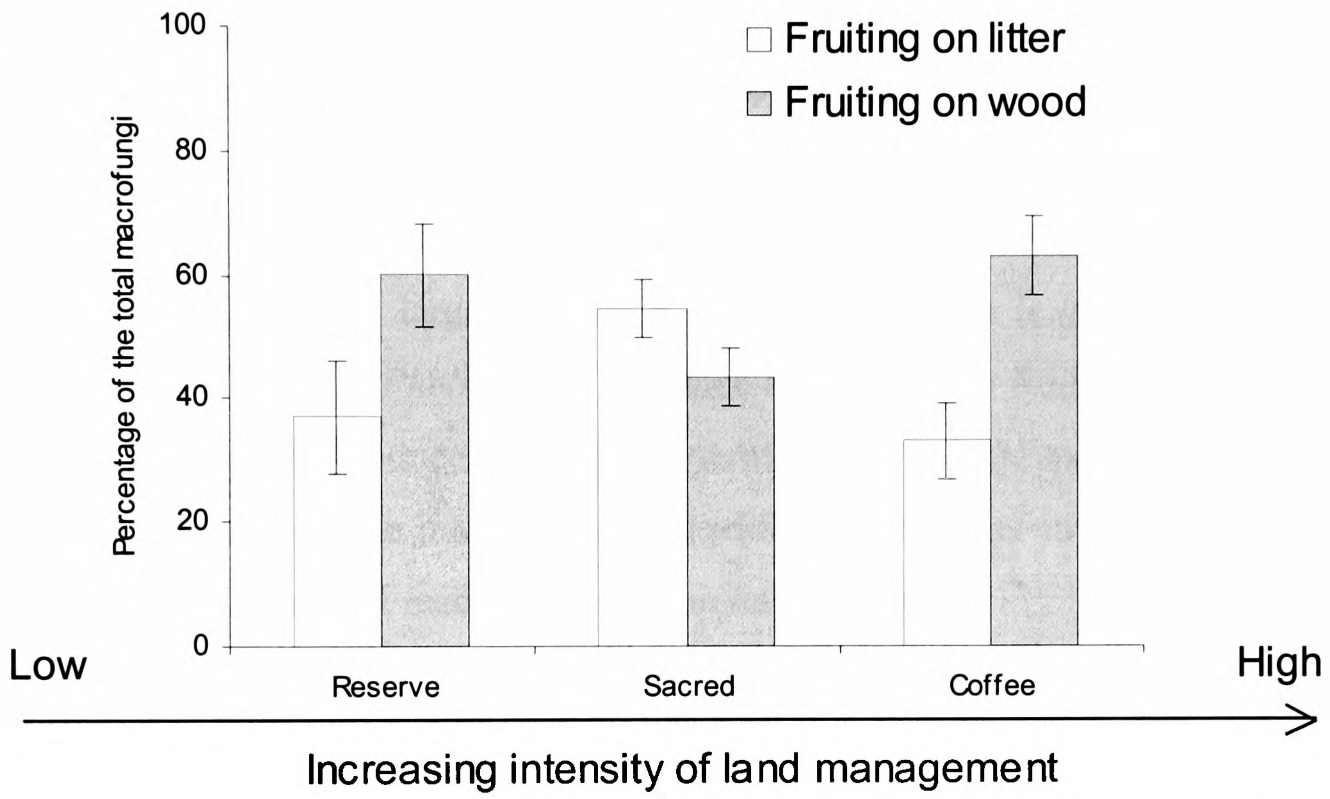


Figure 5.2 The Proportions of sporocarps of macrofungi fruiting on litter and on wood in the forest reserve (n = 10), sacred forests (n = 25) and coffee plantations (n = 23). Note: error bars indicate 95% confidence intervals.

Macrofungi were classified based on their habitat preferences, into two groups namely those fruiting on litter and those fruiting on wood, based on information from the available literature (*e.g.* Jordan 1995) supplemented by field observations. Although the vast majority of macrofungal sporocarps belong to one of these two groups, there are small proportions that belong to either coprophilous macrofungi (growing on cattle dung *e.g. Coprinus* sp.) or entomopathogenic macrofungi (parasitic on insects *e.g. Cordyceps* sp.). The proportional distributions of sporocarps of macrofungi fruiting on litter and on wood are shown in *Figure 5.2* and *Table 5.8*; the proportions of sporocarps belonging to coprophilous and entomopathogenic macrofungi are negligible (1.8, 2.1 and 3.8 percent in the forest reserve, sacred forests and coffee plantations respectively) and are not shown. Sporocarps of macrofungi fruiting on litter are significantly more abundant in sacred forests and those of macrofungi fruiting on wood are significantly less abundant (One-way ANOVA, F = 13.765, df = 2, $P < 0.0001$) in comparison with their sporocarp abundances in the forest reserve and coffee plantations.

Diversity of macrofungal morpho-species

Rarefied richness of morpho-species

In general, saprotropic fungi fruiting on litter tend to produce small sporocarps in high numbers while those which produce few large sporocarps tend to be mycorrhizal (Senn-Irlet & Bieri 1999). Fungi fruiting on wood vary in their sporocarp production – some morpho-species produce a large number of small sporocarps (*e.g.* some *Microporus* spp.) while others such as *Phellinus* sp. produce a single large fruiting body. Thus, the numbers of sporocarps produced can vary enormously between species, depending on the habitat preference of the macrofungus, in addition to several environmental factors. It is not a useful measure of macro-fungal sporocarp diversity.

Frequency of occurrence of sporocarps belonging to a particular morpho-species was used in order to estimate diversity. Morpho-species richness was adjusted according to frequency of occurrence of sporocarps (irrespective of the number of sporocarps produced) in order to calculate rarefied richness at each sampling locality. Hurlbert's (1971) rarefaction method was used. For each sample, richness was calculated by rarefying the frequency to 12 observations. The numbers of sampling localities that permitted calculation of rarefied morpho-species richness for a given frequency of observations of macrofungi are shown in *Table 5.1.* The samples were rarefied to 12 observations because this number allowed the inclusion of all sampling localities in the analysis. Rarefaction curves for all sampling localities are displayed in *Figure 5.3.* In more than 90% of the localities, the increase in rarefied morpho-species richness was consistent with increasing frequency of observations (*Figure 5.3*). Therefore, although the morpho-species are rarefied to a smaller frequency of observations (12) due to the limitations of the data, this estimate can be reliably used as an index of diversity in the present study.

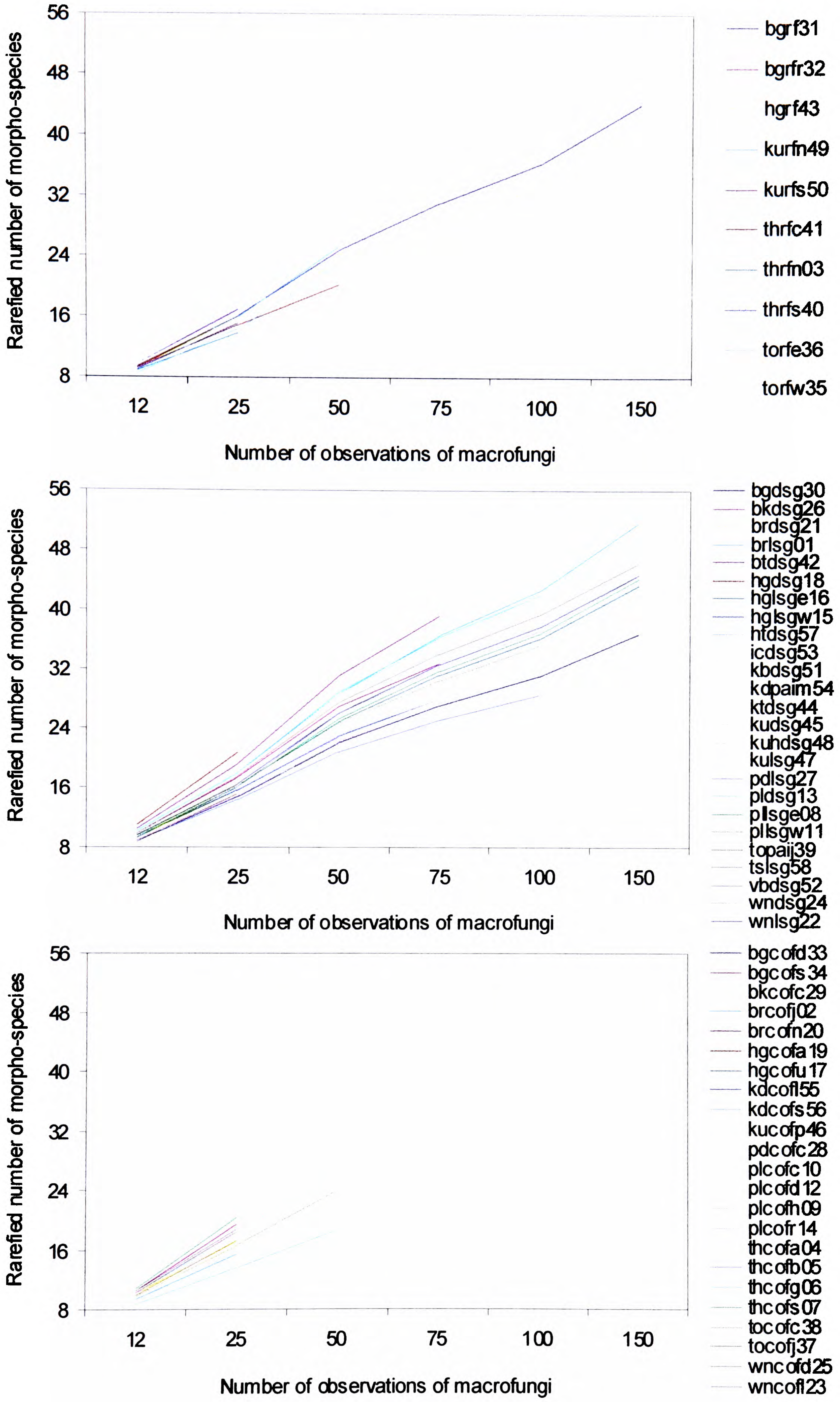


Figure 5.3 Rarefaction curves (macrofungi) for sampling localities; the top panel shows forest reserve localities (n = 10), the middle panel shows sacred forests (n = 25) and the bottom panel shows coffee plantations (n = 23).

Table 5.1 Numbers of sampling localities that permitted calculation of rarefied morpho-species richness for macrofungi

Frequency of observations of macrofungal sporocarps to which samples were rarefied	*12*	*25*	*50*	*75*	*100*	*150*
Numbers of sampling localities that permitted calculation of rarefied morpho-species richness						
Reserve (n = 10)	10	10	5	2	2	1
Sacred (n = 25)	25	24	21	18	14	8
Coffee (n = 23)	23	16	3	0	0	0
Total (n = 58)	***58***	***50***	***29***	***20***	***16***	***9***

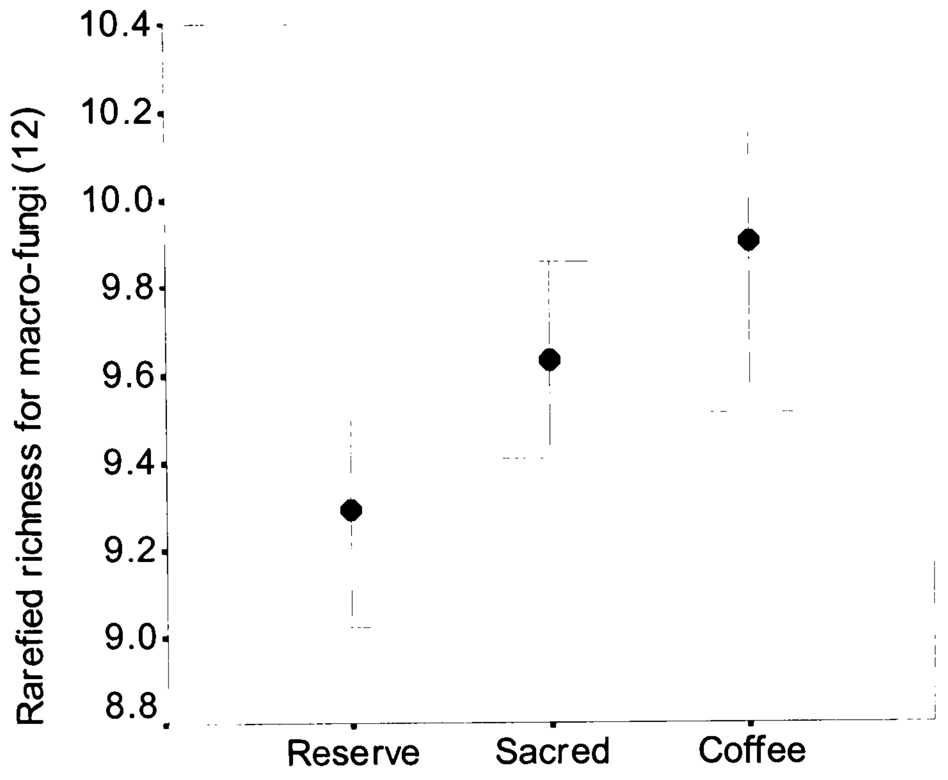


Figure 5.4 Mean rarefied richness for macrofungal morpho-species (for 12 observations) in the three land use types. For each sample, richness was calculated by rarefying the frequency of observations of sporocarps to 12 observations. Number of samples in the forest reserve, sacred forests and coffee plantations were 10, 25 and 23 respectively. Note: error bars indicate 95% confidence intervals.

Although the rarefied richness increased with human intervention, the differences in the mean richness across land use types were not significant (One-way ANOVA, F = 2.805, df = 2, $P = 0.069$) (*Figure 5.4*).

Effect of patch area

It is possible that patch size is important for explaining diversity of macrofungi because large patches are believed to have a greater variety of habitats (*e.g.* Gotelli 1998) that could potentially support a greater number of morpho-species. In order to determine whether diversity is associated with patch size, rarefied richness of macrofungal sporocarps in sacred forests were plotted against the sizes of patches. No significant association was found ($R^2 = 0.035$, $P = 0.383$) (*Figure 5.5*).

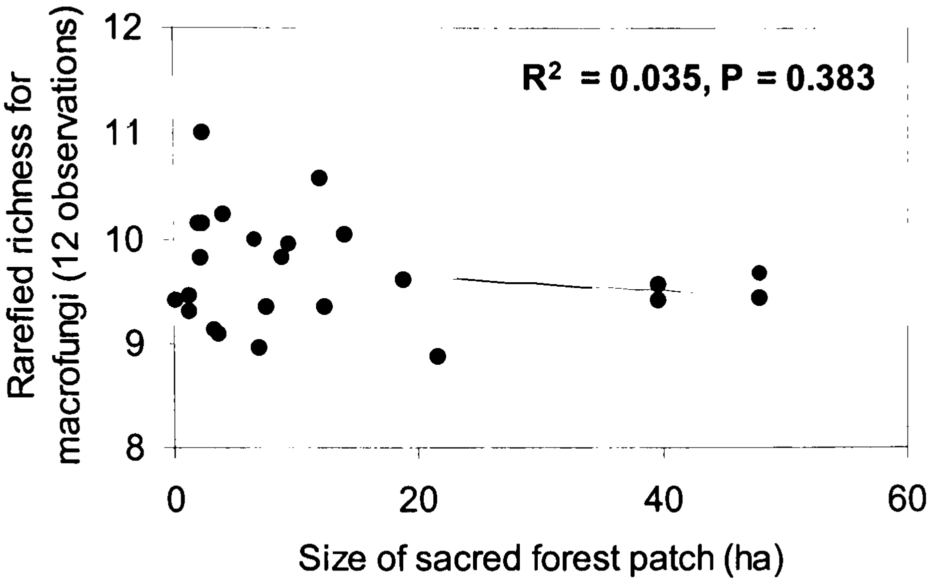


Figure 5.5 The relationship between the size of a sacred forest patch (n = 24) and diversity of macrofungal sporocarps.

Landscape effects

Effect of isolation

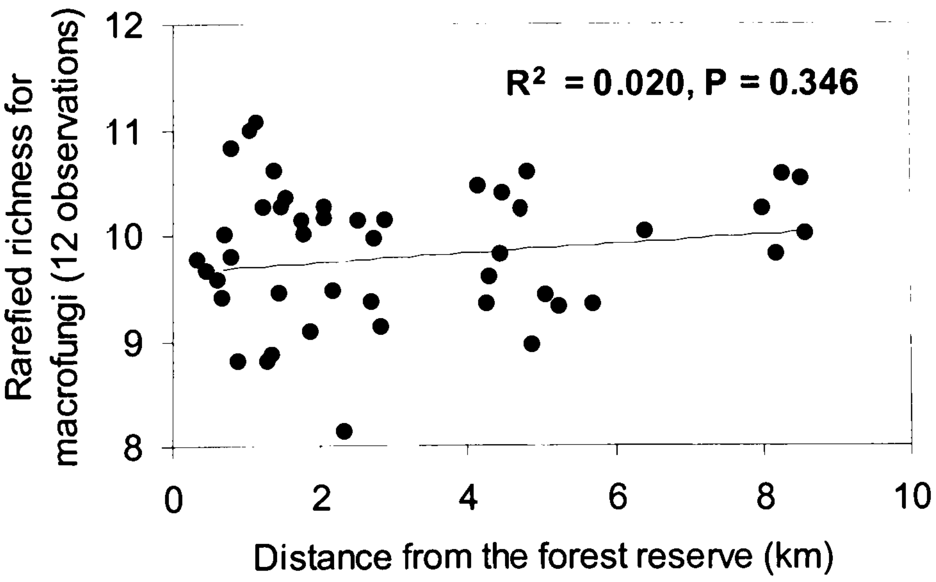


Figure 5.6 The relationship between the isolation of sampling localities (sacred forests, n = 25 and coffee plantations, n = 23) and diversity of macrofungal sporocarps.

Effect of the intervening landscape matrix

CHAID identified landscape complexity in the 1 km surroundings as the most significant ($P < 0.001$) variable for determining macrofungal diversity (*Figure 5.7*). The residual variation in diversity was explained by various other factors related to forest structure, such as variability in canopy closure ($P < 0.001$), density of lianas ($P < 0.005$) and variability in basal area ($P < 0.05$) (*Figure 5.7*).

The results of forward stepwise regression were comparable, although the length of edges in the 1 km surroundings explained only 16% of the variation in macrofungal diversity. The rest, 8% of the variation (amounting to a total of 24%) was explained by another measure of landscape heterogeneity, total number of patches in the surrounding 1 km, although at a low significance ($P = 0.098$) (*Table 5.2*).

The subsidiary factors identified by the two analyses are very different. It may be possible that there is a linear association between the diversity of macrofungal sporocarps and landscape complexity (as suggested by stepwise regression), while the association with the habitat variables, such as the variability in canopy closure, identified by CHAID is non-linear.

Figure 5.7 (Please see Page 149) CHAID diagram showing variables that are significant in explaining macrofungal diversity. Full titles of the variables and the ranges of values they take are shown in boxes.

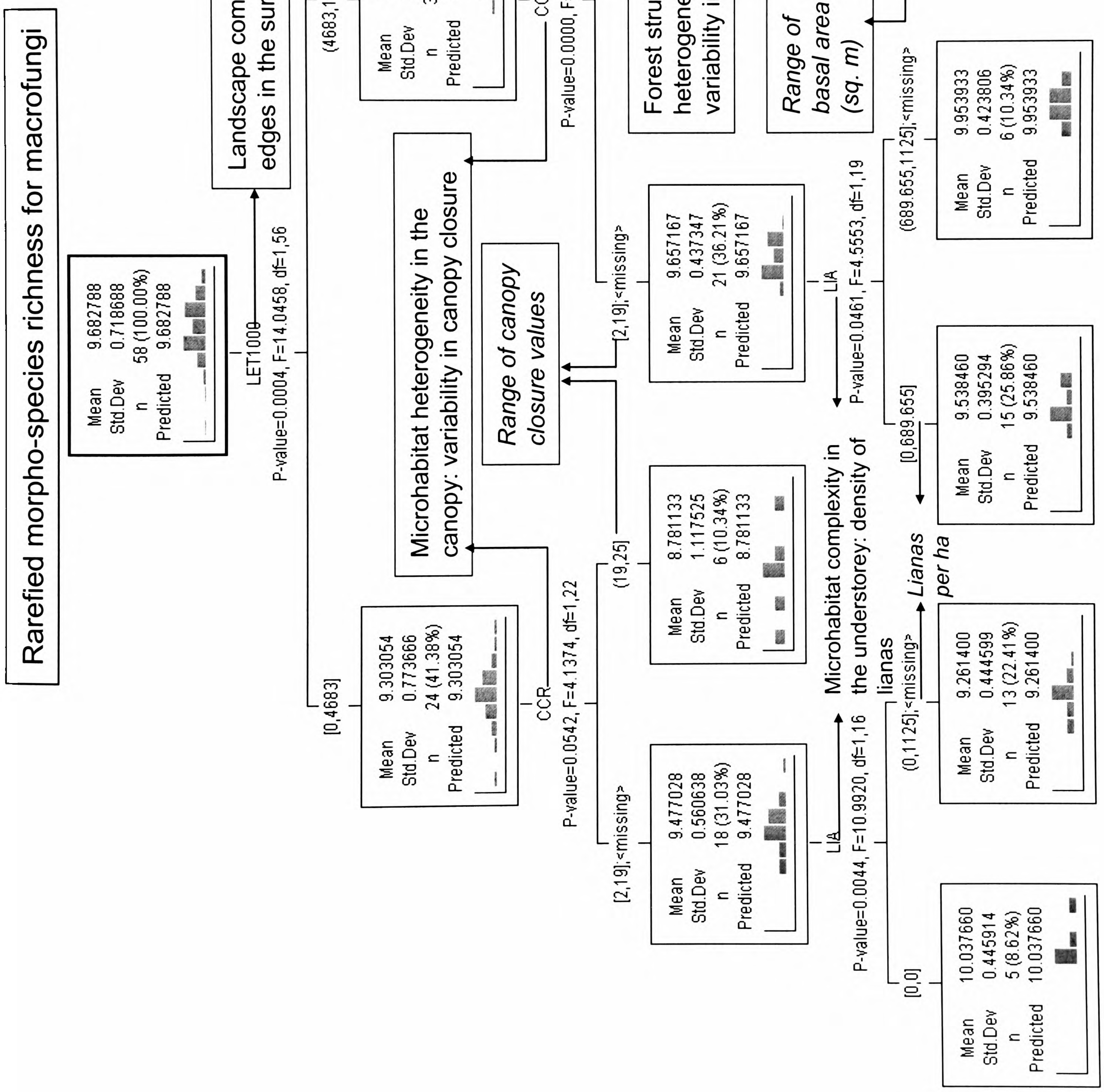

Rarefied morpho-species richness for macrofungi
Mean 9.682788
Std.Dev 0.718688
n 58 (100.00%)
Predicted 9.682788
LET1000
P-value=0.0004, F=14.0458, df=1,56
Landscape complexity: total length of edges in the surrounding 1 km
Length in m
[0,4683]
(4683,11313.3]
Mean 9.303054
Std.Dev 0.773666
n 24 (41.38%)
Predicted 9.303054
Mean 9.950835
Std.Dev 0.544134
n 34 (58.62%)
Predicted 9.950835
CCR
P-value=0.0542, F=4.1374, df=1,22
CCR
P-value=0.0000, F=30.1094, df=1,32
Microhabitat heterogeneity in the canopy: variability in canopy closure
Range of canopy closure values
[2,19];<missing>
(19,25]
[2,19];<missing>
(19,25]
Mean 9.477028
Std.Dev 0.560638
n 18 (31.03%)
Predicted 9.477028
Mean 8.781133
Std.Dev 1.117525
n 6 (10.34%)
Predicted 8.781133
Mean 9.657167
Std.Dev 0.437347
n 21 (36.21%)
Predicted 9.657167
Mean 10.425223
Std.Dev 0.317360
n 13 (22.41%)
Predicted 10.425223
LIA
P-value=0.0044, F=10.9920, df=1,16
LIA
P-value=0.0461, F=4.5553, df=1,19
BAR
P-value=0.0106, F=9.4575, df=1,11
Microhabitat complexity in the understorey: density of lianas
Lianas per ha
Forest structural heterogeneity: variability in basal area
Range of basal area (sq. m)
[0,0]
(0,1125];<missing>
[0,689.655]
(689.655,1125];<missing>
[0.166898,0.361996]
(0.361996,1.03663]
Mean 10.037660
Std.Dev 0.445914
n 5 (8.62%)
Predicted 10.037660
Mean 9.261400
Std.Dev 0.444599
n 13 (22.41%)
Predicted 9.261400
Mean 9.538460
Std.Dev 0.395294
n 15 (25.86%)
Predicted 9.538460
Mean 9.953933
Std.Dev 0.423806
n 6 (10.34%)
Predicted 9.953933
Mean 10.233286
Std.Dev 0.128736
n 7 (12.07%)
Predicted 10.233286
Mean 10.649150
Std.Dev 0.331792
n 6 (10.34%)
Predicted 10.649150

Table 5.2 Forward stepwise regression model for macrofungal diversity.

Step	R^2	Variable	T value	*P*
1	0.1573	Total length of edges in the surrounding 1 km	3.12	0.00009
2	0.2412	Total length of edges in the surrounding 1 km	3.97	0.00015
		Total number of patches in the surrounding 1 km	- 2.37	0.098

The results of these two analyses suggest that the microclimatic changes associated with landscape modification and land management may be responsible for the patterns of diversity of macrofungal sporocarps. Changes associated with landscape modification and land management include various factors such as increased solar radiation reaching the ground, air temperature, amount of precipitation and reduced atmospheric moisture (Geiger 1965). Moisture content of the air is perhaps the most important factor because seasonal changes in sporocarp production in the tropics are known to be influenced by it (Watling 1995).

Relative humidity is a measure of the amount of water in the air, compared with the amount of water the air can hold, at the temperature it happens to be at when the amount of water is measured. However, it does not give a good estimate of the actual moisture content – important for sporocarp production. Dew point is a measure of how much water vapour is actually in the air. It is defined as the temperature at which saturation occurs if air is cooled at constant pressure without addition or removal of vapour (Barry & Chorley 1987, cited in Glassy & Running 1994). Thus, it is important to examine how various landscape factors, and particularly landscape complexity (identified by CHAID as the total length of edges in the surrounding 1 km) is related to the variation in dew point temperature across the landscape.

Effect of the intervening landscape matrix on moisture content of the air

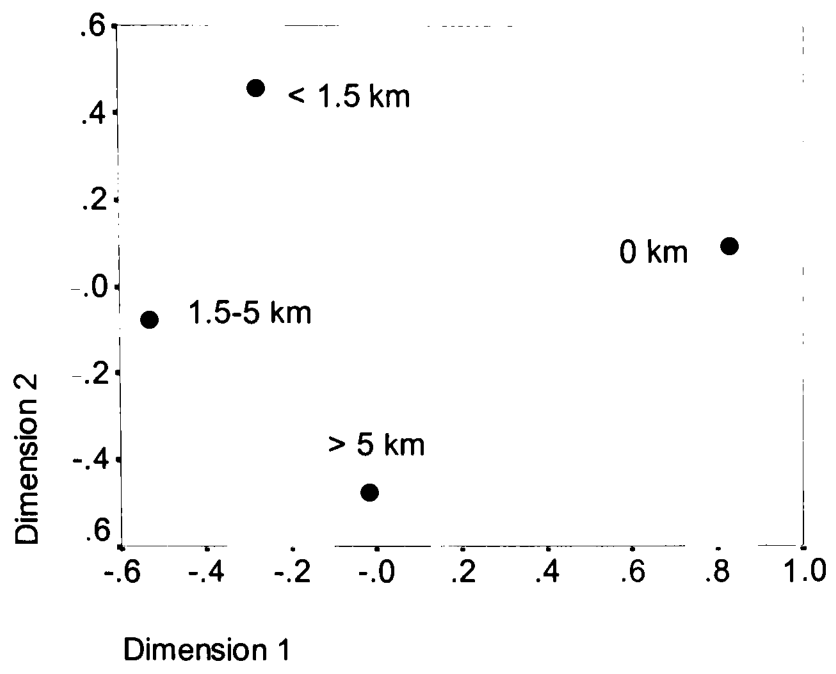
The dew-point temperature was measured at four localities in the study area during three monsoon months, July – September 1999, using microclimate monitoring data loggers. Landscape structure was measured at those four localities using GIS (see *pp. 55-64*). While the measures of landscape structure such as patch size and isolation from the large forest were poorly correlated with dew point temperature ($R^2 = 0.140$, $P = 0.626$ and $R^2 = 0.147$, $P = 0.616$ respectively), the total length of edges in the surrounding 1 km and dew point temperature were strongly negatively correlated ($R^2 = 0.991$, $P = 0.004$). Thus, the landscape complexity in the surroundings influences moisture content of the air, which in turn is one of the most important factors for sporocarp production. Despite having collected microclimatic data, it was not possible to include them in the multivariate models (*Figure 5.7* and *Table 5.2*) due to the lack of sufficient information. The number of localities where trees, birds and macrofungi were sampled was greater than the number of microclimate monitoring devices available. As a result, it was not possible to obtain data for all sampling localities at one time.

Similarity in macrofungal diversity

In this study, similarity in tree diversity decreased with distance, while it had no effect on similarity in bird diversity. Natel & Neumann (1992) have used similarity matrices for comparing diversity of trees and macrofungi along a local vegetation gradient in southern Québec, Canada. In order to examine the pattern that macrofungal diversity followed, Sorenson's quantitative similarity index (Magurran 1988), which gives an estimate of percentage similarity between rank-abundance distributions, was compared across groups of localities. Only 24 sacred forest patches and 8 forest reserve localities are considered in this analysis, while 1 sacred forest and 2 forest reserve localities (that do not have sufficient data for stems ≥ 1 cm dbh) are excluded so that the analysis is comparable to that for trees. The 32 sampling localities in the reserve and sacred forest patches were divided into four distance bands. There was no distinct pattern of decline in similarity with distance (*Table 5.3*).

Table 5.3 Mean Sorenson's quantitative index showing similarity in macrofungal diversity between the reserve (0 km) and the sacred forest patches (at < 1.5, 1.5-5, > 5 km away from the reserve).

	Sorenson's quantitative index			
Distance (km)	**0**	**< 1.5**	**1.5 - 5**	**> 1.5**
% Similarity	100	82	81	90
n	8	9	7	8



Euclidean Distances

Dimensionality: 2

	0 km	**< 1.5 km**	**1.5-5 km**	**> 5 km**
0 km	.000			
< 1.5 km	1.350	.000		
1.5-5 km	1.340	0.316	.000	
> 5 km	0.571	1.050	0.922	.000

Figure 5.8 Non-metric non-metric multidimensional scaling of similarity in diversity of macrofungal sporocarps between the reserve (0 km) and sacred forest patches (at < 1.5, 1.5-5, > 5 km from the reserve).

Non-metric multidimensional scaling (Kenkel & Orloci 1986) assigns observations of community similarities (represented by points) to specific locations in a conceptual low-dimensional space such that the distances between points in the space match the given similarities as closely as possible. The results of non-metric multidimensional scaling are presented in *Figure 5.8*. The results did not reveal any pattern of decline in similarity along the distance gradient.

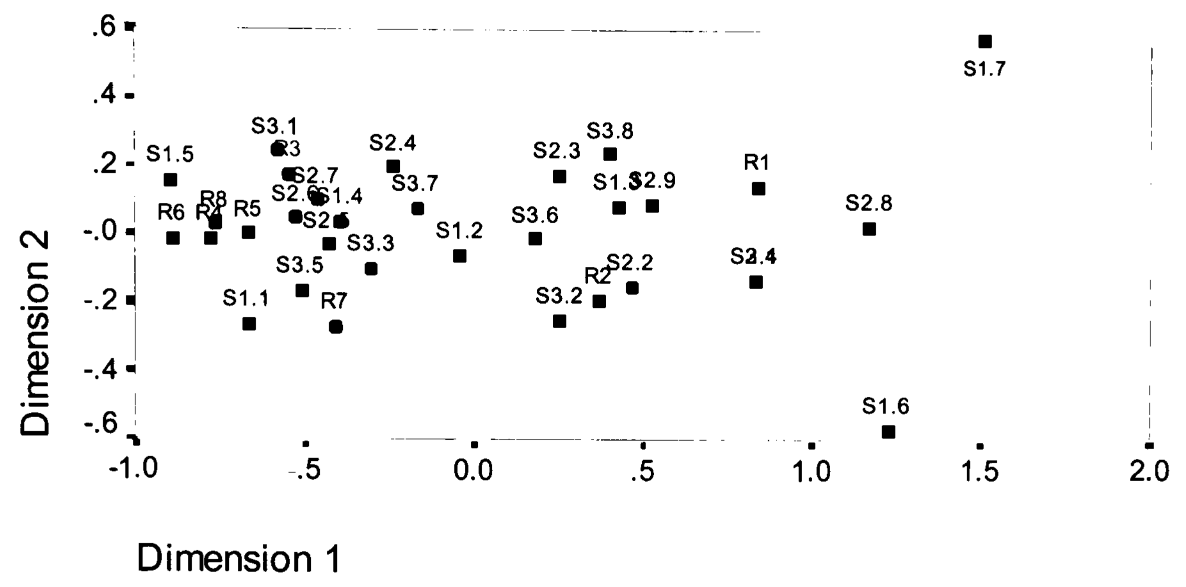


Figure 5.9 Non-metric multidimensional scaling of similarity in diversity of macrofungal sporocarps between the localities in the reserve and sacred forest patches; each sampling locality is represented by a point. Those titled R belong to the forest reserve and those titled S belong to sacred forests. In each title, the number following S represents the distance band that the particular locality belongs to (1 corresponds with < 1 km, 2 corresponds with 1.5-5 km, 3 corresponds with > 5 km away from the forest reserve). Numbers following R and those at the end of titles beginning with S indicate ordering of localities within a particular distance band (e.g. S 2.2 indicates the locality in sacred forest that is 1.5-5 km away from the reserve and is the second nearest locality to the reserve in that distance band).

In order to examine how similar individual sacred forest patches are to each other and to the localities in the reserve, rank-abundance distributions of macrofungal sporocarps were subjected to non-metric multidimensional scaling (*Figure 5.9*). Coffee plantations were excluded from this analysis in order to avoid over-crowding of points in the two-dimensional representation shown in *Figure 5.9*. The forest reserve localities did not cluster together distinctly, and there was no ordering in the localities sampled (*Figure 5.9*) (S-STRESS = 0.00482).

Pair-wise comparisons of dissimilarity between forest reserve localities and sacred forests in three distance bands did not reveal any distinct pattern (*Table 5.4*).

Table 5.4 Pair-wise comparisons of dissimilarity in macrofungal diversity between forest reserve localities and sacred forest patches in the Kodagu landscape; the values show Euclidean distances; larger the distance value, higher is the dissimilarity. Locations of sampling localities within the study area are mapped in Figure 3.8.

Locality	R1	R2	R3	R4	R5	R6	R7	R8
S1.1	35.847	25.593	10.440	7.348	7.874	8.544	8.367	6.782
S1.2	22.561	11.958	13.675	18.385	15.748	20.664	13.711	18.655
S1.3	11.314	8.000	23.875	28.827	26.211	31.337	21.284	28.478
S1.4	29.665	19.287	7.071	9.539	7.416	12.000	8.185	9.434
S1.5	41.485	30.838	10.247	5.292	7.874	5.568	14.967	5.831
S1.6	17.349	24.352	44.933	49.173	46.690	51.352	39.774	48.683
S1.7	20.149	30.692	51.711	56.807	54.378	59.042	47.445	56.152
S2.1	9.849	13.675	33.630	38.314	35.972	40.780	30.757	38.026
S2.2	11.402	6.481	24.454	29.950	26.926	32.187	21.471	29.580
S2.3	15.427	9.592	21.354	25.040	23.000	27.423	16.583	24.310
S2.4	26.439	17.117	10.630	13.491	12.083	16.031	9.798	12.884
S2.5	30.578	20.075	6.557	8.832	7.348	11.533	7.746	8.718
S2.6	32.280	21.494	4.000	7.280	5.000	9.487	7.000	7.550
S2.7	31.718	21.071	6.633	8.660	6.245	11.045	10.050	9.000
S2.8	10.536	20.025	40.485	45.563	43.081	47.906	36.414	44.967
S2.9	10.392	8.485	26.153	31.129	28.513	33.437	24.062	30.773
S3.1	34.627	24.310	7.550	7.483	7.211	9.644	11.832	7.746
S3.2	14.560	6.481	21.071	25.593	22.782	27.857	17.916	25.436
S3.3	27.604	17.088	8.718	11.790	9.434	14.142	7.681	11.619
S3.4	9.849	13.675	33.630	38.314	35.972	40.780	30.757	38.026
S3.5	32.031	21.772	6.928	8.062	6.856	10.296	5.000	7.416
S3.6	17.263	8.832	18.762	23.388	21.048	25.846	16.941	23.087
S3.7	26.363	16.279	11.790	15.748	13.638	18.138	13.856	16.000
S3.8	13.638	8.246	23.238	28.792	26.058	31.145	21.977	28.548

5.1.4 Effects of landscape modification on macrofungal morpho-species

Landscape management in Kodagu has affected populations of habitat-specialist trees negatively, while there has been a positive effect on the populations of disturbance-adapted trees and birds, as seen in the previous chapters. However, there are problems in assessing this for macrofungi for the following reasons:

- What constitutes the population of a species of a macrofungus and how it is distributed in space cannot be defined, based on the estimates of sporocarp production. The occurrence of sporocarps only indicates the presence of mycelia of a particular macrofungus and that the habitat is conducive for its fruiting (Watkinson 2002 *pers. comm.*).
- The effects of landscape management have not been studied for fungi in the Western Ghats, therefore which fungi occupy disturbed niches and how disturbance affects sporocarp production are unknown. Also, ecological characteristics such as

their mycorrhizal associations with plants have been very poorly studied (Hawksworth 1997).

The results presented in this section on individual morpho-species are therefore speculative, based on the data on occurrences and abundances of sporocarps.

Out of a total of 163 morpho-species, 18 representative morpho-species, which were sampled adequately (including those observed at least on 100 occasions in all samples together), were selected for a detailed analysis of sporocarp abundances. The effects of patch area, isolation and the landscape complexity on sporocarp production were examined.

Effect of patch area

Sporocarp production in most morpho-species was unrelated to the area of the sacred forest patch in which they occurred (*Table 5.5*). However, two morpho-species, *Ganoderma* sp. with an orange coloured fruiting body and *Microporus* sp. with a spoon-like fruiting body showed a significant positive association with patch area. A brown coloured *Collybia* sp. showed an almost significant (*P* = 0.09) positive association (*Table 5.5*).

Table 5.5 Effect of patch area (only sacred forest patches, n = 24) on sporocarp production (per hectare) of macrofungal morpho-species. The morpho-species shown in the table were observed at least on 100 occasions in all samples together.

Name of morpho-species	**Fruiting on**	**Number of patches**	**Nature of effect**	**R^2**	***P***
Ascomycetes (yellow-coloured, ball-like sporocarps)	Wood	18	No	.028	.495
Auricularia sp.	Wood	17	No	.031	.487
Collybia sp. (brown in colour)	Litter	19	+ ve	.152	.089
Daedalopsis flavida	Wood	16	No	.141	.137
Ganoderma sp.(big fruiting body)	Wood	19	No	.038	.407
Ganoderma sp. (orange coloured)	Wood	22	+ ve	.177	.045
Marasmius sp.	Litter	22	No	.065	.240
Microporus sp.	Wood	22	No	.027	.451
Microporus sp. (spoon-like)	Wood	10	+ ve	.395	.038*
Microporus sp. (trumpet shaped)	Wood	18	No	.0001	.984
Mycena sp.	Litter	20	No	.007	.724

Polyporus sp.	Wood	21	No	.017	.558
Sarcocypha sp.	Wood	16	No	.037	.458
Tricholomataceae	Litter	16	No	.022	.567
Xylaria sp. (club-shaped)	Wood	19	No	.074	.247
Xylaria sp. (rod-like)	Wood	22	No	.074	.211
Xylaria sp. (short)	Wood	16	No	.148	.127
Xylaria sp. (thin)	Wood	17	No	.005	.781

Landscape effects

Effect of isolation

Sporocarp production in most morpho-species was unrelated to the isolation (distance from the forest reserve) of sacred forests or coffee plantations in which they occurred (*Table 5.6*). However, a species in Genus *Polyporus* and a morpho-species of family Tricholomataceae showed a significant increase in sporocarp production with distance from the reserve, while a morpho-species of the genus *Xylaria* with a thin fruiting body showed almost significant ($P = 0.07$) positive association (*Table 5.6*).

Table 5.6 Effect of isolation (distance from the forest reserve in km) of sacred forests (n = 25) and coffee plantations (n = 23) on sporocarp production (per hectare) of macrofungal morpho-species. The morpho-species shown in the table were observed at least on 100 occasions in all samples together.

Name of morpho-species	**Fruiting on**	**Number of localities**	**Nature of effect**	**R^2**	***P***
Ascomycetes (yellow-coloured, ball-like sporocarps)	Wood	33	No	.013	.514
Auricularia sp.	Wood	32	No	.032	.316
Collybia sp. (brown in colour)	Litter	32	No	.041	.256
Daedalopsis flavida	Wood	37	No	.001	.938
Ganoderma sp.(big fruiting body)	Wood	31	No	.001	.904
Ganoderma sp. (orange coloured)	Wood	42	No	.008	.577
Marasmius sp.	Litter	38	No	.062	.127
Microporus sp.	Wood	39	No	.047	.181
Microporus sp. (spoon-like)	Wood	17	No	.036	.453
Microporus sp. (trumpet shaped)	Wood	33	No	.007	.630
Mycena sp.	Litter	26	No	.035	.353
Polyporus sp.	Wood	35	+ ve	.122	.037
Sarcocypha sp.	Wood	26	No	.016	.536
Tricholomataceae	Litter	28	+ ve	.137	.048
Xylaria sp. (club-shaped)	Wood	33	No	.019	.431
Xylaria sp. (rod-like)	Wood	36	No	.001	.909
Xylaria sp. (short)	Wood	21	No	.057	.283
Xylaria sp. (thin)	Wood	22	+ ve	.150	.067 *ns*

Effect of the intervening landscape matrix

Although the total length of edges in a 1 km radius around a sampling locality was important in determining the richness of sporocarps, as suggested by CHAID analysis, sporocarp abundance in all but one morpho-species was unrelated to this factor (*Table 5.7*). Sporocarp production in *Auricularia* sp. increased with increasing complexity of the surrounding landscape (*Table 5.7*).

Table 5.7 Effect of the intervening landscape matrix (total length of edges in 1 km radius) around the sampling localities in the forest reserve (n = 10), sacred forests (n = 25) and coffee plantations (n = 23), on sporocarp production (per hectare) of macrofungal morpho-species. The morpho-species shown in the table were observed at least on 100 occasions in all samples together.

Name of morpho-species	Fruiting on	Number of localities	Nature of effect	R^2	*P*
Ascomycetes (yellow-coloured, ball-like sporocarps)	Wood	36	No	.027	.327
Auricularia sp.	Wood	34	+ ve	.184	.010
Collybia sp. (brown in colour)	Litter	36	No	.021	.397
Daedalopsis flavida	Wood	42	No	.001	.866
Ganoderma sp.(big fruiting body)	Wood	36	No	.010	.550
Ganoderma sp. (orange coloured)	Wood	47	No	.027	.267
Marasmius sp.	Litter	43	No	.030	.258
Microporus sp.	Wood	44	No	.001	.830
Microporus sp. (spoon-like)	Wood	20	No	.023	.516
Microporus sp. (trumpet shaped)	Wood	36	No	.014	.493
Mycena sp.	Litter	30	No	.001	.945
Polyporus sp.	Wood	39	No	.012	.493
Sarcocypha sp.	Wood	27	No	.011	.590
Tricholomataceae	Litter	31	No	.080	.116
Xylaria sp. (club-shaped)	Wood	38	No	.034	.259
Xylaria sp. (rod-like)	Wood	41	No	.001	.843
Xylaria sp. (short)	Wood	24	No	.041	.333
Xylaria sp. (thin)	Wood	27	No	.019	.485

5.1.5 Conservation of macrofungal diversity

The conservation of macrofungi may be of importance to two prominent interest groups:

- *The scientific community*: Macrofungi form an important component of ecosystems in decomposing dead wood and recycling nutrients. In addition, macrofungi that are mycorrhizal and form symbiotic associations with their tree hosts are known to be important in ecosystem functioning (Fogel 1980; van der Heijden *et al.* 1998; Dahlberg 2001).

- *Local communities*: Macro-fungi are one of the most important non-timber forest products in the local economy of tropical regions (Hartshorn 1995). In Kodagu, the people use some macrofungal sporocarps (mushrooms) in their food. They may be an important dietary substitute to animal protein during the rainy season. This corresponds with the breeding season of some wild animals in the region, when hunting is banned through social conventions (Gadgil *et al.* 1993). A few macrofungi are also used in medicine – indeed a traditional medicine-practitioner from Heggala village reported the use of a *Ganoderma* sp. for treating diseases ranging from the common cold to cancer!

Putative mycorrhizal macrofungi

From the literature on European macrofungi (Jordan 1995), genera such as *Russula* and *Boletus*, and families such as Tricholomataceae commonly have mycorrhizal associations. The number of observations of putative mycorrhizal macrofungi at each locality was compared with the number of observations of non-mycorrhizal macrofungi. It should be emphasized, however, that due to the lack of published information for the Western Ghats the mycorrhizal associations used in this analysis are putative rather than confirmed. Sporocarps of mycorrhizal macrofungi were significantly more frequent in sacred forests (Kruskal-Wallis test, $\chi^2 = 26.262$, df = 2, $P < 0.0001$) than in the forest reserve and coffee plantations.

Useful macrofungi

The utility of macrofungal morpho-species to the local community was determined through local knowledge, by interviewing knowledgeable individuals during the field investigations in 1999 and 2000. Macrofungi such as some *Agaricus* spp. and some from the family Tricholomataceae, were reported to be edible, and a few others such as *Ganoderma* spp. and *Phellinus* spp. were used as medicines. Sporocarps belonging to edible (Kruskal-Wallis test, $\chi^2 = 12.437$, df = 2, $P = 0.002$) and medicinal (Kruskal-

Wallis test, $\chi^2 = 19.077$, df = 2, $P < 0.0001$) macrofungi were significantly more frequent in sacred forests than in the forest reserve and coffee plantations.

Table 5.8 Distribution of macrofungal sporocarps with different ecological traits, and those with utility value to the local people of Kodagu showing observations and percentages (in brackets) in the respective land use type.

Land use type	**(A) DISTRIBUTION OF MORPHO-SPECIES**							
	Ecological characteristics				**Utility value**			
	Habitat preference		**Mycorrhizae**		**Edible**		**Medicinal**	
	L	**W**	**M**	**N**	**E**	**N**	**M**	**N**
Forest reserve	46 (47)	48 (49)	18 (19)	79 (81)	13 (13)	84 (87)	17 (18)	80 (82)
Sacred forests	80 (57)	56 (40)	32 (23)	109 (77)	15 (11)	126 (89)	21 (15)	120 (85)
Coffee plantations	41 (47)	45 (52)	20 (23)	67 (77)	13 (15)	74 (85)	15 (17)	72 (83)

Land use type	**(B) DISTRIBUTION OF SPOROCAPRP OCCURRENCES**							
	Ecological characteristics				**Utility value**			
	Habitat preference		**Mycorrhizae**		**Edible**		**Medicinal**	
	L	**W**	**M**	**N**	**E**	**N**	**M**	**N**
Forest reserve	361 (42.8)	467 (55.4)	68 (8.1)	775 (91.9)	39 (4.6)	804 (95.4)	170 (20.2)	673 (79.8)
Sacred forests	1909 (55.8)	1444 (42.2)	392 (11.5)	3031 (88.5)	200 (5.8)	3223 (94.2)	647 (18.9)	2776 (81.1)
Coffee plantations	269 (32)	541 (64.2)	100 (11.9)	742 (88.1)	92 (10.9)	750 (89.1)	227 (27)	615 (73)

Note: The percentages in 'Habitat preference' columns do not add up to 100 because of the presence of miscellaneous macrofungal sporocarps (coprophilous and entomopathogenic) not shown in the table. The proportions of (A) morpho-species of such macrofungi were 4, 3 and 1; and the proportions of (B) occurrences were 1.8, 2.1 and 3.8 percent in the forest reserve, sacred forests and coffee plantations respectively.

Key to the acronyms: Habitat preference: L – Macrofungi fruiting on litter, W – Macrofungi fruiting on wood; Mycorrhizae: M – putative mycorrhizal, N – non-mycorrhizal; Edible: E – edible, N – non-edible; Medicinal: M – medicinal, N – non-medicinal.

5.2 Discussion

5.2.1 Distribution of macrofungal diversity in the landscape

As in the case of trees and birds, 40% of macrofungal morpho-species are ubiquitous (*Figure 5.1*). However, sacred forests have an unusually high number of unique morpho-species than expected, as revealed by a χ^2 test. Sacred forests have significantly greater variability in canopy height and closure, higher stem density and higher density of lianas in comparison to the reserve, while there is no significant difference in the variability in basal area in the two land use types (*Table 5.9*). Sacred forests offer a greater variety of habitats for macrofungi in addition to the landscape level heterogeneity (Lapin & Barnes 1995), which may be the cause of the greater number of morpho-species unique to those habitats.

Table 5.9 The results of a Mann-Whitney U test for comparison between forest structural diversity between the reserve and sacred forests. Variability in basal area, canopy height and canopy closure was measured as 5-95 percentiles.

Name of the variable	**Forest reserve (n = 8) and Sacred forests (n = 24)**	
	Mann-Whitney U	**P**
Stem density per hectare	27	.003
Variability in basal area	69	.240
Variability in canopy height	38	.030
Variability in canopy closure	46.5	.031
Density of lianas per hectare	18	.001

5.2.2 Effects of landscape modification on macrofungal diversity

Diversity of macrofungal sporocarps (calculated using rarefied species richness for 12 observations) is not significantly different between the forest reserve, sacred forests and coffee plantations (*Figure 5.4*). Therefore, apparently, landscape modification has had no effect on diversity. However, a detailed analysis of the distribution of ecological groups and of landscape factors affecting sporocarp production reveals the following.

Ecological groups

Sporocarps of macrofungi fruiting on litter rather than wood were significantly more frequent in sacred forests; the reverse was true in the two other land use types (*Figure 5.2*). A possible explanation can be found in the different land management practices in the three land use types. Coffee plantations have a large number of standing dead stumps left after cutting canopy trees. The forest reserve, due to its inaccessibility, has a large number of dead logs that are not extracted. Thus, in both these land use types, plenty of dead wood is available in order for macrofungi fruiting on wood to produce sporocarps. By contrast, in sacred forests, the people remove dead logs for fuel or for other domestic uses. This may explain why sporocarps of macrofungi fruiting on wood are significantly fewer in sacred forests.

Effect of patch area

As in the case of tree and bird diversity, patch area was not an important determinant of diversity of macrofungal sporocarps (*Figure 5.5*). Although sporocarp production in two morpho-species was significantly higher in large patches (*Table 5.5*), it's not possible to draw any conclusions about the effect of patch area on their abundances.

Landscape effects

Effect of isolation

Isolation was found to be an important factor for explaining diversity of trees but in the case of birds, the total diversity was not affected. For sporocarp production in macrofungi, isolation was also not important – overall diversity did not correlate with isolation (*Figure 5.6*). Although sporocarp production in one morpho-species was significantly higher in more isolated patches (*Table 5.6*), it's not possible to draw any conclusions about the effect of isolation on its abundance.

Effect of the intervening landscape matrix

CHAID analysis identified the total length of edges in the surrounding 1 km as the most important in determining macrofungal diversity (*Figure 5.7*). Furthermore, there is a good correlation between dew point, the measure of atmospheric moisture content and the total length of edges in the 1 km surroundings ($R^2 = 0.991$, $P = 0.004$), the measure of landscape complexity, suggesting some influence that the surrounding landscape can have on macrofungal sporocarps, since sporocarp production depends among other things, on environmental factors such as atmospheric moisture content (Watling 1995). Furthermore, the longer the total length of edges the greater the landscape complexity, caused by human intervention, and the more complex the mosaic composed of small patches. Thus, atmospheric temperature and moisture (Fogel 1976, cited in Vogt *et al.* 1992), as well as the degree of anthropegenic intervention (Watling 1995), both influence the diversity of sporocarps in Kodagu.

For most morpho-species selected for detailed analysis, the number of sporocarps produced was not influenced by landscape complexity (*Table 5.7*). Although sporocarp production in one morpho-species was significantly higher in large patches (*Table 5.6*), it's not possible to draw any conclusions about the effect of the intervening landscape matrix on its abundance.

It is not possible to draw any conclusions on the ecological niches of morpho-species without a detailed study. In addition, the results presented here may simply be random effects on sporocarp abundances of morpho-species – especially because R^2 values are very low.

5.2.3 Approaches to conservation of macrofungal diversity

Watling (1995) compared fungal lists from a number of different areas within the UK and continental Europe and indicated that ectomycorrhizal fungi generally constitute 35-40% of the total number of macrofungal morpho-species. In the present study, sporocarps of putative mycorrhizal macrofungi were significantly more frequent in sacred forests than in the forest reserve and coffee plantations as revealed by the Kruskal-Wallis test (see *p. 158*). Dighton & Mason (1985) and Jansen & Denie (1988) have reported that canopy closure in coniferous plantations decreases the diversity of mycorrhizal fungi. Furthermore, places with open canopies have small numbers of broad host range macrofungi (early stage fungi) while under closed canopies, macrofungi with narrow host ranges (late-stage fungi) appear. Species richness of macrofungi is generally higher in mature forests than in successional forests in Switzerland, however, sporocarp production in the latter was almost double (Senn-Irlet & Bieri 1999). Canopy closure appears to play a key role in determining the occurrence of mycorrhizal macrofungi in Kodagu. Sacred forests have a significantly greater variability in canopy closure than in the forest reserve as revealed by the Mann-Whitney U test (*Table 5.9*). Therefore, the forest reserve, with more uniform canopies has fewer mycorrhizal fungi compared to sacred forests. It was not possible in the present study to determine the early or late successional nature of mycorrhizal macrofungi because this will require long time-series data. Senn-Irlet & Bieri (1999) speculate that mycelia of late-successional mycorrhizal fungi persist in disturbed localities (such as coffee plantations) and re-connect to the host trees, despite the disturbance to the soil during plantation operations. It is likely that a large set of late-stage mycorrhizal fungi in nearby sacred forests function as a source of continuing propagule supply for the damaged sites. Mycorrhizal fungi are generally considered to be negatively influenced by increased nutrition (Harley & Smith 1983). Landscape modification is known to impoverish soils (*e.g.* Klein 1989) because of reduced rates of nutrient cycling. For this reason, the reserve may be poorer in mycorrhizal fungi than the patches. The higher habitat heterogeneity that the sacred forest patches offer at the landscape scale may be important for the fruiting of mycorrhizal macrofungi (Senn-Irlet & Bieri 1999). As a consequence, sporocarps of mycorrhizal fungi are significantly more frequent in forest patches than in the reserve.

Sporocarps belonging to edible and medicinal macrofungi were significantly more frequent in sacred forests than in the reserve and coffee plantations as revealed by the Kruskal-Wallis test (see *pp. 158-159*). The patches of forest in the landscape are therefore a useful resource of these non-timber forest products for local people.

These observations highlight the importance of the fragmented landscape in Kodagu in meeting the interests of the scientific community (for protecting mycorrhizal macrofungi) as well as the local communities (for protecting useful macrofungi), both interested in their conservation for different reasons.

5.3 Conclusions

5.3.1 Implications for conservation practice

This study has shown that heterogeneity at a landscape scale can be significant in determining diversity of macrofungal sporocarps. The fragmented landscape in Kodagu, with a mosaic of paddy fields, coffee plantations, sacred forests and other land uses, is highly heterogeneous. The patches host a variety of fungi, including those that are mycorrhizal, due to the spatial heterogeneity that they maintain. In addition, sacred forests provide a source of sporocarps of edible and medicinal macrofungal morpho-species for the local people. At a landscape scale, the patchy and the contiguous parts of the landscape play important role in maintaining macrofungal diversity.

Sacred forests offer landscape level heterogeneity and high habitat diversity, and thus harbour a variety of macrofungal morpho-species, many of them unique to one or a few patches. For instance, an undescribed morpho-species of *Xylaria*, an Ascomycete fungus was found in a sacred forest in the village of Nemmale in Kodagu in July 1999. This produced conspicuous, yellow sporocarps about 30 cm tall and was abundant in the Nemmale sacred forest, it was not found at any other locality in Kodagu. Isolated patches also support a large number of mycorrhizal macrofungi. In addition, sporocarps of economically important macrofungi – edible and medicinal – are abundant in sacred forests, providing a valuable resource to the local people.

The forest reserve is an important habitat for macrofungi such as the one that was found growing exclusively on elephant dung at Theralu forest reserve in August 1999 (personal observation). Since large mammals such as elephants are unlikely to survive in the modified Kodagu landscape, the contiguous forest reserve is the only suitable habitat for them; macrofungal morpho-species with co-evolved interdependencies such as this may not survive without their host species.

Like trees and birds, macrofungi are known to face several threats, although information on their threat status is rarely available. In addition to the atmospheric pollution, soil acidification, eutrophication and increased heavy metal concentrations, the composition of macrofungi is also known to change from habitat destruction (*e.g.* Komonen 2001), landscape transformation and land use change (Arnolds 1989, cited in Rydin *et al.* 1997). Unfortunately, knowledge about the macrofungi in the Western Ghats is scarce, let alone information on threat status. However, with the present understanding of fungal diversity in Kodagu, this study has emphasized that the maintenance of landscape-level heterogeneity is a key to conserving mycoflora.

5.3.2 Ideas for future research

The results presented in this chapter are limited to the diversity of macrofungal sporocarps. It is difficult to comment on the patterns of diversity in microcsopic fungal mycelia based on the diversity of sporocarps. Nevertheless, a study such as this may be able to provide information on which more detailed studies can be based. While the overall diversity patterns in macrofungi are discussed in this chapter, the discussion on species-specific analyses have predominantly been speculative due to the lack of authentic information on ecology. It is hoped that this work will provide an empirical example of a study on macrofungi in a modified tropical landscape and direct future research aimed at conserving macrofungal diversity. This study has a variety of shortcomings; and it is recommended that future research should address them:

- Many macrofungal morpho-species fruit sporadically with no consistent pattern of occurrence from year to year (Watling 1995). Furthermore, their sporocarps are ephemeral which even when produced may last only a few days before decomposing or being eaten. Intensive surveys of individual areas have indicated that many years may be required to define accurately macrofungal communities of a particular area (Tofts & Orton 1998). Considering the unpredictability of sporocarp production in macrofungi, the results on the patterns of diversity, without a detailed study may be merely speculative. Therefore, long term studies that should involve sampling and documentation of sporocarps over many years from the same locality, are necessary.

- Specific studies on individual macrofungal morpho-species are essential in order to understand their ecology. For instance, there is little agreement on how canopy closure may affect sporocarp production in different morpho-species. While most are known to produce more sporocarps in open forests than in the closed ones (Senn-Irlet & Bieri 1999), truffle production in Gasteromycetes fungi is said by Vogt *et al.* (1981) to decrease under open canopy. However, another study by Waters *et al.* (1997) showed that there was no difference in production under open or closed canopies. Thus, not only do responses of macrofungi differ across species groups but also across different studies. A detailed study of individual species at a specific locality can help to predict the effects of landscape modification on the species more accurately.

- The analysis of mycorrhizal macrofungi presented in this study is highly speculative. It should be supplemented with a detailed soil sampling for ectomycorrhizae (*e.g.* Jonsson *et al.* 1999; Grogan *et al.* 2000) in order to establish evidence of their presence, and identities. In addition, documentation of the disturbance history, and monitoring of human activities is necessary in order to understand their diversity, because for mycorrhizal macrofungi that are both, host-specific and endemic, land-use history must be an important determinant of diversity – soil disturbance or removal of host trees in the past could reduce diversity irreversibly (Watkinson 2002 *pers. comm.*).

6 . BIODIVERSITY CONSERVATION

6.1 The State-of-the-Art

In the present study, no congruence was found in diversity of trees, birds and macrofungi (calculated using rarefied species richness) (*Table 6.1*).

Table 6.1 Correlation matrix of species diversity (rarefied species richness) between the three groups of organisms; R indicates Pearson's correlation coefficient at the *P* level of significance and n = 58.

Correlations	Tree diversity	Bird diversity	Macrofungal diversity
Tree diversity (≥ 10 cm dbh)	-	-	-
Bird diversity	R = - 0.072 (*P* = 0.591)	-	-
Macrofungal diversity	R = 0.011 (*P* = 0.936)	R = - 0.124 (*P* = 0.352)	-

Mantel's tests for associations between the distance matrices (calculated using Sorenson's quantitative similarity index for localities in the reserve, and those at < 1.5, 1.5-5 and > 5 km away from the reserve) were non-significant (*Table 6.2*).

Table 6.2 Association between similarity matrices for trees (≥1 cm dbh), birds and macrofungi; R is standardised Mantel statistic at the *P* level of significance; T indicates the type of association (positive or negative) between distributions using Mantel's (1967) asymptotic approximation, n = 32.

Correlations	R	T	*P*
Trees (≥ 1 cm dbh) and Birds	0.035	0.070	0.944
Trees (≥ 1 cm dbh) and Macrofungi	0.405	0.997	0.319
Macrofungi and Birds	- 0.612	- 1.420	0.156

This indicates the possibility that entirely different processes govern the distributions of trees, birds and macrofungi. The biological and ecological characteristics of the groups of organisms in question play an important role in determining those processes. Trees and macrofungi are immobile while birds are mobile. As a result, there are no differences in the similarity of birds between the forest reserve and sacred forest localities (*Figure 4.8*). For trees and macrofungi, the localities within the forest reserve show some patterns of clustering (*Figures 3.12* and *5.9*), although this may be due to clumping of sampling localities within the reserve (*Figure 3.8*).

Trees cannot regenerate in coffee plantations due to intensive land management. However, they can be involved in gene flow with the reserve and sacred forest patches in the tree-covered Kodagu landscape. For many bird species isolation of patches is not a significant barrier. So, the habitat preference of a bird will determine where it spends most time foraging. Bird populations will depend on access to sufficient habitat of high quality. As a result, quality of the landscape in the immediate surrounding plays an important role in determining bird diversity. For macrofungi, sporocarp production depends on environmental factors. However, with the information available on the ecology of macrofungi in the Western Ghats, it is arguable whether the production of sporocarps reflects actual patterns of macrofungal diversity.

The poor correspondence between species diversity in different taxa implies that conservation priorities determined on the basis of richness in a few taxonomic groups cannot be relied upon to capture similar patterns in other groups (*e.g.* Kerr 1997; Reid 1998). Prendergast *et al.* (1993) found only 34 percent overlap in diversity of five different taxonomic groups – butterflies, dragonflies, liverworts, aquatic plants and breeding birds – in the UK. Oliver & Beattie (1996); Lawton *et al.* (1998) have shown in tropical regions that no one group can serve as a good indicator taxon for the diversity of other groups. These studies rely on information collected using different methods on various spatial scales, leaving a possibility for error. The data used in the present study were collected from the same landscape, in the same habitats, using similar methods – nevertheless, the results support Prendergast *et al.* and Oliver & Beattie's findings of the lack of congruence between diversity of different taxonomic groups. The findings of the present study with reference to tree, bird and macrofungal diversity are summarised in *Table 6.3*.

Table 6.3 The effects of long-term landscape modification and land management on trees, birds and macrofungi in Kodagu; an asterisk denotes a significant ($P < 0.05$) difference; the forest reserve is abbreviated as 'Reserve', sacred forests as 'Sacred' and coffee plantations as 'Coffee'.

	Trees	**Birds**	**Macrofungi**
Effects of landscape modification on communities			
Intensity of land management on diversity	No effect	No effect	No effect
Intensity of land management on number of unique species	More species in Reserve*	No difference between Reserve, Sacred and Coffee	More morpho-species in Sacred*
Patch size on diversity	No effect	No effect	No effect
Isolation on diversity	Decline	No effect	No effect
Intervening landscape matrix on diversity	No effect	Matrix in the immediate surroundings of patches important for explaining bird diveristy*	Matrix in distant surroundings important for determining macrofungal diversity*
Distribution of ecological groups			
Habitat preference	Deciduous trees increase and evergreen trees decrease with intensity of land management*	Non-forest dwellers increase and forest dwellers decrease with intensity of land management*	Macrofungi fruiting on wood less frequent and those fruiting on litter more frequent in Sacred*
Body size in birds	NA	Large-bodied birds more abundant in Coffee and small-bodied in Sacred*	NA
Fecundity in birds	NA	Birds with low fecundity more abundant and those with high fecundity less abundant in Reserve*	NA
Feeding guilds in birds	NA	Understorey insectivores less abundant in Coffee; nectarivores more abundant in Sacred*	NA
Mycorrhizal fungi	NA	NA	More frequent in Sacred
Distribution of species of conservation importance			
Endemic	More abundant in Reserve*	No difference	No information
Threatened	More abundant in Sacred*	Insufficient sample size	No information
Useful	More abundant in Coffee*	NA	More frequent in Sacred*
Medicinal	More abundant in Coffee*	NA	More frequent in Sacred*

Unlike many tropical landscapes where forests have been lost to agriculture and other land uses, resulting in the alarming loss of species (Laurance *et al.* 1997), Kodagu seems to have retained much of its biodiversity despite the long term land management. The high proportion of native tree cover in Kodagu, unlike many other fragmented landscapes, is important because it provides an invaluable buffer for forest-dwelling organisms.

1. For immobile organisms, the tree cover reduces the severity of the effects of microclimatic changes – higher temperatures, increased wind-speed, lower humidity and lower soil moisture (Geiger 1965; Kapos *et al.* 1997; Freidenberg 1998) – associated with landscape fragmentation. As a result:
   - There is less edge-related disturbance in forest patches;
   - There is more habitat available for forest interior species and less for the disturbance-adapted species;
   - Sacred forest patches support more forest-interior species than would be the case in a landscape where forest patches are surrounded by agricultural land.

2. For mobile organisms, the tree cover facilitates movements through the landscape matrix, which organisms can also use for foraging and other resources. As a result:
   - Samples in coffee plantations include transitory individuals and individuals from populations that are non-viable within plantations alone, as well as more individuals of forest interior species than it would be the case in a purely agricultural landscape matrix;
   - The diversity of samples in coffee plantations is higher than would be the case in an agricultural matrix;
   - Population densities in sacred forests are higher than in equivalent patches surrounded by agricultural land because they include individuals that gain some of their resources from coffee plantations.

Andrén (1994) found that in landscapes with less than 30% coverage of suitable habitat, habitat loss was a good predictor of diversity and the effects of patch isolation and size were important. However, in landscapes with higher habitat cover, habitat loss was not a good predictor of diversity in birds and mammals. O'Neill *et al.* (1988) in their spatial analysis observed that above 30%, suitable habitat becomes almost continuous across the landscape. Many organisms will be able to move easily across such landscape, where connectivity is high. Furthermore, for highly mobile organisms this critical threshold will be much lower than for relatively less mobile ones. As a consequence of the relatively homogenous landscape, the sacred forests in Kodagu are not actually perceived as 'patches' by organisms using them. Therefore:

- Diversity in all the three groups of organisms studied was very similar between the forest reserve, sacred forests and coffee plantations despite the variation in the intensity of land management; many forest interior species were found in all the land uses studied and the types of species were relatively similar in the three land uses.
- In a landscape such as Kodagu, heuristic frameworks that are preoccupied with patch size and isolation are not suitable for explaining the patterns of diversity (*e.g.* Wiens 1997).

In view of the above, maintaining the integrity of the three land use types studied is critical for the conservation of Kodagu landscape.

While the long-term human use has caused landscape modification, the forest reserve is still relatively intact. It accounts for the protection of species with special habitat requirements – 63% of trees species unique to the forest reserve are endemic, in contrast to only 10% of those found in sacred forests and coffee plantations, but not in the reserve. At the same time, sacred forests and coffee plantations together have successfully protected trees useful to the local people – 84% of the tree species found in sacred forests and coffee plantations, but not in the forest reserve, produce useful non-timber forest products.

The forest reserve is instrumental in protecting many species of conservation importance – the Western Ghats' endemic trees, birds with low fecundities and possibly fungi with co-evolved interdependencies such as the one found exclusively growing on elephant dung (personal observation). Conservation of such species cannot take place without the reserve. The removal of coffee plantations from the Kodagu landscape or the replacement of native shade trees with exotics would result in rapid depletion of biodiversity. The disappearance of trees in coffee plantations that are useful and have medicinal values to the local people would result in resource-use pressure on sacred forests and the reserve. The matrix of coffee plantations is likely to change in future, in species composition, unless dwindling populations of native trees are replaced by proactive planting. In the present study, sacred forests were found to be situated in a variety of habitats – some were adjacent to water-logged paddy fields, others were on hill-slopes and still others on tree-covered hill tops. These patches protect a greater variety of habitats than a single protected area would (*e.g.* Quinn & Harrison 1988; Lapin & Barnes 1995). The loss of sacred forests would result in the decline of landscape level heterogeneity that these patches provide.

A study on butterfly diversity in the southern Western Ghats found that about 46% of the butterfly species are in low abundance in a few patchily distributed locations. Species such as the Travancore Evening Brown (*Parntirrhoea marshalli*) or the Malabar Banded Swallowtail (*Papilio liomedon*) are restricted to low elevation evergreen forests in the southern Western Ghats that are excluded from the protected area network (Soubadra Devy & Ganeshaiah 2001 *pers. comm.*). In the present study, tree species such as *Actinodaphne lawsonii*, *Hopea ponga*, *Madhuca neriifolia* and *Syzygium zeylanicum* that are listed as threatened in their home ranges (FRLHT 1999; IUCN 2000), are unique to sacred forest patches. Other threatened species such as *Michelia champaca* and endemic species such as *Pittosporum dasycaulon* are found in sacred forests and coffee plantations but not in the forest reserve (*APPENDIX 4*). While these species are restricted to the low-lying areas, the current protected areas along the western hilly region of the district do not include their preferred habitats. In addition, bird species such as the Loten's sunbird (*Nectarinia lotenia*), an endemic, and the Nilgiri Flycatcher (*Eumyias albicaudata*), an endemic and threatened bird, are restricted to sacred forests and coffee plantations (*APPENDIX 5*). Forty nine out of 163 morpho-species of fungi are unique to sacred forests (*APPENDIX 6*). It is likely that the modified landscape in Kodagu shelters species that cannot be protected within the formal reserve network (Bhagwat 1998), and conservation of such species will require protection of land outside the forest reserve. However, more extensive data are necessary to substantiate the findings of the present study concerning the distribution of species in the Kodagu landscape.

One way to achieve conservation of species outside the boundaries of the current reserve network would be to expand the network to the low-lying areas. This would protect species outside the current reserve structure. However, such an approach is more difficult to implement than community-based conservation because firstly, the expansion of protected areas would require financial resources in order to compensate the people for the foregone development cost, to employ staff and to maintain the necessary infrastructure that the State may not be able to afford; secondly, imposing such expansion of strictly protected areas on villagers would cause a lot of resentment. In addition, although currently the forest reserve that buffers the protected areas appears to be well-protected, a relatively minor change in policy can make it available for non-forest land uses, if the inclination of the ruling governments is towards opening access to the state forest resource for their own vested political interests (*cf.* Jepson *et al.* 2001). A more pragmatic alternative would be to expand the network of sacred forest patches by creating continuous tree cover, or restoring the populations of dwindling native trees, using appropriate silvicultural techniques in the matrix between groups of sacred forests that are far away from the reserve. This will provide a network of small-scale reserves in the parts of the landscape that are currently intensively used. Such networks can be maintained through the initiative and involvement of local people rather than by force.

In many parts of the world the fate of biodiversity, it is now believed, depends on the management of remnants of forest in man-made landscapes (Saunders *et al.* 1987; Brussard *et al.* 1992). Conservation measures in agricultural landscapes often emphasise creation of 'stepping stones' of native woodlands for maintaining biodiversity (*e.g.* Hobbs & Saunders 1993; Schwartz 1997). However, the findings of the present study imply that it is also important to maintain tree cover in the matrix between them. The long-term survival of biodiversity will depend on the effectiveness with which landscape between habitat patches can be managed – Bawa & Ashton (1991) have speculated that from a genetic perspective, conservation of biodiversity can be realised through ensuring gene flow by maintaining a variety of managed forests and farming systems in fragmented landscapes. From the forest management perspective, Whitmore (1997) has recommended that management and land use regimes that maintain biodiversity should be designed for the long-term conservation in landscapes where humans are an integral part. Therefore, the involvement of local people in management is the key to successful biodiversity conservation.

6.2 Strategy for biodiversity conservation

It appears that Kodagu has been successful in maintaining a balance between conservation and development through the formal (state-initiated conservation) and the informal (community-based conservation) systems, besides a comparatively thriving coffee economy (private enterprise) that makes the region prosperous. However, currently, there are threats to the integrity of the forest reserve as well as of sacred forests and coffee plantations.

6.2.1 Current problems in Kodagu

Unplanned forestry activities and illegal smuggling of timber has made the current system of protection of the forest reserve vulnerable. Unlike the haphazard timber operation before Independence – particularly during the second World War (Akbar Sha 1987), the post-Independence working plans for the forest reserve, by the State Forest Department, implemented conservation measures such as the prevention of cardamom cultivation inside the reserve, restriction to selective felling only in certain areas and a legal ban on timber exploitation from other areas (Somiah 1954). However, after the expiry of the plan in 1974, and before the new working plan was written in 1987, the guidelines in Somiah's plan were totally neglected and unplanned forestry operations were carried out in order to obtain timber (Akbar Sha 1987). Despite a moratorium on logging in hilly areas imposed by the Indian government since 1983 (Ministry of Environment and Forests 1985) timber was, and is still illegally extracted from these forests. Furthermore, for cultural reasons, gun ownership is legal in Kodagu (Ponnappa 1997), which has exacerbated illegal hunting.

While the Government has always been the custodian of the sacred forests, the local people have had a say in their management. For instance, the law gave the local people "right to take fire-wood for the temple, worship materials for constructing pendals, and with special permission, timbers for repairing the temple [were] allowed to the temple authorities and servants, while the villagers generally [had] the right of way, water and grazing" (Haller 1910). The tenure of sacred forests was transferred from the State Forest Department to the Revenue Department in 1905 until 1985, when they were again transferred to the State Forest Department. The extent and quality of the woodland in these sacred forests is reported to have suffered due to their misappropriation under the 'open access' regime of the revenue department (Kalam 1996), in accordance with the 'tragedy of the commons' model described by Hardin (1968). This has been attributed to the careless policy changes and unprecedented tenure transfer (from the Forest to the Revenue Department) between government departments resulting in the confusion among local people about the actual ownership and exact boundaries of sacred forests (Kalam 2000). In addition, the unobtrusive temples in sacred forests have been replaced by sophisticated concrete constructions (Chandran *et al.* 1998) in the process of *Sanskritisation* (transformation of local belief system and its homogenisation with the main stream Hindu religious practice) (Kalam 1996). This has caused decline in the quality of vegetation within sacred forests.

After coffee cultivation was privatised in the mid-1990s, the small landowners in the study area were able to sell their produce on the free market making the venture more profitable. Subsequently, the Human Development Index[19] of Kodagu (Narayana 1999) increased far beyond that of other rural parts of Karnataka state and the rest of India. In order to maintain the increased standard of living, valuable native tree species were illegally felled for timber and exotic shade trees such as *Grevillea robusta* and *Eucalyptus* spp. were planted in coffee estates. In Kodagu, a growing demand for coffee on the free market encouraged planters to intensify coffee cultivation resulting in modification of forest wherever circumvention of local rules and regulations was possible. As a result, more shaded cardamom plantations were converted into coffee after thinning native tree-growth. Simultaneously, paddy fields were converted into ginger, banana and areca nut plantations. In addition, sacred forests were encroached upon by the land-grabbing landowners in their neighbourhoods (Bonn 2000).

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[19] Human Development Index is a composite index covering life expectancy at birth (and indicator of general health of people), literacy plus enrolment (educational attainment) and per capita income (standard of living). It has widely been accepted as a better measure of well being of the people when compared to the traditional measure of per capita income introduced by the United Nations Development Programme (UNDP) in 1990.

6.2.2 Possible solutions

A three-prong strategy for the conservation of biodiversity in Kodagu is proposed. This includes preventative measures for activities that result in the modification and supportive ones for those that achieve conservation, in the three land use types studied, so as to accomplish the maintenance of biodiversity at the landscape level.

Prevention of hunting and timber-smuggling from the forest reserve

Illegal hunting and timber smuggling are the two activities that currently pose the greatest threat to the forest reserve, which, in the present study was found to be valuable in protecting species of conservation importance. In addition, Akbar Sha (1987) and Forest Survey of India (1995) also mention the role of the forest reserve in protecting the populations of large mammals.

Control of illegal hunting and logging

Besides the local people, poachers and timber smugglers from the neighbouring state of Kerala are involved in the illegal activities. Currently there are regulations designed to prevent hunting and logging – the moratorium on logging since 1983 and the Wildlife (Protection) Act amended in 1991. However, the Forest Department has insufficient manpower to police illegal activities because of the vast expanse of the forest reserve. Along with a strong legislation at the state level (that complies with the international initiatives such as CBD[20] and CITES[21] - The latest developments in India in that direction are: The Biodiversity Bill, 2000[22] and the National Biodiversity Strategy and Action Plan, NBSAP[23]), governments should develop and implement innovative incentives to interest local communities in forest stewardship (*e.g.* Alcorn 1996). The co-operation in preventing and monitoring illegal forest exploitation should be linked to concrete benefits (*e.g.* Glastra 1999). Currently, there is a practice of recruiting

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[20] Convention on Biological Diversity.
[21] Convention on International Trade in Endangered Species of Wild Fauna and Flora.
[22] http://www.lawcommissionofindia.nic.in/biod.htm
[23] http://sdnp.delhi.nic.in/nbsap/

members of the local community as temporary 'forest watchers' in Kodagu in return for small monetary compensation. The Government should introduce more secure employment as an incentive for the qualified local inhabitants to work as guards and officers in the Forest Department. In order to secure the forest reserve from illegal logging by outsiders, other measures such as the certification of timber should be enforced.

Control of recreational hunting by local people

The depletion of wildlife is partly attributable to recreational hunting by local people (Akbar Sha 1987). It has been a common experience that management systems are likely to fail if there are changes in technologies, such as the introduction of guns (Redford 1992, cited in Brandon 1997), although such changes are inevitable when there is technological revolution in the world outside. One of the ways for tackling this problem would be to implement changes in the legislation for unrestricted licensing of guns in Kodagu. The beneficiaries of modern technological change are often the urban middle-classes. On the other hand, the people living in villages near to the forest reserve boundary often have their crops damaged by wild boars and elephants and need to possess firearms in order to protect their livelihoods. The licensing legislation should therefore be able to recognise those who are in real need of guns for self-defence. In addition, hunting with traditional weapons (subsistence hunting) should be allowed, as it does not cause as much damage to wildlife as sophisticated modern weapons.

Alternative sources of timber for local people

Jennings *et al.* (2000) have argued that the only viable strategy for conservation of Mahogany (*Swietenia macrophylla*) forests in the Brazilian Amazon is to find sustainable livelihoods for rural populations. In Kodagu, unless local people have access to timber, conservation of the rest of the forest area cannot be effectively implemented. Therefore, small-scale logging strictly for non-commercial purposes should be legalised in accessible areas. Furthermore, in the present study, coffee plantations in Kodagu were found to have been a valuable source of timber and non-timber forest products, and will continue to be so if managed responsibly. Trees do not regenerate in coffee plantations because of intensive land management, thus coffee plantations are likely to change in future, in tree species composition. Therefore, proactive planting of native timber trees by coffee plantation owners is essential.

Maintaining the integrity of coffee plantations

Larson & Bromley (1990, cited in Miranda & LaPalme 1997) have suggested that when market prices are exogenous and do not reflect local scarcity, intensified agricultural production to meet subsistence needs can lead to resource degradation. In Kodagu, the privatisation of coffee market has had exactly this effect.

Revitalisation of co-operative institutions

The co-operative institutions such as The Coffee Board that used to exist in the late 1990s are now defunct. There is a direct correlation between privatisation of the coffee market and modification of the native tree cover, which planters look at as an alternative source of income when coffee prices dwindle. Co-operative institutions are necessary to regulate coffee prices in the domestic market as well as to bargain for good prices so that small planters have a stable income. In this way, there will be fewer fluctuations in coffee prices, an assured income for small planters and control of the depletion of native trees in Kodagu.

Disincentives for planting exotic trees

Evidence form Latin America suggests that although shaded coffee plantations help to preserve biodiversity (Greenberg *et al.* 1997; Moguel & Toledo 1999), a recent trend among planters, of replacing native trees with exotics, has been motivated by increased coffee yields per hectare (Perfecto *et al.* 1996, cited in Primack 1998). In Kodagu better irrigation facilities have allowed farmers to fell native trees (which retain moisture in the plantations due to their dense foliage) and replace them with exotics (which often have sparse canopies). Trees such as *Grevillea robusta* have straight boles, which can be easily used to train black pepper (*Piper nigrum*) vines, an important source of additional income for coffee planters. A system of sanctions – in form of higher taxes and no subsidies – on planters who use exotic trees in coffee plantations (and thereby violate ecological consideration in plantations management) could be enforced in order to prevent the native tree cover from dwindling further.

Support for planting native trees

Plantation owners could be encouraged to maintain tree cover of native species because of its benefits to wildlife. In order to facilitate this, the Government should make available saplings of native shade trees from the State Forest Department nurseries. The Government can develop incentive systems by way of cash rewards, reduced taxes, greater subsidies *etc.* for those landowners who employ ecological consideration in plantations management.

'Biodiversity-friendly' coffee cultivation

It has been shown in South America that if the local landowners reach some basic level of economic security, they become interested in ecological sustainability, long term environmental planning and biodiversity conservation (Southgate & Clark 1993). Gobbi (2000) has suggested that in El Salvador, for biodiversity conservation in shade coffee plantations to be viable, incentives to small farmers from the government are necessary. These could be in form of tax reductions, loan facilities, subsidies and a secure market for biodiversity-friendly coffee. For such cultivation practices to be successful in Kodagu, it is necessary that the Government takes interest in the issues of small-scale farmers in order to encourage them to help biodiversity conservation. Furthermore, certification of coffee plantations by setting standards for sustainable management can encourage planters to grow coffee in a biodiversity-friendly manner. Market access of certified products should be facilitated by specific Government policies and appropriate legislation. In addition, market-driven mechanisms *i.e.* effective marketing of 'biodiversity friendly' coffee in the international market can complement the policies and the legislation.

Participatory management and conservation of sacred forests

Miranda & LaPalme (1997) suggest that if there is a strong tradition of community-based ownership or customary rights, governments should take advantage of such relationships for effective biodiversity conservation. Despite the disputed effectiveness in biodiversity conservation, community-managed traditional reserves are one of the few mechanisms available that allow local groups to manage natural resources over the long term and to achieve a sustainable linkage between conservation and development (Brandon 1997).

Official recognition of sacred forest traditions

In modern-day India, although many traditions are eroding, a large number of sacred forests are still conserved through taboos and religious beliefs (Malhotra *et al.* 2001). Chandrakanth *et al.* (1990) suggest that such sacred acts should be recognised by the Government. Bhatt & Kothari (1997) identify the incompleteness of the IUCN protected area categories (*APPENDIX 1*) and suggest an additional category, 'Community Reserve', that is appropriate in the Indian context. It is recommended that such areas should be allocated mostly for the protection of traditional conservation practices, besides controls on limited resource use and protection against outside commercial forces. While official recognition at the national and the international level is important for land-tenure security of sacred forests to the stakeholder communities, participation of local people in the management is also essential.

A system of rewards for effective protection

A large number of sacred forests in the southern part of Kodagu (where the study took place) are still well protected (Kushalappa & Kushalappa 1996). Maintaining sacred forests is important to the local people as well as to the community outside (Chandrakanth & Nagaraja 1997) because of the cultural and spiritual appeal that such landscapes have (Posey 1998). However, sacred forests are currently under threat of encroachment by coffee plantation owners (Bonn 2000). In such a situation, a system of rewards may work for the effective protection of sacred forests (Gadgil & Rao 1994).

The success of protection could be assessed periodically by a panel of impartial judges, and the village committees that are making sincere efforts at conserving their village sacred forests could be rewarded by the State (*e.g.* Chandrashekara & Sankar 1998). The funds thus generated will help local people to undertake development activities in villages. Non-governmental organisations in Kodagu have a very good rapport with the local people. Their involvement in co-ordinating the system of positive incentives is essential.

A mechanism for sharing of benefits

While the ownership of sacred forests should rest initially with the State Forest Department, in order to avoid idiosyncratic management decisions (such as misappropriation of land for other land uses) by individuals or committees, representatives from the local community should be involved in making decisions about access to the resources in such forests, so that no particular social group can monopolise their management. Sacred forests are important resources for non-timber forest products. Local people depend on them for fuelwood, green fodder, medicinal herbs and other livelihood necessities. An organised system for harvesting, utilising and marketing such products is essential. It should be recognised that the local people will not be able to operate entirely on their own – the State should, apart from facilitating their efforts and assisting with planning and management, oversee the smooth running by resolving conflicts among the local communities (*e.g.* Gadgil 1992). The profit generated from such enterprises should be shared equitably between the State and the local community. Such efforts need to be complemented by appropriate legislation at the State and National levels that can provide the necessary land-tenure security (such as devolution of ownership to those local village committees who have been successful in conservation of their sacred forests) and resource-use rights to local communities, while recognising local peoples' rights as a precondition (Chambers *et al.* 1989).

Towards joint planning and management

Khare *et al.* (2000) have found that foresters and villagers in some parts of India view Joint Forest Management (JFM) very differently – many forestry department officials see JFM primarily as a means of ensuring the rehabilitation of degraded forests, while village communities view it as a solution to the growing shortage of biomass, a means of obtaining the daily requirements of forest products and a way to increase income. Therefore, it is necessary to bring foresters and villagers together to initiate dialogue and to reach agreement on the objectives of JFM of sacred forests. Currently, JFM is not recognised as a way to manage government protected areas in India (Khare *et al.* 2000). The State Forest Department should explore ways in which the JFM philosophy could be used in managing the forest reserve in Kodagu.

Recently, there have been some serious attempts in Kodagu at adapting JFM to the local socio-cultural setting – a scheme of joint forest planning and management of sacred forests, which was proposed through a public initiative, has been accepted by the Government (Kushalappa 2001 *pers. comm.*). This scheme proposes to form a federation of sacred forest committees (who look after the management of local sacred forests), also consisting of local forest officers and community leaders. In order to mediate the negotiations between foresters and villagers, impartial representatives from the local Forestry College and active non-governmental organisations are also members of the committee and participate in meetings (Kushalappa *et al.* 2001). Efforts such as this will be important in protecting the integrity of the cultural landscape in Kodagu.

Financial arrangements

It is recognised that maintaining traditional conservation areas in the landscape can be costly because the people will incur opportunity costs from foregone development activities (Brown 2000). In addition, local governments do not have funds to finance such conservation areas, and to promote biodiversity-friendly agricultural practices. If the world community wants tropical developing countries to protect more of their natural forests for conservation of biodiversity, these costs should be borne by the citizens of the global community, who indirectly benefit from protection. Because the conservation areas do not generate the magnitude of cash flows that agriculture does, innovative means of financing such areas will be necessary, including financial mechanisms that involve the international movement of funds from benefactors located far away from the reserves where conservation activities take place (Kramer & Sharma 1997). The debt-for-nature swaps are proposed as one such way, where developing nations get a waiver on their international debts in return to conserving nature and natural resources for the greater common good (World Bank 1994, cited in Kahn & McDonald 1997). Although Kahn & McDonald (1997) caution that their success has been disputable in African and Latin American countries, debt-for-nature swaps have rarely been tried in Asia. If such swaps are successful in Kodagu, the Governments can utilise such funds to provide economic incentives to local inhabitants for biodiversity conservation. Carbon-offset projects have become another increasingly popular way of generating revenue from forest conservation and management in developing nations (Putz & Pinard 1993). Such funds can be deposited in a local trust, for instance the federation of sacred forest committees in Kodagu, in order to insulate conservation from government deficits and changing political priorities.

6.3 Beyond Kodagu

It would be arrogant to present a definitive list of recommended courses of action that would be guaranteed to conserve tropical biodiversity in other similar forested landscapes – systems are appropriate only within their own cultural and ecological contexts. Therefore, deriving management lessons for biodiversity in the tropics, based on the cultural landscape in Kodagu is not within the scope of the present study. Yet, the example of Kodagu can provide some guidelines for conservation philosophy in other similar landscapes.

Conservation outside protected areas

Human use of landscapes is a reality – people have been a part of virtually all the world's ecosystems for thousands of years – this must be considered in the design of any conservation effort (Kramer *et al.* 1997). The concept of cultural landscapes recognises the complex interrelationships between humans and nature in the construction, formation and evolution of landscapes. Cultural landscapes and their links to the conservation of biodiversity are recognised under the 1972 UNESCO[24] convention concerning the protection of the world's cultural and natural heritage – The World Heritage Convention (Posey 1998). In the most optimistic scenario, 10% of the world's land will be protected according to the IUCN recommendations (IUCN 1994). The target has nearly been achieved by the end of the 20th century (WCPA 2001). However, the remaining 90% of the land area is still left behind outside protected areas. Because most land in most countries will never be protected, numerous species will inevitably occur outside these areas. Western (1989) is of the opinion that if nature outside protected areas cannot be saved, not much will survive inside. The landscapes in the immediate surroundings of protected areas are used by local inhabitants – traditional activities such as hunting and gathering, agriculture, religious practices and pastoralism continue as part of livelihood necessities of the local people (*e.g.* Gadgil *et al.* 1993). Thus, strategies for reconciling human needs and conservation interests in the land outside protected areas are critical for successful biodiversity conservation (Gradwohl &

Greenberg 1988; FAO 2001). Adams (1998) advocates that conservationists should recognise where traditional conservation has a place, create the institutions that will pay for it and the value systems that will give it legitimacy.

Conservation within protected areas

The importance of taking needs, ideas and aspirations of local people seriously in conservation planning within protected areas is now widely recognised (MacKinnon *et al.* 1992; Kothari *et al.* 1996; Adams 1998). It is recommended that governments should form and strengthen partnerships with local communities. McNeely (1996) offers some suggestions as to how this can be done. First, he argues that a protected area must provide direct benefits to the local people and that the local people should be involved in the planning process. Second, protected areas should be planned and managed in such a way that the local needs of people are met, apart from achieving the biodiversity conservation goal. Third, the planning of protected areas must be integrated with surrounding human uses. Also, protected areas must be planned as a system that addresses both, national and international objectives and must be managed individually in collaboration with the local people. Although some authors, such as Bruner *et al.* (2001) of Conservation International, as well as WWF (WWF 2001), recommend providing more funding to the existing mechanisms for strict protection of biodiversity, there is increasing scepticism among conservation professionals about this approach and about the displacement of local inhabitants in order to establish 'fortress' conservation areas. Community-based conservation has become more popular as an effective means for integrating conservation with sustainable development (Getz *et al.* 1999; ICPL 2001).

Linking protected areas and the land outside

Integrated conservation and development projects (ICDPs) are those in which the defining factor is some linkage between what happens inside a protected area and outside, in adjacent areas (Wells *et al.* 1992). UNESCO's Man and the Biosphere Programme first introduced the notion of integrating conservation inside protected areas

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[24] United Nations Educational, Scientific and Cultural Organisation.

and development outside (Batisse 1982). Such an approach provides a form of compromise in which international conservation agencies supply aid in return for local stewardship of biodiversity (Sinclair *et al.* 2000). However, currently there are problems in the implementation of such programmes. A study undertaken by the Indian Institute of Science showed that the programme in India was ineffective because the objectives were not clearly understood by most government institutions involved in management of biosphere reserves. As a result, the funds were utilised for strict protection and the community-based approach to conservation was ignored (Bhagwat & Gadgil 2000). In most developing countries, it is likely that the State institutions will confront similar problems since forest departments are mainly administrative organisations with limited capacity, resources and rapport with the local people for supporting multiple objectives related to conservation and development (Kramer & Sharma 1997). Nevertheless, linking conservation and development within and outside protected areas seems to be the most balanced approach for biodiversity conservation in the future. Active involvement of institutions other than the State apparatus, for instance local non-governmental organisations that have good rapport with the local inhabitants, is important for making ICDPs more effective at the grassroots. At the same time, long-term commitment for economic assistance from the international community is necessary (Sinclair *et al.* 2000). The success of ICDPs will depend on how harmoniously local and national government, non-governmental organisations and international aid agencies are able to co-operate.

In conclusion, protected areas and community-managed areas are not alternatives; rather, they are complementary and neither is self-sustaining (*e.g.* Sinclair *et al.* 2000). A strategy that combines the two approaches, where they are appropriate, is essential. This study has contributed to the current conservation debate by providing empirical evidence for the complementarity between the two approaches – it has discussed the situations where formal protection is preferable and illustrated those where community-based approaches work better for biodiversity conservation (Bhagwat *et al.* 2001 – see *APPENDIX 7*).

7. LITERATURE CITED

Achar K.P., Bhagwat S., Bhat D., Bhat H., Bhat K.G., Bhat I.P., Bhat P., Bhatta G.K., Chandran M.D.S., Gokhale Y., Gour-Broome V., Daniels A.E.D., Deviprasad K.N., Hebbar V.A., James M.R., Jayadeva G.S., Krishna K., Nirmalkumar J.I., Madhyastha N.A., Meenakshi K., Menon P., Murugan K., Palat R., Pandit P., Patil R.P., Poojari C.K., Pramod P., Ramachandran V.S., Rao K.S.M., Sivan V.V., Srinidhi S., Thomas W., Utkarsh G. & Velankar V. (2000) Tree communities and human influence in the Western Ghats, India. *Journal of Indian Institute of Science*, 80, 553-569.

Adams W.M. (1998) Conservation and Development. In: *Conservation Science and Action* (ed. Sutherland WJ), pp. 286-315. Blackwell Science Ltd., Oxford.

Akbar Sha A. (1987) *Working plan for Kodagu forest division, Karnataka.* Karnataka State Forest Department, Working Plan Division, Mysore.

Alcorn J.B. (1996) Forest use and ownership: patterns, issues and recommendations. In: *Forest Patches in Tropical Landscapes* (eds. Schelhas J & Greenberg R), pp. 233-257. Island Press, Washington DC.

Ali S. (1996) *The book of Indian birds.* 12th (Salim Ali centenary) edn. Oxford University Press, Mumbai.

Ambuel B. & Temple S.A. (1983) Area dependent changes in the bird communities and vegetation of southern Wisconsin forests. *Ecology*, 64, 1057-1068.

Ammirati S.R., Ammirati J.F. & Bledsoe C. (1987) Spatial and temporal distribution of ectomycorrhizal fungi in a Dougls-Fir plantation. In: *Mycorrhizae in the Next Decade - Proceedings of the 7th North American Conference on Mycorrhizae (NACOM)* (eds. Sylvia DM, Hung LL & Graham JH). NACOM, Gainesville, Florida, USA.

Andrén H. (1994) Effects of habitat fragmentation on birds and mammals in landscapes with different proportions of suitable habitat: A review. *Oikos*, 71, 355-366.

Arnolds E. (1989) A preliminary red data list of macrofungi in the Netherlands. *Persoonia*, 14, 121-130.

Arnolds E. (1992) The analysis and classification of fungal communities with special reference to macrofungi. In: *Fungi in Vegetation Science: Handbook of Vegetation Science 19(1)* (ed. Winterhoff W). Kluwer Academic Publishers, Dordrecht, The Netherlands.

Arrehnius O. (1921) Species and Area. *Journal of Ecology*, 9, 95-99.

Askins R.A. (1995) Hostile landscapes and the decline of migratory songbirds. *Science*, 267, 1956-1957.

Baker A.J. & Moeed A. (1987) Rapid genetic differentiation and founder effect in colonising populations of common mynas (*Acridotheres tristis*). *Evolution*, 41, 525-538.

Balmford A., Lyon A.J.E. & Lang R.M. (2000) Testing higher-taxon approach to conservation planning in a megadiverse group: the macrofungi. *Biological Conservation*, 93, 209-217.

Bar Association Virajpet (1998) *Souvenir - Centenary Celebrations of Virajpet Courts (1868 - 1998).* Bar Association, Virajpet, Kodagu.

Barry R.G. & Chorley R.J. (1987) *Atmosphere, weather and climate.* 5th edn. Routledge, London.

Baskent E. & Jordan G. (1995) Characterising spatial structure of forest landscapes. *Canadian Journal of Forest Research*, 25, 1830-1849.

Batisse M. (1982) The biosphere reserve: a tool for environmental conservation and management. *Environmental Conservation*, 9, 101-111.

Bawa K.S. & Ashton P.S. (1991) Conservation of rare trees in tropical rainforest: a genetic perspective. In: *Genetics and Conservation of Rare Plants* (eds. Falk DA & Holsinger KE), pp. 62-71. Oxford University Press, Oxford.

Berkes F. & Folke C. (1998) *Linking social and ecological systems*. Cambridge University Press, Cambridge.

Betts M. (2000) In Search of Ecological Relevancy: A Review of Landscape Fragmentation Metrics and Their Application for the Fundy Model Forest. In, p. 38. Greater Fundy Ecosystem Research Group (GFERG), Cooperative Fish and Wildlife Research Unit, University of New Brunswick, Fredericton, NB, Canada.

Bhagwat S. (1998) Sacred Groves and Protected Areas: Implications for Nature Reserve Design in the Western Ghats of India (MSc Dissertation). In: *Oxford Forestry Institute, Department of Plant Sciences*, p. 73. University of Oxford, Oxford.

Bhagwat S., Brown N.D., Evans T., Jennings S.B. & Savill P.S. (2001) Parks and factors in their success (Letters to Science). *Science*, 293, 1045-1047.

Bhagwat S. & Gadgil M. (2000) Biosphere Reserves of India. In: *Setting Biodiversity Conservation Priorities for India* (eds. Singh S, Sastry ARK, Mehta R & Uppal V), pp. 583-600. World Wide Fund for Nature - India, New Delhi.

Bhatt S. & Kothari A. (1997) Protected Areas in India: Proposal for an Expanded System of Categories. In: *Collaboration for Conservation*. Unpublished draft.

Bibby C., Jones M. & Marsden S. (1998) *Bird Surveys*. Royal Society for the Protection of Birds, Sandy, Bedfordshire, UK.

Bierregaard R.O., Laurance W.F., Sites J.W., Lynam A.J., Didham R.K., Andersen M., Gascon C., Tocher M.D., Smith A.P., Viana V.M., Lovejoy T.E., Sieving C.E., Kramer E.A., Restrepo C. & Moritz C. (1997) Key priorities for the study of fragmented tropical ecosystem (Chapter 33). In: *Tropical Forest Remnants: Ecology, Management and Conservation of Fragmented Communities* (eds. Laurance WF & Bierregaard RO), pp. 515-525. The University of Chicago Press, Chicago.

Bierregaard R.O., Lovejoy T.E., Kapos V., dos Santos A.A. & Hutchings R.W. (1992) The biological dynamics of tropical rain forest fragments. *Bioscience*, 42, 859-866.

Bierregaard R.O. & Stouffer P.C. (1997) Understorey birds and dynamic habitat mosaics in Amazonian rainforests (Chapter 10). In: *Tropical Forest Remnants: Ecology, Management and Conservation of Fragmented Communities* (eds. Laurance WF & Bierregaard RO), pp. 138-155. The University of Chicago Press, Chicago.

Biggs D., deVille B. & Suen E. (1991) A method of choosing multiway partitions for classification and decision trees. *Journal of Applied Statistics*, 18, 49-62.

Blackburn T.M., Harvey P.H. & Pagel M.D. (1990) Species number, population density and body size relationships in natural communities. *Journal of Animal Ecology*, 59, 335-345.

Bland J.M. & Altman D.G. (1995) Multiple significance tests: the Bonferroni method. *British Medical Journal*, 310, 170.

Bonn E. (2000) An economic framework to land extensification (The Dynamics of Land Use - Chapter 5). In: *Mountain Biodiversity, Land Use Dynamics, and*

Traditional Knowledge (eds. Ramakrishnan PS, Chandrashekara UM, Elouard C, Guilmoto CZ, Maikhuri RK, Rao KS, Sankar S & Saxena KG), pp. 79-88. Man and the Biosphere Programme, Oxford and IBH Publishing Co. Pvt. Ltd., New Delhi.

Bonnet E. (2001) Software for Mantel's test. URL http://www.geocities.com/eb_ce/mantel.html (Last accessed: March 2002).

Borges R.M. (2000) The anatomy of fragmentation. *Journal of the Indian Institute of Science*, 80, 601-608.

Bowman J., Forbes G. & Dilworth T. (2001) Landscape context and small-mammal abundance in a managed forest. *Forest Ecology and Management*, 140, 249-255.

Brandis D. (1897) *Indian Forestry*. Oriental University Institute, Woking, UK.

Brandon K. (1997) Policy and practical considerations in land use strategies for biodiversity conservation. In: *Last Stand: Protected Areas and the Defense of Tropical Biodiversity* (eds. Kramer R, van Schaik C & Johnson J), pp. 90-114. Oxford University Press, New York.

Breiman L., Friedman J.H., Olshen R.A. & Stone C.J. (1984) *Classification and regression trees*. Wadsworth, Belmont.

Bridgewater S. (1992) The Sacred Groves of India: A Systematic Analysis (MSc Dissertation). In: *Oxford Forestry Institute, Department of Plant Sciences*. University of Oxford, Oxford.

Brockelman W.Y. (1998) Study of tropical forest canopy height and cover using a point intercept method. In: *Forest Biodiversity Research, Monitoring and Modeling* (eds. Dallmeier F & Comiskey JA), pp. 521-531. Parthenon Publishing Group, Carnforth.

Brown J.H. & Kondric-Brown A. (1977) Turnover rates in insular biogeography: effects of immigration on extinction. *Ecology*, 58, 445-449.

Brown K.S.J. & Hutchings R.W. (1997) Disturbance, Fragmentation and the Dynamics of Diversity in Amazonian Forest Butterflies (Chapter 7). In: *Tropical Forest Remnants: Ecology, Management and Conservation of Fragmented Communities* (eds. Laurance WF & Bierregaard RO), pp. 91-110. The University of Chicago Press, Chicago.

Brown N.D. (1993) The implications of climate and gap microclimate for seedling growth in a Bornean lowland rainforest. *Journal of Tropical Ecology*, 9, 153-168.

Brown N.D. (2000) Maintaining biodiversity (Chapter 15). In: *Applied Geography: Principles and Practice* (ed. Pacione M), pp. 222-235. Routledge, London and New York.

Brown N.D., Jennings S.B., Wheeler P. & Nabe-Nielsen J. (2000) An improved method for rapid assessment of forest understorey light environments. *Journal of Applied Ecology*, 37, 1044-1053.

Bruner A.G., Gullison R.E., Rice R.E. & da Fonseca G.A.B. (2001) Effectiveness of parks in protecting tropical biodiversity. *Science*, 291, 125-128.

Brussard P.F., Murphy D.D. & Noss R.F. (1992) Strategy and tactics for conserving biological diversity in the United States. *Conservation Biology*, 6, 157-159.

BSRSI Michigan State University (2001) Landsat.org. URL Basic Science and Remote Sensing Initiative http://bsrsi.msu.edu (Last accessed: March 2002).

Burkey T.G. (1989) Extinction in nature reserves: The effect of fragmentation and the importance of migration between reserve fragments. *Oikos*, 55, 75-81.

CBD (1992) Convention on Biological Diversity. URL http://www.biodiv.org (Last accessed: March 2002).

CBD (2001) Convention on Biological Diversity. URL http://www.biodiv.org/meetings/cop-06.asp# (Last accessed: March 2002).

Chambers R., Saxena N.C. & Shah T. (1989) *To the hands of the poor*. Oxford & IBH, New Delhi.

Chandrakanth M.G., Gilless J.K., Gowramma V. & Nagaraja M.G. (1990) Temple forests in India's forest development. *Agroforestry systems*, 11, 199-211.

Chandrakanth M.G. & Nagaraja M.G. (1997) Existence value of Kodagu sacred groves: Implications for policy. In: *The Challenge of the Balance: Environmental Economics in India*, pp. 217-224. Centre for Science and Environment, New Delhi.

Chandran M.D.S. (1993) Vegetation Changes in the Evergreen Forest Tract of Uttara Kannada District of Karnataka State (PhD Thesis). In: *Department of Botany*. Karnataka University, Dharwad, India.

Chandran M.D.S. (1997) On the ecological history of the Western Ghats. *Current Science*, 73, 146-155.

Chandran M.D.S., Gadgil M. & Hughes J.D. (1998) Sacred groves of the Western Ghats of India. In: *Conserving the Sacred for Biodiversity Management* (eds. Ramakrishnan PS, Saxena KG & Chandrashekara UM), pp. 211-231. Oxford and IBH publishing co. pvt. ltd., New Delhi, Calcutta.

Chandrashekara U.M. & Sankar S. (1998) Structure and functions of sacred groves: case studies in Kerala. In: *Conserving the Sacred for Biodiversity Management* (eds. Ramakrishnan PS, Saxena KG & Chandrashekara UM), pp. 323-335. Oxford and IBH Publishing Co. Pvt. Ltd., New Delhi.

Charlesworth D. & Charlesworth B. (1987) Inbreeding depression and its evolutionary consequences. *Annual Review of Ecology and Systematics*, 18, 237-268.

Clark D.B. (1990) The role of disturbance in the regeneration of neotropical moist forests. In: *Reproductive Ecology of Tropical Forest Plants* (eds. Bawa KS & Hadley M). United Nations' Edicational Scientific and Cultural Organisation, Paris.

Colding J. & Folke C. (1997) The relations among threatened species, their protection, and taboos. *Conservation Ecology [online]*, 1, http://www.consecol.org/vol1/iss1/art6 (Last accessed: March 2002).

Commander S. (1986) Managing Indian forests: a case for the reform of property rights. *Developmental Policy Review*, 4, 325-344.

Condit R. (1994) *A field guide to large forest censuses in the tropics: The BCI 50 ha plot and other large-scale studies*. Centre for Tropical Forest Science, Panama.

Condit R., Hubbell S.P., Lafrankie J.V., Sukumar R., Manokaran N., Foster R.B. & Ashton P.S. (1996) Species-area and species-individual relationships for tropical trees: A comparison of three 50-ha plots. *Journal of Ecology*, 84, 549-562.

Condit R. & Markham A. (1998) Ecological implications of changes in drought patterns: shifts in forest composition in Panama. *Climatic Change*, 39, 413-427.

Condit R., Sukumar R., Hubbell S.P. & Foster R.B. (1998) Predicting population trends from size distributions: a direct test in a tropical tree community. *The American Naturalist*, 152, 495-509.

Condit R., Watts K., Bohlman S.A., Perez R., Foster R.B. & Hubbell S.P. (2000) Quantifying the deciduousness of tropical forest canopies under varying climates. *Journal of Vegetation Science*, 11, 649-658.

Connell J.H. (1978) Diversity in tropical rain forests and coral reefs. *Science*, 199, 1302-1310.

Connor E.F. & McCoy E.D. (1979) The statistics and biology of the species-area relationship. *American Naturalist*, 113, 791-833.

Corlett R.T. (1998) Frugivory and seed dispersal by vertebrates in the Orietal (Indomalayan) region. *Biological Reviews of the Cambridge Philosophical Society*, 73, 413-448.

Corlett R.T. & Turner I.M. (1997) *Long-term survival in tropical forest remnants in Singapore and Hong Kong*. The University of Chicago Press, Chicago.

Council for Scientific and Industrial Research (1989) *The Useful Plants of India*. Council for Industrial and Scientific Research, New Delhi, India.

Csuti B., Polasky S., Williams P.H., Pressey R.L., Camm J.D., Kershaw M., Kiester A.R., Downs B., Hamilton R., Huso M. & Sahr K. (1997) A comparison of reserve selection algorithms using data on terrestrial vertebrates in Oregon. *Biological Conservation*, 80, 83-97.

Dahlberg A. (2001) Community ecology of ectomycorrhizal fungi: an advancing interdisciplinary field. *New Phytologist*, 150, 555-562.

Daniels R.J.R., Gadgil M. & Joshi N.V. (1995) Impact of human extraction on tropical humid forests in the Western Ghats in Uttara Kannada, South India. *Journal of Applied Ecology*, 32, 866-874.

Dasmann R.F. (1991) The importance of cultural and biological diversity. In: *Biodiversity: Culture, Conservation and Ecodevelopment* (eds. Oldfield ML & Alcorn JB), pp. 7-15. Westview Press, Boulder, CO, USA.

Davidson C. (1998) Issues in measuring landscape fragmentation. *Wildlife Society Bulletin*, 26, 32-37.

Dawson D. (1994) Are habitat corridors conduits for animals and plants in a fragmented landscape? A review of the scientific evidence. In. English Nature Research Reports No. 94.

Déath G. & Fabricius K.E. (2000) Classification and regression trees: a powerful yet simple technique for ecological data analysis. *Ecology*, 81, 3178-3192.

Diamond J.M. (1975) The island dilemma: lessons of modern biogeographic studies for the design of nature reserves. *Biological Conservation*, 7, 129-146.

Diamond J.M. (1980) Patchy distributions of tropical birds. In: *Conservation Biology: An Evolutionary-Ecological Perspective* (eds. Soulé ME & Wilcox BA), pp. 57-74. Sinauer Associates, Sunderland, MA.

Diamond J.M. (1986) Design of a nature reserve system for Indonesian New Guinea. In: *Conservation Biology: The Science of Scarcity and Diversity* (ed. Soulé ME), pp. 485-503. Sinauer, Sunderland, Massachusetts.

Didham R.K. (1997) The influence of edge effects and forest fragmentation on leaf litter invertebrates in Central Amazonia (Chapter 5). In: *Tropical Forest Remnants: Ecology, Management and Conservation of Fragmented Communities* (eds. Laurance WF & Bierregaard RO), pp. 55-70. The University of Chicago Press, Chicago.

Dighton J. & Mason P. (1985) Mycorrhizal dynamics during forest tree development. In: *Developmental Biology of Higher Fungi* (eds. Moore D, Casselton LA, Wood DA & Frankland JC), pp. 117-139. Cambridge University Press, Cambridge, UK.

Doak D.F., Marino P.C. & Kareiva P.M. (1992) Spatial scale mediates the influence of habitat fragmentation on dispersal success: Implications for conservation. *Theoretical Population Biology*, 41, 315-336.

Doak D.F. & Mills L.S. (1994) A useful role for theory in conservation. *Ecology*, 75, 615-626.

Durning A.T. (1993) Saving The Forests: What Will It Take? In. Worldwatch Institute, Washington DC.

Elouard C. (2000a) The role of coffee plantations in biodiverisity conservation (Coffee and Biodiversity Conservation - Chapter 6). In: *Mountain Biodiversity, Land Use Dynamics, and Traditional Knowledge* (eds. Ramakrishnan PS, Chandrashekara UM, Elouard C, Guilmoto CZ, Maikhuri RK, Rao KS, Sankar S & Saxena KG), pp. 120-144. Man and the Biosphere Programme, Oxford and IBH Publishing Co. Pvt. Ltd., New Delhi.

Elouard C. (2000b) Vegetation features in relation to biogeography (Landscape and Society - Chapter 3). In: *Mountain Biodiversity, Land Use Dynamics, and Traditional Knowledge* (eds. Ramakrishnan PS, Chandrashekara UM, Elouard C, Guilmoto CZ, Maikhuri RK, Rao KS, Sankar S & Saxena KG), pp. 25-42. Man and the Biosphere Programme, Oxford and IBH Publishing Co. Pvt. Ltd., New Delhi.

Erdilek A. & Wolf M.A. (1997) Technology origins of foreign-owned firms in Ohio. *Technovation*, 17, 63-72.

Fahrig L. (1998) When does fragmentation of breeding habitat affect population survival? *Ecological Modelling*, 105, 273-292.

Fahrig L. & Merriam G. (1994) Conservation of fragmented populations. *Conservation Biology*, 8, 50-59.

Fahrig L. & Paloheimo J. (1988) Effects of spatial arrangement of habitat patches on local population size. *Ecology*, 69, 468-475.

FAO (2001) State of the World's Forests. URL Food and Agricultural Organisation of the United Nations http://www.fao.org/docrep/003/y0900e/y0900e00.htm (Last accessed: March 2002).

Feinsinger P. (1976) Organisation of a tropical guild of nectarivorous birds. *Ecological Monographs*, 46, 257-91.

Fogel R. (1976) Ecological studies of hypogeal fungi: II Sporocarp phenology in a western Oregon Douglas Fir stand. *Canadian Journal of Botany*, 54, 1152-1162.

Fogel R. (1980) Mycorrhizae and nutrient cycling in natural forest ecosystems. *New Phytologist*, 86, 199-212.

Forest Survey of India (1995) Report on the Inventory of Forest Resources of Kodagu District (Karnataka). In. Forest Survey of India (Southern zone), Bangalore.

Forest Survey of India (1999) State of Forest Report. URL Forest Survey of India http://envfor.nic.in/fsi/sfr99/sfr.html (Last accessed: March 2002).

Forman R.T.T. (1995) *Land Mosaics: The Ecology of Landscapes and Regions*. Cambridge University Press, Cambridge, UK.

Forman R.T.T. & Gordon M. (1986) *Landscape Ecology*. John Wiley and Sons, New York.

Freemark K.E. & Merriam H.G. (1986) Importance of area and habitat heterogeneity to bird assemblages in temperate forest fragments. *Biological Conservation*, 36, 115-141.

Freidenberg L.K. (1998) Physical effects of habitat fragmentation (Chapter 4). In: *Conservation Biology for the Coming Decade* (eds. Fielder PL & Kareiva PM), pp. 66-79. Chapman and Hall, New York.

FRLHT (1999) *Priority List of Medicinal Plants in South India*. Foundation for Revitalisation of Local Health Traditions (Unpublished Version), Bangalore.

FRLHT (2000a) Kodagu maps. In. Foundation for Revitalisation of Local Health Traditions, Bangalore.

FRLHT (2000b) Medicinal plants under threat! URL Foundation for Revitalisation of Local Health Traditions http://www.frlht-india.org/Conservation.htm (Last accessed: March 2002).

Gadgil M. (1992) Conserving biodiversity as if people matter: A case study from India. *Ambio*, 21, 266-270.

Gadgil M., Berkes F. & Folke C. (1993) Indigenous knowledge for biodiversity conservation. *Ambio*, 22, 151-156.

Gadgil M. & Guha R. (1992) *This Fissured Land.* Oxford University Press, New Delhi.

Gadgil M. & Guha R. (1995) *Ecology and Equity.* Penguin Books, New Delhi.

Gadgil M. & Rao P.R.S. (1994) A system of positive incentives to conserve biodiversity. *Economic and Political Weekly, India.*

Gadgil M. & Vartak V.D. (1976) The sacred groves of the Western Ghats in India. *Economic Botany*, 30, 152-160.

Ganesan B. (1993) Extraction of non-timber forest products including fodder and fuelwood in Mudumalai, India. *Economic Botany*, 47, 268-274.

Gardner R.H., O'Neill R.V., Turner M.G. & Dale V.H. (1989) Quantifying scale-dependent effects of animal movements with simple percolation models. *Landscape Ecology*, 3, 217-227.

Garrison G.A. (1949) Uses and modifications for the 'Moosehorn' crown closure estimator. *Journal of Forestry*, 47, 733-735.

Gaston K.J. & Blackburn T.M. (1995) Birds, body size and the threat of extinction. *Philosophical Transactions of the Royal Society, London: Biological Sciences*, 347, 205-212.

Gaston K.J. & Lawton J.H. (1988) Pattrens in body size, population dynamics and regional distribution of bracken herbivores. *American Naturalist*, 132, 662-680.

Geiger R. (1965) *The Climate Near The Ground.* Harvard University Press, Cambridge, MA.

Gentry A.H. (1990) *Four Neotropical Rainforests.* Yale University Press, Branford.

Getz W.M., Fortmann L., Cumming D., du Toit J., Hilty J., Martin R., Murphree M., Owen-Smith N., Starfield A.M. & Westphal M.I. (1999) Sustaining natural and human capital: Villages and scientists. *Science*, 283, 1855-1856.

Ghate U., Bhagwat S., Gokhale Y., Gour-Broome V. & Barve V. (1997) Assessing the tropical forest plant diversity: A case study from the Western Ghats, India. *International Journal of Ecology and Environmental Sciences*, 23, 419-444.

Gilbert L.E. (1980) Food web organisation and conservation of neotropical diversity. In: *Conservation Biology: An Evolutionary-Ecological Perspective* (eds. Soulé ME & Wilcox BA), pp. 11-33. Sinauer Associates, Sunderland, MA.

Gillepsie T.W. (2000) Rarity and conservation of forest birds in the tropical dry forest region of Central America. *Biological Conservation*, 96, 161-168.

Glassy J.M. & Running S.W. (1994) Validating diurnal climatology logic of the MT-CLIM model across a climatic gradient in Oregon. *Ecological Applications*, 4, 248-257.

Glastra R. (1999) *Cut and Run: Illegal Logging and Timber Trade in the Tropics.* Friends of Earth International; International Development Research Centre, Ottawa, Canada.

Gobbi J.A. (2000) Is biodiversity-friendly coffee financially viable?: An analysis of five different coffee production systems in western El Salvador. *Ecological Economics*, 33, 267-281.

Godley S.H., Fiedler E.M. & Funk R.R. (1998) Consumer satisfaction of parents and their children with child/adolescent mental health services. *Evaluation and Program Planning*, 21, 31-45.

Gotelli N.J. (1998) *A Primer of Ecology*. Sinauer Associates, Sunderland, MA.

Gotelli N.J. & Colwell R.K. (2001) Quantifying biodiversity: Procedures and pitfalls in the measurement and comparison of species richness. *Ecology Letters*, 4, 379-391.

Gradwohl J. & Greenberg R. (1988) *Saving the Tropical Forest*. EARTHSCAN, London.

Greenberg R. (1981) The abundance and seasonality of forest canopy birds on Barro Colorado Island, Panama. *Biotropica*, 13, 241-251.

Greenberg R. (1996) Managed forest patches and the diveristy of birds in southern Mexico. In: *Forest Patches in Tropical Landscapes* (eds. Schelhas J & Greenberg R), pp. 59-90. Island Press, Washington DC.

Greenberg R., Bichier P. & Sterling J. (1997) Bird populations and planted shade coffee plantations in eastern Chiapas, Mexico. *Biotropica*, 29, 501-514.

Greenwood J.J.D., Gregory R.D., Harris S., Morris P.A. & Yalden D.W. (1996) Relations between abundance, body size and species number in British birds and mammals. *Philosophical Transactions of the Royal Society, London: Biological Sciences*, 351, 265-278.

Grimmett R., Inskipp C. & Inskipp T. (1998) *Birds of the Indian Subcontinent*. Christopher Helm A & C Black, London.

Grogan P., Baar J. & Bruns T.D. (2000) Below-ground ectomycorrhizal community structure in a recently burned bishop pine forest. *Journal of Ecology*, 88, 1051-1062.

GTCO CalComp Inc. (2000) DrawingBoard III. URL http://www.gtcocalcomp.com/productdrawingboard3large.htm (Last accessed: March 2002).

Guha R. (1983) Forestry in British and post-British India: A historical analysis. *Economic and Political Weekly, India.*, 18, 1882-1896; 1940-1946.

Gustafson E.J. (1998) Quantifying landscape spatial pattern: What is the state of the art? *Ecosystems*, 1, 143-156.

Gustafson E.J. & Parker G.R. (1992) Relationship between landcover proportion and indices of landscape spatial pattern. *Landscape Ecology*, 7, 101-110.

Haller G. (1910) Coorg Land and Revenue Regulation 1899. In. Deputy Commissioner's Office, Madikeri.

Hammond D.S., Gourlet-Fleury S., van der Hout P., ter Steege H. & Brown V.K. (1996) A compilation of known Guianan timber trees and the significance of their dispersal mode, seed size and taxonomic affinity to tropical rain forest management. *Forest Ecology and Management*, 83, 99-116.

Hanski I. (1996) Metapopulation ecology. In: *Population Dynamics in Ecological Space and Time* (eds. Rhodes OEJ, Chesser RK & Smith MH), pp. 13-43. Chicago University Press, Chicago.

Hanski I.A. (1997) Metapopulation dynamics: From concepts and observations to predictive models. In: *Metapopulation Biology: Ecology, Genetics and Evolution* (eds. Hanski IA & Gilpin ME), pp. 69-91. Academic Press, San Diego, California.

Hanski I.A. & Gilpin M.E. (1991) Metapopulation dynamics: Brief history and conceptual domain. *Biological Journal of the Linnaen Society*, 42, 3-16.

Hanski I.A. & Gilpin M.E. (1997) *Metapopulation Biology: Ecology, Genetics and Evolution*. Academic Press, San Diego, California.

Hardin G. (1968) The tragedy of the commons. *Science*, 162, 1243-1248.

Hargis C., Bissonette J. & David J. (1998) The behaviour of landscape metrics commonly used in the study of habitat fragmentation. *Landscape Ecology*, 13, 167-186.

Harley J.L. & Smith S.E. (1983) *Mycorrhizal Symbiosis*. Academic Press, London.

Harrington G.N., Irvine A.K., Crome F.H.J. & Moore L.A. (1997) Regeneration of large-seeded trees in Australian rainforest fragments: A study of higher-order interactions (Chapter 19). In: *Tropical Forest Remnants: Ecology, Management and Conservation of Fragmented Communities* (eds. Laurance WF & Bierregaard RO), pp. 292-303. The University of Chicago Press, Chicago.

Harrison S.P. (1994) Metapopulations and conservation. In: *Large-scale Ecology and Conservation Biology* (eds. Edwards PJ, May RM & Webb NR), pp. 111-128. Blackwell Scientific Publishers, Oxford.

Harrison S.P., Stahl A. & Doak D.F. (1993) Spatial models and spotted owls: Exploring some biological issues behind recent events. *Conservation Biology*, 7, 950-953.

Harrison S.P. & Taylor A.D. (1997) Empirical evidence for metapopulation dynamics. In: *Metapopulation Biology: Ecology, Genetics and Evolution* (eds. Hanski IA & Gilpin ME), pp. 27-42. Academic Press, San Diego, California.

Hartshorn G.S. (1995) Ecological basis for sustainable development in tropical forests. *Annual Review of Ecology and Systematics*, 26, 155-175.

Hawksworth D.L. (1997) The fascination of fungi: Exploring fungal diversity. *Mycologist*, 11, 18-22.

Hawthorne W.D. (1996) Holes and the sums of parts in Ghanaian forest: Regeneration, scale and sustainable use. *Proceedings of the Royal Society of Edinburgh*, 104B, 75-176.

Hays S.P. (1987) *Beauty, Health and Permanence: Environmental Politics in the United States, 1955-1985*. Cambridge University Press, New York.

Heck K.L.J., van Belle G. & Simberloff D. (1975) Explicit calculation of the rarefaction diversity measurement and the determination of sufficient sample size. *Ecology*, 56, 1459-1461.

Henderson H.V. & Velleman P.F. (1981) Building multiple regression models interactively. *Biometrics*, 37, 391-411.

Hering T.F. (1966) The terricolous higher fungi of four lake district woodlands. *Transactions of the British Mycological Society*, 49, 369-383.

Hill J.L. & Curran P.J. (2001) Species composition in fragmented forests: Conservation implications for changing forest area. *Applied Geography*, 21, 157-174.

Hobbs R.J. & Saunders D.A. (1993) *Reintegrating Fragmented Landscapes: Towards Sustainable Production and Nature Conservation*. Springer-Verlag, New York, USA.

Honnay O., Hermy M. & Coppin P. (1999) Effects of area, age and diversity of forest patches in Belgium on plant species richness, and implications for conservation and reforestation. *Biological Conservation*, 87, 73-84.

Hoopes M.F. & Harrison S.P. (1998) Metapopulation, source-sink and disturbance dynamics (Chapter 7). In: *Conservation Science and Action* (ed. Sutherland WJ), pp. 135-151. Blackwell Science, Oxford.

Huba G.J. (2000) CHAID. URL http://www.themeasurementgroup.com/Definitions/chaid.htm (Last accessed: March 2002).

Hubbell S.P. (2001) *The Unified Neutral Theory of Biodiversity and Biogeography.* Princeton University Press, Princeton, NJ.

Hubbell S.P. & Foster R.B. (1986) Commonness and rarity in a neotropical forest: Implications for tropical tree conservation. In: *Conservation Biology: The Science of Scarcity and Diversity* (ed. Soulé ME), pp. 205-231. Sinauer Associates, Sunderland, MA.

Hughes J.D. & Chandran M.D.S. (1998) Sacred groves around the earth: An overview. In: *Conserving the Sacred for Biodiversity Management* (eds. Ramakrishnan PS, Saxena KG & Chandrashekara UM), pp. 69-86. Oxford and IBH Publishing Co., New Delhi.

Hurlbert S.H. (1971) The non-concept of species diversity: A critique and alternative parameters. *Ecology*, 52, 577-586.

Hussey T., Kott B., Zalunardo D. & McNair R. (1989) Fragmentation Model Mount Hood National Forest. In. U. S. Department of Agriculture, Forest Service, Mount Hood National Forest, Gresham, OR, USA.

Hutto R.L., Pletschet S.M. & Hendricks P. (1986) A fixed-radius point count method for non-breeding and breeding season use. *Auk*, 103, 593-602.

ICPL (2001) Protected Area Management for the Twenty-First Century. URL International Centre for Protected Landscapes http://www.protected-landscapes.org/protarea.html (Last accessed: March 2002).

Inamdar A., de Jode H., Lindsay K. & Cobb S. (1999) Capitalising on nature: Protected area management. *Science*, 283, 1856-1857.

IUCN (1994) Guidelines for Protected Area Management Categories. In. IUCN The World Conservation Union, Gland, Switzerland.

IUCN (2000) 2000 IUCN red list of threatened species. URL IUCN The World Conservation Union http://www.redlist.org/ (Last accessed: March 2002).

IUCN/WCMC (1992) Protected Areas of the World: A Review of National Systems (Volume 1: Indomalaya, Oceania, Australia and Antarctic). In. IUCN The World Conservation Union; World Conservation Monitoring Centre (WCMC), Gland, Switzerland; Cambridge, UK.

James F.C. & McCulloch C.E. (1990) Multivariate analysis in ecology and systematics: Panacea or Pandora's box? *Annual Review of Ecology and Systematics*, 21, 129-166.

Jansen A.E. & Denie H.W. (1988) Relations between mycorrhizas and fruit bodies of mycorrhizal fungi in Douglas Fir plantations in the Netherlands. *Acta Botanica Neerlandica*, 37, 243-249.

Janzen D.H. (1983) No park is an island: Increase in interference from outside as park size decreases. *Oikos*, 41, 402-410.

Jennings S.B., Brown N.D., Whitmore T.C., Silva J.N.M., Lopes J.d.C.A. & Baima A.M.V. (2000) To conserve rainforest, we have to help local people live sustainably. *Nature*, 405, 507.

Jepson P., Jarvie J.K., MacKinnon K. & Monk K.A. (2001) The end of Indonesia's lowland forests? *Science*, 292, 859-861.

Johns A.D. (1991) Responses of Amazonian rainforest birds to habitat modification. *Journal of Tropical Ecology*, 7, 417-437.

Jonsson L., Dahlberg A., Nilsson M.-C., Zackrisson O. & Kårén O. (1999) Ectomycorrhizal fungal communities in late-successional Swedish boreal forests, and their composition following wildfire. *Molecular Ecology*, 8, 205-215.

Jordan M. (1995) *The Encyclopaedia of Fungi of Britain and Europe.* David & Charles, UK.

Kahn J.R. & McDonald J.A. (1997) The role of economic factors in tropical deforestation (Chapter 2). In: *Tropical Forest Remnants: Ecology, Management and Conservation of Fragmented Communities* (eds. Laurance WF & Bierregaard RO), pp. 13-31. The University of Chicago Press, Chicago.

Kalam M.A. (1996) *Sacred groves in Kodagu district of Karnataka (South India): A socio-historical study.* Institute Francais de Pondichery, Pondichery, South India.

Kalam M.A. (2000) Devarakadus and encroachments. In: *Mountain Biodiversity, Land Use Dynamics, and Traditional Knowledge* (eds. Ramakrishnan PS, Chandrashekara UM, Elouard C, Guilmoto CZ, Maikhuri RK, Rao KS, Sankar S & Saxena KG), pp. 98-106. Man and the Biosphere Programme; Oxford and IBH Publishing Co. Pvt. Ltd., New Delhi; Calcutta.

Kalina J. (1988) Ecology and Behaviour of the Black and White Casqued Hornbill (*Buceros subcylindricus subquadratus*) in the Kibale Forest (PhD Thesis). In. Michigan University, MI, USA.

Kanowski P. & Boshier D.H. (1997) Conserving the genetic resources of trees *in situ*. In: *Plant Genetic Conservation* (eds. Maxted N, Ford-Lloyd BV & Hawkes JG). Chapman and Hall, New York.

Kapos V. (1989) Effects of isolation on the water status of forest patches in the Brazilian Amazon. *Journal of Tropical Ecology*, 5, 173-185.

Kapos V., Wandelli E., Camargo J.L. & Ganade G. (1997) Edge-related changes in environment and plant responses due to forest fragmentation in Central Amazonia (Chapter 3). In: *Tropical Forest Remnants: Ecology, Management and Conservation of Fragmented Communities* (eds. Laurance WF & Bierregaard RO), pp. 33-44. The University of Chicago Press, Chicago.

Karnataka State Forest Department (1985) Transfer of Devarakadu Lands in Kodagu District from Revenue Department to the Forest Department. In. Karnataka State Forest Department, Mysore.

Kass G. (1980) An exploratory technique for investigating large quantities of categorical data. *Applied Statistics*, 29, 119-127.

Kellman M., Tackaberry R. & Meave J. (1996) The consequences of prolonged fragmentation: Lessons from tropical gallery forests. In: *Forest Patches in Tropical Landscapes* (eds. Schelhas J & Greenberg R), pp. 37-58. Island Press, Washington DC.

Kenkel N.C. & Orloci L. (1986) Applying metric and non-metric multidimensional scaling to ecological studies: Some new results. *Ecology*, 67, 919-928.

Kerr J.T. (1997) Species richness, endemism and the choice of areas for conservation. *Conservation Biology*, 11, 1094-1100.

Keshavamoorthy K.R. & Yoganarasimhan S.N. (1989) *Flora of Coorg.* Vimsat Publishers, Bangalore.

Khare A., Sarin M., Saxena N.C., Palit S., Bathla S., Vania F. & Satyanarayana M. (2000) *Joint Forest Management: Policy, Practice and Prospects.* World Wide Fund for Nature-India; International Institute for Environment and Development, New Delhi; London.

Kindvall O. & Ahlen I. (1992) Geometrical factors and metapopulation dynamics of the bush cricket, *Metrioptera bicolor philippi* (Orthoptera: Tettigoniidae). *Conservation Biology*, 6, 520-529.

Klein B.C. (1989) The effects of forest fragmentation on dung and carrion beetle (Scarabaeinae) communities in Central Amazonia. *Ecology*, 70, 1715-1725.

Komonen A. (2001) Structure of insect communities inhabiting old-growth forest specialist bracket fungi. *Ecological Entomology*, 26, 63-75.

Kosambi D.D. (1962) *Myth and Reality: Studies in the Formation of Indian Culture.* Popular Press, Bombay (Mumbai), India.

Kothari A., Pande P., Singh S. & Variava D. (1989) Management of National Parks and Sanctuaries in India: A Status Report. In. Environmental Studies Division, Indian Institute of Public Administration, New Delhi.

Kothari A., Pathak N., Suryanarayanan J. & Vania F. (1997) *Community Involvement in Wildlife Conservation - South Asia Regional Review.* International Institution for Environment and Developement, London.

Kothari A., Singh N. & Suri S. (1996) *People and Protected Areas: Towards Participatory Conservation in India.* Sage Publications, New Delhi.

Kramer R. & van Schaik C. (1997) Preservation paradigms and tropical rain forests (Chapter 1). In: *Last Stand: Protected Areas and the Defence of Tropical Biodiversity* (eds. Kramer R, van Schaik C & Johnston J), pp. 3-14. Oxford University Press, New York.

Kramer R., van Schaik C. & Johnston J. (1997) *Last Stand: Protected Areas and the Defence of Tropical Biodiversity.* Oxford University Press, New York.

Kramer R.A. & Sharma N. (1997) Tropical forest biodiversity protection: Who pays and why. In: *Last Stand: Protected Areas and the Defense of Tropical Biodiversity* (eds. Kramer R, van Schaik C & Johnson J), pp. 162-186. Oxford University Press, New York.

Krummel J.R., Gardner R.H., Sugihara G., O'Neill R.V. & Coleman P.R. (1987) Landscape patterns in a disturbed environment. *Oikos*, 48, 321-324.

Kushalappa C.G. (2001) (*Personal Communication*) Devarakadus of Kodagu. In, Ponnampet, Kodagu.

Kushalappa C.G. & Bhagwat S. (2001) Sacred groves: Biodiversity, threats and conservation. In: *Forest Genetic Resources: Status, Threats, and Conservation Strategies* (eds. Shaanker UR, Ganeshaiah KN & Bawa KS), pp. 21-29. Oxford and IBH Co. Pvt. Ltd., New Delhi.

Kushalappa C.G., Bhagwat S. & Kushalappa K.A. (2001) Conservation and management of sacred groves of Kodagu, Karnataka, South India - A unique approach. In: *Tropical Ecosystems: Structure, Diversity and Human Welfare: Proceedings of the International Conference on Tropical Ecosystems* (eds. Ganeshaiah KN, Shaanker UR & Bawa KS), pp. 565-569. Oxford and IBH Co. Pvt. Ltd., New Delhi.

Kushalappa C.G. & Kushalappa K.A. (1996) Preliminary Report: Impact Assessment of Working in the Western Ghats Forests. In. College of Forestry, University of Agricultural Sciences - Bangalore, Ponnampet.

Lacy R.C. (1987) Loss of genetic diversity from managed populations: Interacting effects of drift, mutation, immigration, selection and population subdivision. *Conservation Biology*, 1, 143-158.

Lamberson R.H., McKelvey K.S., Noon B.R. & Voss C. (1992) A dynamic analysis of Northern Spotted Owl viability in a fragmented forest landscape. *Conservation Biology*, 6, 505-512.

Lapin M. & Barnes B.V. (1995) Using the landscape ecosystem approach to assess species and ecosystem diversity. *Conservation Biology*, 9, 1148-1158.

Larson B.A. & Bromley D. (1990) Property rights, externalities and resource degradation: Locating the tragedy. *Journal of Development Economics*, 33, 235-262.

Laurance W.F. (1991a) Ecological correlates of extinction proneness in Australian tropical rainforest mammals. *Conservation Biology*, 5, 79-89.

Laurance W.F. (1991b) Edge effects in tropical forest fragments: Application of a model for the design of nature reserves. *Biological Conservation*, 57, 205-219.

Laurance W.F. (1994) Rainforest fragmentation and the structure of small mammal communities in tropical Queensland. *Biological Conservation*, 69, 23-32.

Laurance W.F. (1997a) Hyper-disturbed parks: Edge effects and the ecology of isolated rainforest reserves in tropical Australia (Chapter 6). In: *Tropical Forest Remnants: Ecology, Management and Conservation of Fragmented Communities* (eds. Laurance WF & Bierregaard RO), pp. 71-83. The University of Chicago Press, Chicago.

Laurance W.F. (1997b) Responses of mammals to rainforest fragmentation in tropical Queensland: A review and synthesis. *Wildlife Research*, 24, 603-612.

Laurance W.F. & Bierregaard R.O. (1997) *Tropical Forest Remnants: Ecology, Management and Conservation of Fragmented Communities.* The University of Chicago Press, Chicago, IL.

Laurance W.F., Bierregaard R.O., Gascon C., Didham R.K., Smith A.P., Lynam A.J., Viana V.M., Lovejoy T.E., Sieving C.E., Sites J.W., Andersen M., Tocher M.D., Kramer E.A., Restrepo C. & Moritz C. (1997) Tropical forest fragmentation: A synthesis of a diverse and dynamic discipline (Chapter 32). In: *Tropical Forest Remnants: Ecology, Management and Conservation of Fragmented Communities* (eds. Laurance WF & Bierregaard RO), pp. 502-514. The University of Chicago Press, Chicago.

Laurance W.F., Ferreira L.V., Rankin-de Merona J.M. & Laurance S.G. (1998a) Rain forest fragmentation and the dynamics of Amazonian tree communities. *Ecology*, 79, 2032-2040.

Laurance W.F., Ferreira L.V., Rankin-de Merona J.M., Laurance S.G., Hutchings R.W. & Lovejoy T.E. (1998b) Effects of forest fragmentation on recruitment patterns in Amazonian tree communities. *Conservation Biology*, 12, 460-464.

Lawton J.H., Bignell D.E., Bolton B., Bloemers G.F., Eggleton P., Hammond P.M., Hodda M., Holt R.D., Larsen T.B., Mawdsley N.A., Stork N.E., Srivastava D.S. & Watt A.D. (1998) Biodiversity inventories, indicator taxa and effects of habitat modification in tropical forest. *Nature*, 391, 72-76.

Legendre P. (2000) Comparison of permutation methods for the partial correlation and partial Mantel tests. *Journal of Statistical Computing and Simulations*, 67, 37-73.

Levenson J.B. (1981) Woodlots as biogeographic islands in southeastern Wisconsin. In: *Forest Island Dynamics in Man-Dominated Landscapes* (eds. Burgess RL & Sharpe DM), pp. 13-39. Springer-Verlag, New York, USA.

Levins R. (1969) Some demographic and genetic consequences of environmental heterogeneity for biological control. *Bulletin of Entomological Society of America*, 15, 237-40.

Levins R.A. (1970) Extinction. *Lectures on Mathematics in Life Sciences*, 2, 75-107.

Li H. & Reynolds J.H. (1993) A new contagion index to quantify spatial patterns of landscapes. *Landscape Ecology*, 8, 155-162.

Longhorn R. (2001) (*Personal Communication*) Spatial Information access problems. In, Oxford.

Luiselli L. & Capizzi D. (1997) Influences of area, isolation and habitat features on distribution of snakes in Mediterranean fragmented woodlands. *Biodiversity and Conservation*, 6, 1339-1351.

Lynam A.J. (1997) Rapid decline of small-mammal diversity in monsoon evergreen forest fragments in Thailand (Chapter 15). In: *Tropical Forest Remnants: Ecology, Management and Conservation of Fragmented Communities* (eds. Laurance WF & Bierregaard RO), pp. 502-514. The University of Chicago Press, Chicago.

Lynch J.F. & Whigham D.F. (1984) Effects of forest fragmentation on breeding bird communities in Maryland, USA. *Biological Conservation*, 28, 287-324.

MacArthur R.H., Diamond J.M. & Karr J.R. (1972) Density compensation in island faunas. *Ecology*, 53, 330-342.

MacArthur R.H. & MacArthur J.W. (1961) On bird species diversity. *Ecology*, 42, 594-598.

MacArthur R.H. & Wilson E.O. (1967) *The Theory of Island Biogeography*. Princeton University Press, Princeton, NJ.

Mace G.M., Balmford A., Boitani L., Cowlishaw G., Dobson A.P., Faith D.P., Gaston K.J., Humphries C.J., Vane-Wright R.I., Williams P.H., Lawton J.H., Margules C.R., May R.M., Nicholls A.O., Possingham H.P., Rahbek C. & van Jaarsveld A.S. (2000) It's time to work together and stop duplicating conservation efforts ... *Nature*, 405, 393.

MacKinnon J. & MacKinnon K. (1986) Review of Protected Area System in Indo-Malayan Realm. In. IUCN, UNEP, Gland, Switzerland.

MacKinnon J., MacKinnon K., Child G. & Thorsell J. (1992) *Managing Protected Areas in the Tropics*. IUCN, Gland, Switzerland.

MacKinnon K. (1997) Minimising species loss: The imperative of protection. In: *Last Stand: Protected Areas and the Defense of Tropical Biodiversity* (eds. Kramer R, van Schaik C & Johnston J), pp. 36-63. Oxford University Press, New York.

Magellan Corporation (2000) Magellan Consumer Products. URL http://www.magellangps.com/frames/frame2.htm (Last accessed: March 2002).

Magurran A.E. (1988) *Ecological Diversity and its Measurement*. Chapman and Hall, London.

Malhotra K.C. (1998) Anthropological dimensions of sacred groves in India: An overview. In: *Conserving the Sacred for Biodiversity Management* (eds. Ramakrishnan PS, Saxena KG & Chandrashekara UM), pp. 423-438. Oxford and IBH Publishing Co., New Delhi.

Malhotra K.C., Gokhale Y., Chatterjee S. & Srivastava S. (2001) *Cultural and Ecological Dimensions of Sacred Groves in India*. Indian National Science Academy; Indira Gandhi Rashtriya Manav Sangrahalaya, New Delhi; Bhopal, India.

Mandelbrot B. (1983) *The Fractal Geometry of Nature*. Freeman, New York.

Mantel N. (1967) The detection of disease clustering and generalized regression approach. *Cancer Research*, 27, 209-220.

Mantel N. & Valand R.S. (1970) A technique of nonparametric multivariate analysis. *Biometrics*, 26, 547-558.

MapInfo Corporation (2001a) The Cookie Cutter 1997 Version. URL http://www.mapinfo.com/free/mb_cookie.cfm (Last accessed: March 2002).

MapInfo Corporation (2001b) MapBasic 3.0 Utilities. URL http://www.mapinfo.com/free/mb_30util.cfm (Last accessed: March 2002).

MapInfo Corporation (2001c) MapInfo Professional. URL http://dynamo.mapinfo.com/products/web/Overview.cfm?productid=44 (Last accessed: March 2002).

Margules C.R., Higgs A.J. & Rafe R.W. (1982) Modern biogeograhic theory: Are there any lessons for nature reserve design? *Biological Conservation*, 24, 115-128.

Margules C.R. & Pressey R.L. (2000) Systematic conservation planning. *Nature*, 405, 243-253.

McCune B. & Mefford M.J. (1999) Multivariate Analysis of Ecological Data. In. MjM Software, Gleneden Beach, OR, USA.

McGarigal K. & Marks B. (1995) FRAGSTATS: Spatial Pattern Analysis Program for Quantifying Landscape Structure (General Technical Report PNW-GTR-351). In. USDA Forest Service, Pacific Northwest Research Station, Portland, OR, USA.

McNeely J.A. (1996) Partnerships for conservation: An introduction. In: *Expanding Partnerships in Conservation* (ed. McNeely JA), pp. 1-10. Island Press, Washington DC.

Meffe G.K. & Carroll R.C. (1997) *Principles of Conservation Biology*. 2nd edn. Sinauer Associates, Sunderland, MA.

Miller J., Brookes R. & Croonquist M. (1997) Effects of landscape patterns on biotic communities. *Landscape Ecology*, 12, 137-153.

Miller J.R. & Cale P. (2000) Behavioral mechanisms and habitat use by birds in a fragmented agricultural landscape. *Ecological Applications*, 10, 1732-1748.

Miller R.M. & Lodge D.J. (1997) Fungal responses to disturbance: Agriculture and forestry. In: *The Mycota IV: Environmental and Microbial Relationships* (ed. Wicklow DT). Springer-Verlag, Berlin, Heidelberg.

Milne B.T. (1988) Measuring the fractal geometry of landscapes. *Applied Mathematics and Computation*, 27, 67-79.

Milne B.T. & Forman R.T.T. (1986) Peninsulas in Maine: Woody plant diversity, distance and environmental patterns. *Ecology*, 67, 967-974.

Ministry of Environment and Forests (1985) India: Country Report 1985 (Presented to IX World Forestry Congress). In. Government of India, New Delhi.

Miranda M.L. & LaPalme S. (1997) User rights and biodiversity conservation. In: *Last Stand: Protected Areas and the Defense of Tropical Biodiversity* (eds. Kramer R, van Schaik C & Johnson J), pp. 133-161. Oxford University Press, New York.

Moguel P. & Toledo V. (1999) Biodiversity conservation in traditional coffee systems in Mexico. *Conservation Biology*, 13, 11-21.

Moppert B. (2000) Expansion of coffee plantations and landscape changes (The Dynamics of Land Use - Chapter 5). In: *Mountain Biodiversity, Land Use Dynamics, and Traditional Knowledge* (eds. Ramakrishnan PS, Chandrashekara UM, Elouard C, Guilmoto CZ, Maikhuri RK, Rao KS, Sankar S & Saxena KG), pp. 88-98. Man and the Biosphere Programme, Oxford and IBH Publishing Co. Pvt. Ltd., New Delhi

Morris D.W. (1995) Earth's peeling veneer of life. *Nature*, 373, 25.

Murakami Y. (1987) Spatial distribution of *Russula* species in *Castanopsis cuspidata* forest. *Transactions of the British Mycological Society*, 89, 187-193.

Murcia C. (1995) Edge effects in fragmented forests: Implications for conservation. *Trends in Ecology and Evolution*, 10, 58-62.

Myers N. (1988) Tropical forests: Much more than stocks of wood. *Journal of Tropical Ecology*, 4, 209-221.

Myers N. (1989) Threatened biotas: "Hotspots" in tropical forests. *Environmentalist*, 8, 1-20.

Myers N. (1990) The biodiversity challenge: Expanded hotspots analysis. *Environmentalist*, 10, 243-256.

Myers N., Mittermeier R.A., Mittermeier C.G., da Fonseca G.A.B. & Kent J. (2000) Biodiversity hotspots for conservation priorities. *Nature*, 403, 853-858.

Narayana A. (1999) New index for assessment. In: *The Deccan Herald (6th July 1999)*, Bangalore, India.

Natel P. & Neumann P. (1992) Ecology of ectomycorrhizal-basidiomycete communities on a local vegetation gradient. *Ecology*, 73, 99-117.

National Geographic Society (2001) India's Western Ghats. URL http://magma.nationalgeographic.com/ngm/data/2002/01/01/html/ft_20020101.6.html (Last accessed: March 2002).

Nekola J.C. & White P.S. (1999) The distance decay of similarity in biogeography and ecology. *Journal of Biogeography*, 26, 867-878.

Nikora V., Pearson C. & Shankar U. (1999) Scaling properties in landscape patterns: New Zealand experience. *Landscape Ecology*, 14, 17-33.

Nilsson S.G. (1986) Are bird comunities in small biotope patches random samples from communities in large patches? *Biological Conservation*, 38, 179-204.

Norton-Griffiths N. & Southey C. (1995) The opportunity costs of biodiversity conservation in Kenya. *Ecological Economics*, 12, 125-139.

Oliver I.O. & Beattie A.J. (1996) Designing a cost-effective invertebrate survey: A test of methods for rapid assessment of biodiversity. *Ecological Applications*, 6, 594-607.

O'Neill R.V., Milne B.T., Turner M.G. & Gardner R.H. (1988) Resource utilisation scales and landscape pattern. *Landscape Ecology*, 2, 63-69.

Onset Computer Corporation (1996-2002a) BoxCar Pro 4.0 for Windows. URL http://www.onsetcomp.com/Products/Product_Pages/BoxCar_Pro_pages/4262_BCP40.html (Last accessed: March 2002).

Onset Computer Corporation (1996-2002b) HOBO® H8 Pro RH/ Temperature Logger. URL http://www.onsetcomp.com/Products/Product_Pages/HOBO_H08/3929_HOBO_ProRHtemp.html (Last accessed: March 2002).

Opdam P., van Apeldoorn R., Schotman A. & Kalkhoven J. (1993) Population responses to landscape fragmentation (Chapter7). In: *Landscape Ecology of a Stressed Environment* (eds. Vos CC & Opdam P), pp. 147-171. Chapman and Hall, London.

Opdam P.F.M., Rijsdijk G. & Hustings F. (1985) Bird communities in small woods in an agricultural landscape: Effects of area and isolation. *Biological Conservation*, 34, 333-352.

Owiunji I. & Plumptre A.J. (1998) Bird communities in logged and unlogged compartments in Budongo forest, Uganda. *Forest Ecology and Management*, 108, 115-126.

Panayotou T. & Ashton P.S. (1992) *Not By Timber Alone: Economics and Ecology of Sustaining Tropical Forests*. Island Press, Washington DC.

Panwar H.S. (1982) Project Tiger. In: *Wildlife in India* (ed. Saharia VB), pp. 130-137. Natraj Publications, Dehra Dun.

Parker-Rhodes A.F. (1951) The basidiomycets of Skokoholm island. VII. Some floristic and ecological calculations. *New Phytologist*, 50, 227-243.

Parresol B.S. & McCollum J. (1997) Characterizing and comparing landscape diversity using GIS and a contagion index. In: *Sustainable Forests: Global Challenges and Local Solutions* (eds. Bouman OT & Brand DG), pp. 249-261. Food Products Press, Binghamton, New York, USA.

Pascal J.P. (1988) *Wet Evergreen Forests of the Western Ghats of India: Ecology, Structure, Floristic Composition and Succcession.* Institute Francais de Pondichery, Pondicherry.

Pascal J.P. & Meher-Homji V.M. (1986) Phytochorology of Kodagu (Coorg) district, Karnataka. *Journal of Bombay Natural History Society*, 83, 43-56.

Pascal J.P. & Pelissier R. (1996) Structure and floristic composition of a tropical evergreen forest in southern India. *Journal of Tropical Ecology*, 12, 191-214.

Pascal J.P. & Ramesh B.R. (1987) *A field key to the trees and lianas of the evergreen forests of the Western Ghats (India).* Institute Francais de Pondichery, Publication du Department d'ecologie, Pondicherry.

Pascal J.P., Shyam Sunder S. & Meher-Homji V.M. (1982) Forest Map of South India - Mercara-Mysore. In. State Forest Departments (Karnataka and Kerala); French Institute, Pondicherry.

Pearson S.M., Turner M.G., Gardner R.H. & O'Neill R.V. (1996) An organism-based perspective of habitat fragmentation. In: *Biodiversity in Managed Landscapes: Theory and Practice* (ed. Szaro RC), pp. 77-95. Oxford University Press, Oxford.

Pegler D.N. (1977) *A preliminary Agaric Flora of East Africa.* Her Majesty's Stationary Office, London.

Peres C.A. & Terborgh J.W. (1995) Amazonian nature reserves: An analysis of the defensibility status of existing conservation units and design criteria for the future. *Conservation Biology*, 9, 34-46.

Perfecto I., Rice R.A., Greenberg R. & van der Voort M.E. (1996) Shade coffee: A disappearing refuge for biodiversity. *BioScience*, 46, 598-608.

Pimm S.L., Lee Jones H. & Diamond J.M. (1988) On the risk of extinction. *The American Naturalist*, 132, 757-785.

Pinedo-Vasquez M., Zarin D., Jipp P. & Chota-Inuma J. (1990) Use-values of tree species in a communal forest reserve in northeast Peru. *Conservation Biology*, 4, 405-416.

Poffenberger M. (1996) Community restoration of forests in India. In: *Forest Patches in Tropical Landscapes* (eds. Schelhas J & Greenberg R), pp. 366-380. Island Press, Washington DC.

Ponnappa K.C. (1997) *A Study of the Origins of Coorgs.* Ponnappa, K. C. Lt. Col. (Retd.), Chettalli, Kodagu, Karnataka, India.

Posey D.A. (1998) *Cultural and spiritual values of biodiversity.* IUCN, UNEP, Gland, Switzerland.

Prendergast J.R., Quinn R.M., Lawton J.H., Eversham B.C. & Gibbons D.W. (1993) Rare species, the coincidence of diversity hotspots and conservation strategies. *Nature*, 365, 335-337.

Preston F.W. (1960) Time and space variation of species. *Ecology*, 41, 611-627.

Primack R.B. (1998) *Essentials of Conservation Biology.* 2nd edn. Sinauer Associates, Sunderland, MA.

Prins H.H.T. (1992) The pastoral road to extinction: Competition between wildlife and traditional pastoralism in East Africa. *Environmental Conservation*, 19, 117-23.

Pugh G.J.F. & Boddy L. (1988) A view of disturbance and life strategies in fungi. *Proceedings of the Royal Society of Edinburgh*, 94, 3-11.

Putz F.E. (1991) Silvicultural effects of lianas. In: *The Biology of Vines* (eds. Putz FE & Mooney HA), pp. 493-501. Cambridge University Press, Cambridge, UK.

Putz F.E. & Pinard M. (1993) Reduced impact logging as a carbon-offset method. *Conservation Biology*, 7, 755-757.

Quinn J.F. & Harrison S.P. (1988) Effects of habitat fragmentation and isolation on species richness: Evidence from biogeographic patterns. *Oecologia*, 75, 132-140.

Rakowski W. & Clark M.A. (1998) Do groups of women aged 50 to 75 match the national average mammography rate? *Americal Journal of Preventive medicine*, 15, 187-197.

Ramesh B.R., Menon S. & Bawa K.S. (1997) A vegetation-based approach to biodiversity gap analysis in the Agastyamalai region, Western Ghats, India. *Ambio*, 26, 529-536.

Ramesh B.R. & Pascal J.P. (1997) *Atlas of Endemics of the Western Ghats (India): Distribution of Tree Species in the Evergreen and Semi-Evergreen Forests.* Institute Francais de Pondichery, Publications du Department d'ecologie, Pondicherry.

Rangarajan M. (1994) *Fencing the Forest.* Oxford University Press, Oxford.

Reddy A.N.Y. (1990) *Grevillea robusta* in coffee plantations of Karnataka. *Myforest*, 26, 335-351.

Redford K.H. (1992) The empty forest. *Bioscience*, 42, 412-422.

Reid W.V. (1998) Biodiversity hotspots. *Trends in Ecology and Evolution*, 13, 275-280.

Rejwan C., Collins N.C., Brunner L.J., Shuter B.J. & Ridgway M.S. (1999) Tree regression analysis on the nesting habitat of smallmouth bass. *Ecology*, 80, 341-348.

Renjifo L.M. (1999) Composition changes in a sub-Andean avifauna after long-term forest fragmentation. *Conservation Biology*, 13, 1124-1139.

Restrepo C., Renjifo L.M. & Marples P. (1997) Frugivorous birds in fragmented neotropical montane forest: Landscape patterns and body mass distribution (Chapter 12). In: *Tropical Forest Remnants: Ecology, Management and Conservation of Fragmented Communities* (eds. Laurance WF & Bierregaard RO), pp. 171-187. The University of Chicago Press, Chicago.

Richter G. (1870) *The Gazetteer of Coorg.* Low Price Publications, Delhi.

Robinson S.K., Thompson F.R.I., Donovan T.M., Whitehead D.R. & Faaborg J. (1995) Regional forest fragmentation and the nesting success of migratory birds. *Science*, 267, 1987-1990.

Rodgers W.A. & Panwar H.S. (1988) Planning a Wildlife Protected Area Network in India, 2 Volumes, Project FO:IND/82/003. In. Wildlife Institute of India, Dehradun.

Rosenzweig M.L. (1995) *Species Diversity in Time and Space.* Cambridge University Press, Cambridge, UK.

Rydin H., Diekmann M. & Hallingback T. (1997) Biological characteristics, habitat associations and distribution of macrofungi in Sweden. *Conservation Biology*, 11, 628-640.

Sanderson S.E. & Redford K.H. (1997) Biodiversity politics and the contest for ownership of the world's biota. In: *Last Stand: Protected Areas and the Defense of Tropical Biodiversity* (eds. Kramer R, van Schaik C & Johnson J), pp. 115-132. Oxford University Press, New York.

Saunders D.A., Arnold G.W., Burbidge A.A. & Hopkins A.J.M. (1987) *Nature Conservation: The Role of Remnants of Native Vegetation*. Surrey Beatty & Sons, Chipping Norton, New South Wales, Australia.

Saunders D.A., Hobbs R.J. & Margules C.R. (1991) Biological consequences of ecosystem fragmentation: A review. *Conservation Biology*, 5, 18-32.

Schonewald-Cox C.M. & Bayless J.W. (1986) The boundary model: A geographical analysis of design and conservation of nature reserves. *Biological Conservation*, 38, 305-322.

Schröder V.W., Fränzle O., Vetter L. & Saager W. (1992) CHAID - Analyse der Terrestrischen Waldschadensinventur 1988 in Schleswig Holstein. *Allgemeine Forst und Jagdzeitung*, 163, 93-98.

Schwartz M. (1997) *Conservation in highly fragmented landscapes*. Chapman and Hall, New York, USA.

Senn-Irlet B. & Bieri G. (1999) Sporocarp succession of soil-inhabiting macrofungi in an autochtonous subalpine Norway Spruce forest of Switzerland. *Forest Ecology and Management*, 124, 169-175.

Shafer C.L. (1990) *Nature Reserves: Island Theory and Conservation Practice*. Smithsonian Institute Press, Washington DC.

Sharma N., Rowe R., Grut M., Kramer R. & Gregersen H. (1992) Conditions for Sustainable Development. In: *Managing the World's Forests: Looking for Balance Between Conservation and Development* (ed. Sharma NP), pp. 489-513. International Bank for Reconstruction and Development; Kendall/Hunt Publishing Company, Dubuque, Iowa, USA.

Sieving C.E. & Karr J.R. (1997) Avian extinction and persistence mechanisms in lowland Panama (Chapter 11). In: *Tropical Forest Remnants: Ecology, Management and Conservation of Fragmented Communities* (eds. Laurance WF & Bierregaard RO), pp. 156-170. The University of Chicago Press, Chicago.

Simberloff D. (1991) Review of Theory Relevant to Acquiring Land: Report to Florida Department of Natural Resources. In. Florida State University, Tallahassee, FL, USA.

Simberloff D.S. & Abele L.G. (1976) Island biogeography theory and conservation practice. *Science*, 191, 285-286.

Sinclair A.R.E., Ludwig D. & Clark C.W. (2000) Conservation in the real world. *Science*, 289, 1875.

Singh S. (1999) Assessing management effectiveness of wildlife protected areas in India. In: *International workshop on management effectiveness of protected areas, June 14-16, 1999*. IUCN / WWF, CATIE, Turrialba, Costa Rica.

Smith A.P. (1997) Deforestation, fragmentation and reserve design in western Madagascar (Chapter 27). In: *Tropical Forest Remnants: Ecology, Management and Conservation of Fragmented Communities* (eds. Laurance WF & Bierregaard RO), pp. 415-441. The University of Chicago Press, Chicago.

Somiah K.K. (1954) *Working Plan for The Ghat Forests of Coorg*. Karnata State Forest Department, Working Plan Division, Mysore.

Soubadra Devy M. & Ganeshaiah K.N. (2001) (*Personal Communication*) Butterfly diversity left outside the Protected Area (PA) network! In: (ed. RANWA e-mail discussion group), Bangalore.

Southgate D. & Clark H.L. (1993) Can conservation projects save biodiversity in South America? *Ambio*, 22, 163-166.

SPSS Inc. (1989-1999) SPSS for Windows Release 10.0.5. In. SPSS Inc.

Stärk K.D.C. & Pfeiffer D.U. (1999) The application of non-parametric techniques to solve classification problems in complex data sets in veterinary epidemiology: An example. *Intelligent Data Analysis*, 3, 23-35.

Stouffer P.C. & Bierregaard R.O. (1995) Effects of forest fragmentation on understory hummingbirds in Amazonian Brazil. *Conservation Biology*, 9, 1085-1094.

Swaine M.D. & Whitmore T.C. (1988) On the definition of ecological species groups in tropical rainforests. *Vegetatio*, 75, 81-86.

Temple S.A. (1986) Recovery of the endangered Mauritius Kestrel from an extreme population bottleneck. *Auk*, 103, 632-633.

Terborgh J. (1974) Preservation of natural diversity : The problem of extinction-prone species. *Bioscience*, 24, 715-722.

Terborgh J., Robinson S.K., Parker T.A.I., Munn C.A. & Pierpont N. (1990) Structure and organisation of an Amazonian bird community. *Ecological Monographs*, 60, 213-238.

Terborgh J.W. (1997) Transitory states in relaxing ecosystems of land bridge islands (Chapter 17). In: *Tropical Forest Remnants: Ecology, Management and Conservation of Fragmented Communities* (eds. Laurance WF & Bierregaard RO), pp. 256-274. The University of Chicago Press, Chicago.

Thébaud C. & Strasberg D. (1997) Plant dispersal in fragmnented landscapes: A field study of woody colonisation in rainforest remnants of the mascarene archipelago (Chapter 21). In: *Tropical Forest Remnants: Ecology, Management and Conservation of Fragmented Communities* (eds. Laurance WF & Bierregaard RO), pp. 321-332. The University of Chicago Press, Chicago.

Thiollay J.-M. (1994) The role of traditional agroforests in the conservation of rain forest bird diversity in Sumatra. *Conservation Biology*, 9, 335-353.

Thogmartin W.E. & Schaeffer B.A. (2000) Landscape attributes associated with mortality events of wild turkeys in Arkansas. *Wildlife Society Bulletin*, 28, 865-874.

Thomas C.D. & Hanski I.A. (1997) Butterfly Metapopulations. In: *Metapopulation Biology: Ecology, Genetics and Evolution* (eds. Hanski IA & Gilpin ME), pp. 359-386. Academic Press, San Diego, California.

Tocher M.D., Gascon C. & Zimmerman B.L. (1997) Fragmentation effects on a Central Amazonian frog community: A ten-year study (Chapter 9). In: *Tropical Forest Remnants: Ecology, Management and Conservation of Fragmented Communities* (eds. Laurance WF & Bierregaard RO), pp. 124-137. The University of Chicago Press, Chicago.

Tofts R.J. & Orton P.D. (1998) The species accumulation curve for Agarics and Boleti from a Caledonian Pinewood. *Mycologist*, 12, 98-102.

Tokeshi M. (1993) Species abundance patterns and community structure. *Advances in Ecological Research*, 24, 111-186.

Turner I.M. (1996) Species loss in fragments of tropical rain forest: A review of evidence. *Journal of Applied Ecology*, 33, 200-209.

Turner I.M., Chua K.S., Ong J., Soong B. & Tan H. (1996) A century of plant species loss from an isolated fragment of lowland tropical rain forest. *Conservation Biology*, 10, 1229-1244.

Turner I.M., Tan H.T.W., Wee Y.C., Ibrahim A.B., Chew P.T. & Corlett R.T. (1994) A study of plant species extinction in Singapore: Lessons for the conservation of tropical biodiversity. *Conservation Biology*, 8, 705-712.

Turner M.G. (1989) Landscape ecology: The effect of pattern on processes. *Annual Review of Ecology and Systematics*, 20, 171-197.

Utkarsh G., Joshi N.V. & Gadgil M. (1998) On the patterns of tree diversity in the Western Ghats of India. *Current Science*, 75, 594-603.

van der Heijden M.G.A., Klironomos J.N., Ursic M., Moutoglis P., Streitwolf-Engel R., Boller T., Wiemken A. & Sanders I.R. (1998) Mycorrhizal fungal diversity determines plant biodiversity, ecosystem variability and productivity. *Nature*, 396, 69-72.

van Dorp D. & Opdam P.F.M. (1987) Effects of patch size, isolation and regional abundance on forest bird communities. *Landscape Ecology*, 1, 59-73.

Viana V.M., Tabanez A.A.J. & Batista J.L.F. (1997) Dynamics and restoration of forest fragments in the Brazilian Atlantic moist forest (Chapter 23). In: *Tropical Forest Remnants: Ecology, Management and Conservation of Fragmented Communities* (eds. Laurance WF & Bierregaard RO), pp. 502-514. The University of Chicago Press, Chicago.

Vijaya T.P. (2000) Contemporary Society and Land Tenure (Landscape and Society - Chapter 3). In: *Mountain Biodiversity, Land Use Dynamics, and Traditional Knowledge* (eds. Ramakrishnan PS, Chandrashekara UM, Elouard C, Guilmoto CZ, Maikhuri RK, Rao KS, Sankar S & Saxena KG), pp. 44-53. Man and the Biosphere Programme, Oxford and IBH Publishing Co. Pvt. Ltd., New Delhi.

Vogt K.A., Bloomfield J., Ammirati J.F. & Ammirati S.R. (1992) Sporocarp production by basidiomycetes, with emphasis on forest ecosystems. In: *The Fungal Community: Its Organisation and Role in the Ecosystem* (eds. Carroll GC & Wicklow DT), pp. 563-581. Marcel Dekker, New York.

Vogt K.A., Edmonds R.L. & Grier C.C. (1981) Biomass and nutrient concentrations of sporocarps produced by mycorrhizal and decomposer fungi in *Abies amabilis* stands. *Oecologia*, 50, 170-175.

Warburton N.H. (1997) Structure and conservation of forest avifauna in isolated rainforest remnants in tropical Australia (Chapter 13). In: *Tropical Forest Remnants: Ecology, Management and Conservation of Fragmented Communities* (eds. Laurance WF & Bierregaard RO), pp. 190-206. The University of Chicago Press, Chicago.

Wasterlund I. (1989) Hur paverkar skogens skostel forekomsten av storsvamparnas fruktkroppar? *Svensk. Bot. Tidskr.*, 83, 103-112.

Waters J.R., McKelvey K.S., Luoma D.L. & Zabel C.J. (1997) Truffle production in old-growth and mature Fir stands in northeastern California. *Forest Ecology and Management*, 96, 155-166.

Watkinson S.C. (2002) (*Personal Communication*) Mycorrhizal fungi. In, Oxford.

Watling R. (1995) Assessment of fungal diversity: Macromycetes, the problems. *Canadian Journal of Botany*, 73 (Suppl.1), S15-S24.

WCMC (1992) *Global Biodiversity: Status of the Earth's Living Resources*. World Conservation Monitoring Centre; Chapman and Hall, London, UK.

WCPA (2001) World Commission on Protected Areas. URL http://wcpa.iucn.org (Last accessed: March 2002).

Wells M., Brandon K. & Hannah L. (1992) *People and Parks: Linking Protected Area Management and Local Communities*. World Bank; Worldwide Fund for Nature; and the United States Agency for International Development, Washington DC.

Western D. (1989) Conservation without parks: Wildlife in the rural landscape. In: *Conservation for the Twenty-first Century* (eds. Western D & Pearl M). Oxford University Press, New York.

Whitcomb R.F., Robbins L.S., Lynch J.F., Whitcomb B.L., Klimkiewicz M.K. & Bystrak D. (1981) Effects of forest fragmentation on avifauna of the eastern deciduous forest. In: *Forest Island Dynamics in Man-Dominated Landscapes* (eds. Burgess RL & Sharpe DM), pp. 125-205. Springer, New York, USA.

Whitmore T.C. (1997) Rainforest disturbance, disappearance and species loss (Chapter 1). In: *Tropical Forest Remnants: Ecology, Management and Conservation of Fragmented Communities* (eds. Laurance WF & Bierregaard RO), pp. 3-12. The University of Chicago Press, Chicago.

Whitmore T.C., Brown N.D., Swaine M.D., Kennedy D., Goodwin-Bailey C.I. & Gong W.K. (1993) Use of hemispherical photographs in forest ecology: Measurement of gap size and radiation totals in a Bornean tropical rainforest. *Journal of Tropical Ecology*, 9, 131-151.

Whittaker R.J. (1998) *Island Biogeography: Ecology, Evolution and Conservation*. Oxford University Press, Oxford, UK.

Wiens J.A. (1997) Metapopulation dynamics and landscape ecology. In: *Metapopulation Biology: Ecology, Genetics and Evolution* (eds. Hanski IA & Gilpin ME), pp. 43-62. Academic Press, San Diego, California.

Wiens J.A., Stenseth N.C., van Horne B. & Ims R.A. (1993) Ecogical mechanisms and landscape ecology. *Oikos*, 45, 421-427.

Wilcox B.A. (1980) Insular ecology and conservation. In: *Conservation Biology: An Evolutionary-Ecological Perspective* (eds. Soulé ME & Wilcox BA), pp. 95-117. Sinauer Associates, Sunderland, MA.

Wilkins W.H., Ellis E.M. & Harley J.L. (1937) The ecology of larger fungi. I. Constancy and frequency of fungal species in relation to certain vegetation communities, particularly Oak and Beech. *Annals of Applied Biology*, 24, 703-732.

Williams P., Gibbons D., Margules C., Rebelo A., Humphries C. & Pressey R. (1996) A comparison of richness hotspots, rarity hotspots and complementary areas for conserving diversity of British birds. *Conservation Biology*, 10, 155-174.

Willis E.O. (1974) Populations and local extinctions of birds on Barro Colorado Island, Panama. *Ecological Monographs*, 44, 153-169.

Wilson E.O. (1992) *The Diversity of Life*. Allen Lane Penguin Press, London.

Wilson M.H., Kepler C.B., Snyder N.F.R., Derrickson S.R., Dein F.J., Wiley J.W., Wunderle J.M.J., Lugo A.E., Graham D.L. & Toone W.D. (1994) Puerto Rican parrots and potential limitations of the metapopulation approach to species conservation. *Conservation Biology*, 8, 114-123.

World Bank (1994) *Dissemination Notes: 10*. World Bank, Washington DC.

World Bank (1997) Investing in Biodiversity: A Review of Indonesia's Integrated Conservation and Development Projects. In. World Bank, Indonesia and Pacific-Island Country Department, Jakarta.

WWF (2001) Forests for Life. URL World-Wide Fund for Nature and Natural resources http://www.panda.org/forests4life/about.cfm (Last accessed: March 2002).

Zenner E.K. & Hibbs D.E. (2000) A new method for modeling the heterogeneity of forest structure. *Forest Ecology and Management*, 129, 75-87.

8. APPENDICES

APPENDIX 1

IUCN PROTECTED AREA CATEGORIES (IUCN 1994)

CATEGORY I

Strict Nature Reserve/Wilderness Area: Protected area managed mainly for science or wilderness protection.

Category I a

Strict Nature reserve: Protected area managed mainly for science.

Definition: Area of land and/or sea possessing some outstanding or representative ecosystems, geological or physiographical features and/or species, available primarily for scientific research and/or environmental monitoring.

Category I b

Wilderness Area: Protected area managed mainly for wilderness protection.

Definition: Large area of unmodified or slightly modified land and/or sea, retaining its natural character and influence, without permanent or significant habitation, which is protected and managed so as to preserve its natural condition.

CATEGORY II

National Park: Protected area managed mainly for ecosystem protection and recreation.

Definition: Natural area of land and/or sea, designated to (a) protect the ecological integrity of one or more ecosystems for present and future generations, (b) exclude exploitation or occupation inimical to the purposes of designation of the area and (c) provide a foundation for spiritual, scientific, educational, recreational and visitor opportunities all of which must be environmentally and culturally compatible.

CATEGORY III

Natural monument: Protected area managed mainly for conservation of specific natural features.

Definition: Area containing one, or more, specific natural or natural/cultural feature which is of outstanding or unique value because of its inherent rarity, representative or aesthetic qualities or cultural significance.

CATEGORY IV

Habitat/Species Management Area: Protected area managed mainly for conservation through management intervention.

Definition: Area of land and/or sea subject to active intervention for management purposes so as to ensure the maintenance of habitats and/or to meet the requirements of specific species.

CATEGORY V

Protected Landscape / Seascape: Protected area managed mainly for landscape/seascape conservation and recreation.

Definition: Area of land, with coast and sea as appropriate, where the interaction of people and nature over time has produced an area of distinct character with significant aesthetic, ecological and/or cultural value and often with high biological diversity. Safeguarding the integrity of this traditional interaction is vital to the protection, maintenance and evolution of such an area.

CATEGORY VI

Managed Resource Protected Area: Protected area managed mainly for the sustainable use of natural ecosystems.

Definition: Area containing predominantly unmodified natural systems, managed to ensure long term protection and maintenance of biological diversity, while providing at the same time a sustainable flow of natural products and services to meet community needs.

APPENDIX 2

LAND TENURES IN KODAGU (Adapted from Haller 1910)

Note: An asterisk indicates a non-tree covered land use type. Those land use types without asterisks are tree-covered in most cases.

1. ***Bane***: Forest land granted for the service of the holding of wet land to which it is allotted, to be held free of revenue by the cultivator for grazing and to supply leaf manure, firewood and timber required for the agricultural and domestic purposes of the cultivator, so long as he continues in possession of the wet land.

- *1a.* ***Alienated Bane***: Land originally granted as Bane but which has become subject to assessment owing to its separation from the wet land for the service of which it was formerly held.
- *1b.* ***Jahagir Bane***: Bane attached to Jahagir land.
- *1c.* ***Jamma Bane***: Bane attached to Jamma wet land.
- *1d.* ***Privileged Bane***: (Jamma, Jagu, Jodi, Jahagir, Umbli) Bane lands attached to wet lands and obtained by the owners prior to 21-05-1886. Cultivation not exceeding 10 acres is exempted from assessment.
- *1e.* ***Umbli Bane***: Bane attached to Umbli land.
- *1f.* ***Unprivileged Bane***: (Jamma, Sagu, Jodi) Bane lands obtained by purchase or otherwise subsequent to 21-05-1886 having no assessment-free area of 10 acres as in the case of privileged Bane lands.

2. ***Barike***: Low-lying Bane lands capable of being brought under paddy cultivation is known as Barike and was assessed when brought under cultivation.

3. ***Battamanya***: Lands granted by the Rajas to Brahmins and their lineal descendants on condition of the performance of certain religious ceremonies.

4. ***Devarakadu***: Sacred forests usually assigned to some particular deity or temple. The right to take firewood for temple worship, materials for constructing pandals and with special permission timbers for repairing the temples are allowed to the temple authorities and servants, while the villagers generally have the right of way, water and for grazing.

5. ***Garekad***: A narrow strip of highland adjoining the wet land, not more than 16 feet in breadth. This is attached to the wet land.

6. ***Gaudumbali***: Lands held by village Patels (headman). These lands are inalienable.

7. ***Hithlu and Hithlumanedala***: Forest land granted in connection with a wet holding, to be held free of revenue by the cultivator as a site for houses, cattle sheds and gardens so long as it is not separated from the wet land.

- *7a.* ***Alienated Hithlu***: Refers to Hithlumanedala, which has been alienated apart from the wet land to which it was originally attached. This is liable to assessment from the date of alienation and the owner has got a right to utilise timber for bona-fide purpose in the village only.

8. ***Hola or Sarige*** (*): Assessed dry lands.

9. ***Hullugavalu (Gomala)*** (*): Lands granted for grazing purpose and which cannot be utilised for any other purpose.

10. ***Jahagiri*** (*): Wet land held free of all assessment and granted by the Rajas and the British government in recognition of service rendered.

11. Jamma **(*)**: Wet land assessed at one half the normal (Sagu) rate of assessment. It was originally a tenure in consideration of military services.

12. Jodi land: Land of which one half of the revenue is assigned to a religious institution.

13. Karavu: Lands 16 to 30 feet in width granted by the Rajas to protect the margins of wet lands from ingression of cattle, over-hanging branches *etc.*, held free of assessment. This is attached to the wet land.

14. Maley: Mountainous or hilly land

> ***14a. Jamma Maleys***: Jamma maleys are portions of the forest reserve on the Western Ghats in which the hereditary rights of growing cardamom according to the indigenous system is recognised.
>
> ***14b. Leased Cardamom Maleys***: These are maleys leased from time to time by the commissioner and for which the resultant revenue is credited to the Forest Department. The lessee has no right to the land or timber. Originally these were called Cardamom Geni Maleys.

15. Manedala **(*)**: House sites given to non-agriculturalists. If cultivated, the land is liable to assessment at the ordinary rates.

16. Matta lands: Lands given to Matta for their maintenance, by the Rajas of Coorg (Kodagu). These lands are held free of all assessments.

17. Naimannu: Lands held by Kulawadis *i.e.* village watchmen. These are inalienable.

18. Ooruguppe: Land set apart for a collective village site.

19. Paisari: Wastelands belonging to the government.

20. Sagu: The ordinary ryotwari (citizenship) tenure of land held on full assessment.

> ***20a. Cardamom Sagu***: Lands granted for coffee cultivation, but as coffee failed, cardamom has almost entirely replaced it. Originally these lands were called as Coffee Saguvali Maleys.

21. Redeemed: The word 'redeemed' is used when timber has been paid for by the occupant of the land.

22. Sarvamanya: Lands granted for the benefit of religious institutions on the condition of the performance of religious ceremonies. Land revenue with respect to these lands is fully relinquished.

23. Un-redeemed: The word 'un-redeemed' is used when timber has not been paid for by the occupant of the land.

24. Urambala and Mandu **(*)**: Communal village lands reserved for Panchayat (village council) meetings and for dancing on festival occasions. The villagers have the right of grazing thereon.

25. Umbli **(*)**: Wet land originally granted for services rendered and held subject to the payment of specially light assessment, which is not liable for periodical revision.

26. Urudve: Communal lands *i.e.* village forest from where the ryots (citizens) are allowed to graze their cattle and take firewood and timber for agricultural purposes.

27. Wantihola **(*)**: Inferior dry lands held under a nominal rate of assessment.

APPENDIX 3

DISTRIBUTION OF TREE (≥ 10 cm dbh) SPECIES

Note: An asterisk indicates the occurrence of the species in the land use type.

Scientific name of species	Forest reserve	Sacred forests	Coffee plantations
Acrocarpus fraxinifolius	*	*	*
Acronychia pedunculata		*	*
Actinodaphne bourdillonii	*	*	*
Actinodaphne lawsonii		*	
Actinodaphne malabarica	*	*	*
Aglaia anamallayana	*	*	
Aglaia barberi	*		*
Aglaia elaeagnoidea	*		
Aglaia jainii	*	*	*
Aglaia simplicifolia	*	*	
Agrostistachys meeboldii	*		
Albizzia amara		*	*
Albizzia chinensis			*
Albizzia lebbek			*
Alstonia scholaris	*	*	*
Antidesma menasu	*	*	*
Antiaris toxicaria		*	
Aphananthe cuspidata	*	*	*
Aphanamixis polystachya	*	*	*
Apodytes beddomei	*	*	*
Aporosa lindleyana		*	*
Archidendron monadelphum	*	*	*
Ardisia solanacea		*	
Areca catechu			*
Artocarpus heterophyllus	*	*	*
Artocarpus hirsuta	*	*	*
Baccouria courtallensis	*		
Beilschmiedia wightii	*	*	*
Bischofia javanica	*	*	*
Blachia denudata	*		
Bombax ceiba			*
Bombax malabaricum		*	
Bridelia retusa			*
Calophyllum polyanthum	*	*	*
Callicarpa tomentosa	*	*	*
Canthium dicoccum		*	*
Canarium strictum	*	*	*
Careya arborea		*	*
Carallia brachiata			*
Caryota urens		*	*
Cassia fistula		*	*
Cassine glauca		*	*
Casearia ovata	*	*	

	Reserve	Sacred	Coffee
Casearia rubescens			*
Casearia wynadensis		*	
Ceiba pentandra			*
Celtis philippensis	*	*	*
Celtis tetrandra		*	
Chionanthus malabaricum	*	*	*
Chrysophyllum lanceolatum	*	*	*
Cinnamomum macrocarpum	*	*	*
Cinnamomum sulphuratum			*
Citrus reticulata			*
Clausena dentata	*	*	*
Cleidion spiciflorum	*	*	
Clerodendron viscosum	*	*	*
Coffea arabica		*	
Cryptocarya bourdillonii	*	*	*
Chrysophylum lanceolatum		*	
Cytheroxylon subserratum			*
Dalbergia latifolia	*	*	*
Dillenia pentagyna	*	*	*
Dimocarpus longan	*	*	*
Diospyros candolleana	*	*	*
Diospyros montana		*	*
Diospyros paniculata	*		*
Diospyros pruriens	*		
Diospyros sp.	*	*	
Diospyros sylvatica	*	*	*
Drypetes elata	*	*	*
Drypetes oblongifolia	*		
Dysoxylum malabaricum	*	*	*
Elaeocarpus munronii	*		
Elaeocarpus serratus	*	*	*
Elaeocarpus tuberculatus	*	*	*
Emblica officinalis			*
Erythrina indica		*	*
Euonymus indicus	*	*	*
Evodia lunu-ankenda	*	*	*
Excoecaria crenulata		*	*
Fragraea ceilanica		*	
Ficus amplissima		*	*
Ficus asperima		*	*
Ficus beddomei	*	*	*
Ficus benghalensis			*
Ficus callosa		*	*
Ficus glomerata			*
Ficus hispida		*	*
Ficus microcarpa		*	
Ficus mysorensis	*	*	*
Ficus nervosa	*	*	*
Ficus racemosa		*	*
Ficus sp.		*	*
Ficus tsjahela	*	*	*
Ficus virens	*	*	*
Flacourtia montana		*	*
Garcinia gummi-gutta	*	*	*

	Reserve	Sacred	Coffee
Garcinia indica	*		
Garcinia morella	*	*	*
Garcinia pictorius	*		
Glochidion bourdillonii	*	*	*
Glochidion malabaricum		*	
Glyricidia maculata			*
Gmelina arborea		*	*
Grevillea robusta			*
Grewia tiliaefolia			*
Harpullia arborea		*	*
Heritiera papilio	*		
Holigarna arnottiana	*	*	*
Holigarna beddomei	*	*	*
Holigarna grahamii	*	*	*
Holigarna nigra	*	*	*
Homalium travancoricum	*	*	
Homalium zeylanicum		*	*
Hopea parviflora		*	*
Hopea ponga		*	
Humboldtia brunonis	*		
Hydnocarpus alpina	*	*	
Hydnocarpus pentandra	*	*	*
Isonandra lanceolata		*	
Knema attenuata	*	*	*
Kydia calycina			*
Lagerstroemia lanceolata	*	*	*
Lannea coromandelica			*
Laportea crenulata	*	*	
Leea indica		*	
Lepisanthes deficiens	*	*	*
Ligustrum perottetti		*	*
Litsea floribunda	*	*	*
Litsea glabrata	*		
Litsea insignis	*		
Litsea mysorensis	*	*	*
Litsea oleoides	*	*	*
Litsea stocksii	*	*	
Lophopetalum wightianum	*	*	*
Macaranga peltata	*	*	*
Madhuca neriifolia		*	
Mallotus philippensis	*	*	*
Mallotus stenanthes	*		
Mangifera indica	*	*	*
Margaritaria indica	*	*	*
Mastixia arborea	*	*	*
Maytenus rothiana	*	*	
Memecylon malabaricum	*	*	
Memecylon talbotianum	*	*	*
Memecylon umbellatum		*	*
Memecylon wightii	*		
Mesua ferrea	*	*	*
Michelia champaca		*	*
Microtropis wallichiana	*	*	
Mimusops elengi	*	*	*

	Reserve	Sacred	Coffee
Mitrephora heyneana	*		
Mitragyna tubulosa			*
Myristica dactyloides	*	*	*
Neolitsea zeylanica	*	*	*
Nothopegia beddomei	*	*	*
Nothapodytes foetida	*	*	*
Ochna lanceolata	*		*
Olea dioica	*	*	*
Oroxylum indicum			*
Otonephelium stipulaceum	*	*	*
Pajanelia rheedii	*	*	*
Palaquium ellipticum	*	*	*
Pavetta sp.	*	*	*
Persea macrantha	*	*	*
Pittosporum dasycaulon		*	*
Polyalthia coffeoides	*		
Polyalthia fragrans			*
Pongamia pinnata		*	*
Premna tomentosa		*	*
Prunus ceilanica	*	*	*
Psidium guajava			*
Pterocarpus marsupium		*	*
Schefflera capitata		*	
Schefflera micrantha	*	*	*
Schleichera oleosa		*	*
Schefflera sp.		*	*
Schefflera wallichiana		*	
Scleropyrum pentandrum	*	*	*
Scolopia crenata	*	*	*
Spondias indica		*	
Spondias pinnata	*	*	*
Stereospermum chelonioides	*	*	*
Sterculia guttata	*	*	*
Stereospermum personatum		*	*
Streblus asper			*
Strombosia ceylanica	*		*
Symplocos macrophylla	*	*	*
Symplocos racemosa	*	*	*
Syzygium cumini	*	*	*
Syzygium gardnerii	*	*	*
Syzygium hemisphericum	*	*	*
Syzygium heyneanum	*	*	
Syzygium lanceolatum	*		
Syzygium mundagam	*	*	*
Syzygium munronii	*	*	
Syzygium phyllareoides		*	*
Syzygium rubicundum	*		*
Syzygium zeylanicum		*	
Tabernaemontana heyniana	*	*	*
Terminalia bellarica	*	*	*
Toona ciliata	*	*	*
Trema orientalis		*	*
Trichilia connaroides	*	*	*
Turpinia malabarica	*	*	*

	Reserve	Sacred	Coffee
Unidentified		*	
Vateria indica	*	*	*
Vepris bilocularis	*	*	*
Vernonia monosis		*	*
Viburnum punctatum	*	*	*
Villebrunea integrifolia	*	*	*
Vitex altissima		*	*
Xanthophyllum flavescens	*	*	*
Xanthoxylum rhetsa		*	
Xeromphis spinosa		*	*

APPENDIX 4

IUCN THREAT CATEGORIES (IUCN 2000)

Note: An asterisk indicates those categories used to define the threat status of trees and birds in the present study.

Extinct - A taxon is Extinct when there is no reasonable doubt that the last individual has died.

Extinct in the Wild - A taxon is Extinct in the Wild when it is known only to survive in cultivation, in captivity or as a naturalised population (or populations) well outside the past range. A taxon is presumed extinct in the wild when exhaustive surveys in known and/or expected habitat, at appropriate times (diurnal, seasonal, annual), throughout its historic range have failed to record any individual. Surveys should be over a time frame appropriate to the taxon's life cycle and life form.

Critically Endangered (*) - A taxon is Critically Endangered when it is facing an extremely high risk of extinction in the wild in the immediate future, as defined by any of the criteria (A to E) as described at the end of this appendix.

Endangered (*)- A taxon is Endangered when it is not Critically Endangered but is facing a very high risk of extinction in the wild in the near future, as defined by any of the criteria (A to E) as described at the end of this appendix.

Vulnerable (*) - A taxon is Vulnerable when it is not Critically Endangered or Endangered but is facing a high risk of extinction in the wild in the medium-term future, as defined by any of the criteria (A to E) as described at the end of this appendix.

Lower Risk - A taxon is Lower Risk when it has been evaluated, does not satisfy the criteria for any of the categories Critically Endangered, Endangered or Vulnerable. Taxa included in the Lower Risk category can be separated into three subcategories:

> **Conservation Dependent (*)** - Taxa which are the focus of a continuing taxon-specific or habitat-specific conservation programme targeted towards the taxon in question, the cessation of which would result in the taxon qualifying for one of the threatened categories above within a period of five years.
> **Near Threatened (*)** - Taxa which do not qualify for Conservation Dependent, but which are close to qualifying for Vulnerable.
> **Least Concern** - Taxa which do not qualify for Conservation Dependent or Near Threatened.

Data Deficient - A taxon is Data Deficient when there is inadequate information to make a direct, or indirect, assessment of its risk of extinction based on its distribution and/or population status. A taxon in this category may be well studied, and its biology well known, but appropriate data on abundance and/or distribution is lacking. Data Deficient is therefore not a category of threat or Lower Risk. Listing of taxa in this category indicates that more information is required and acknowledges the possibility that future research will show that threatened classification is appropriate. It is important to make positive use of whatever data are available. In many cases great care should be exercised in choosing between Data Deficient and threatened status. If the range of a taxon is suspected to be relatively circumscribed, if a considerable period of time has elapsed since the last record of the taxon, threatened status may well be justified.

Not Evaluated - A taxon is Not Evaluated when it is has not yet been assessed against the criteria.

Criteria for Critically Endangered, Endangered and Vulnerable taxa

A) Population reduction in the form of either of the following:

- *1) An observed, estimated, inferred or suspected reduction of at least 80, 50 and 20% respectively, over the last 10 years or three generations, whichever is the longer, based on (and specifying) any of the following:*
    - *a) Direct observation;*
    - *b) An index of abundance appropriate for the taxon;*
    - *c) A decline in area of occupancy, extent of occurrence and/or quality of habitat;*
    - *d) Actual or potential levels of exploitation;*
    - *e) The effects of introduced taxa, hybridisation, pathogens, pollutants, competitors or parasites.*
- *2) A reduction of at least 80, 50 and 20% respectively, projected or suspected to be met within the next 10 years or three generations, whichever is the longer, based on (and specifying) any of (b), (c), (d) or (e) above.*

B) Extent of occurrence estimated to be less than 100, 5000 and 20,000 km^2 *respectively, or area of occupancy estimated to be less than 10, 500 and 2000* km^2 *respectively, and estimates indicating any two of the following:*

- *1) Severely fragmented or known to exist at only a single location.*
- *2) Continuing decline, observed, inferred or projected, in any of the following:*
    - *a) Extent of occurrence;*
    - *b) Area of occupancy;*
    - *c) Area, extent and/or quality of habitat;*
    - *d) Number of locations or subpopulations;*
    - *e) Number of mature individuals.*
- *3) Extreme fluctuations in any of the following:*
    - *a) Extent of occurrence;*
    - *b) Area of occupancy;*
    - *c) Number of locations or subpopulations;*
    - *d) Number of mature individuals.*

C) Population estimated to number less than 250, 2500 and 10,000 mature individuals respectively, and either:

- *1) An estimated continuing decline of at least 25, 20 and 10% respectively, within three years or one generation, whichever is longer or*
- *2) A continuing decline, observed, projected, or inferred, in numbers of mature individuals and population structure in the form of either:*
    - *a) Severely fragmented (i.e. no subpopulation estimated to contain more than 50, 250 and 1000 mature individuals respectively);*

b) All individuals are in a single subpopulation.

*D) Population estimated to number less than 50 and 250 mature individuals for **Critically Endangered** and **Endangered** taxa respectively; and population very small or restricted in the form of either of the following for **Vulnerable** taxa:*

1) Population estimated to number less than 1000 mature individuals;

2) Population is characterised by an acute restriction in its area of occupancy (typically less than 100 km^2) or in the number of locations (typically less than five). Such a taxon would thus be prone to the effects of human activities (or stochastic events whose impact is increased by human activities) within a very short period of time in an unforeseeable future, and is thus capable of becoming Critically Endangered or even Extinct in a very short period.

E) Quantitative analysis showing the probability of extinction in the wild is at least 50, 20 and 10% respectively within 10, 20 and 100 years respectively or three generations, whichever is the longer.

APPENDIX 5

DISTRIBUTION OF BIRD SPECIES

Note: An asterisk indicates the occurrence of the species in the land use type.

Common name	Scientific name of species	Forest reserve	Sacred forests	Coffee plantations
Asian Brown Flycatcher	*Muscicapa dauurica*		*	
Asian Fairy Bluebird	*Irena puella*	*	*	*
Asian Koel	*Eudynamic scolopacea*		*	*
Alexandrine Parakeet	*Psittacula eupatria*		*	
Asian Paradise Flycatcher	*Terpsiphone paradisi*		*	*
Ashy Woodswallow	*Artamus fuscus*			*
Banded Bay Cuckoo	*Cacomantis sonneratii*		*	
Black-crested Bulbul	*Pycnonotus melanicterus*	*	*	*
Brown-cheeked Fulvetta	*Alcippe poioicephala*	*	*	*
Black-hooded Oriole	*Oriolus xanthornus*			*
Black Bulbul	*Hypsipetes leucocephalus*		*	*
Black Drongo	*Dicrurus macrocerus*	*	*	*
Black Eagle	*Ictinaetus malayensis*			*
Black-naped Monarch	*Hypothymis azurea*	*	*	*
Black-rumped Flameback	*Dinopium benghalense*		*	*
Blue-capped Rock Thrush	*Monticola cinclorhynchus*	*		*
Black-shouldered Kite	*Elanus caeruleus*		*	
Blue-winged Leafbird	*Chloropsis cochinchinensis*		*	*
Baya Weaver	*Ploceus philippinus*		*	*
Chestnut-bellied Nuthatch	*Sitta castanea*		*	*
Crimson-backed Sunbird	*Nectarinia minima*	*	*	*
Common Chiffchaff	*Phylloscopus collybita*		*	*
Crimson-fronted Barbet	*Megalaima rubricapilla*	*	*	*
Chestnut-headed Bee-eater	*Merops leschenaulti*		*	*
Common Flameback	*Dinopium javanense*	*	*	*
Common Iora	*Aegithina tiphia*	*	*	*
Common Myna	*Acridotheres tristis*		*	*
Common Tailorbird	*Orthotomus sutoris*			*
Common Woodshrike	*Tephrodornis pondicerianus*		*	*
Crested Serpant Eagle	*Spilornis cheela*	*	*	*
Chestnut-shouldered Petronia	*Petronia xanthocollis*		*	
Chestnut-tailed Starling	*Sturnus Malabaricus*		*	*
Dark Fronted Babbler	*Rhopocichla atriceps*	*	*	
Eurasian Golden Oriole	*Oriolus oriolus*	*	*	*
Emerald Dove	*Chalcophaps indica*		*	
Green Bee-eater	*Merops orientalis*		*	*
Grey-headed Canary Flycatcher	*Culicicapa ceylonensis*	*	*	*
Gold-fronted Leafbird	*Chloropsis aurifrons*	*		*
Greater Coucal	*Centropus sinensis*		*	*

		Reserve	**Sacred**	**Coffee**
Greater Racket-tailed Drongo	*Dicrurus paradiseus*	*	*	*
Greater Flameback	*Chrysocolaptes lucidus*	*	*	*
Hill Myna	*Gracula religiosa*	*	*	*
House Sparrow	*Passer domesticus*		*	
House Crow	*Corvus splendens*		*	
House Swift	*Apus affinis*	*	*	*
Heart-spotted Woodpecker	*Hemicircus canente*		*	*
Indian Scimitar Babbler	*Pomatorhinus horsfieldii*	*	*	*
Jungle Myna	*Acridotheres fuscus*		*	*
Jungle Prinia	*Prinia sylvatica*	*	*	*
Large-billed Crow	*Corvus macrorhyncus*	*	*	*
Loten's Sunbird	*Nectarinia lotenia*		*	*
Long-tailed Shrike	*Lanius schach*			*
Malabar Grey Hornbill	*Ocyceros griseus*	*	*	*
Mountain Imperial Pigeon	*Ducula badia*	*	*	*
Malabar Parakeet	*Psittacula columboides*	*	*	*
Malabar Trogon	*Harpactes fasciatus*	*	*	
Malabar Whistling Thrush	*Myophonus horsfieldii*	*	*	*
Nilgiri Flycatcher	*Eumyias albicaudata*		*	*
Nilgiri Wood Pigeon	*Columba elphinstonii*	*	*	*
Orange-headed Thrush	*Zoothera citrina*	*		
Oriental Magpie Robin	*Copsychus saularis*		*	*
Oriental White-eye	*Zosterops palpebrosus*			*
Pale-billed flowerpecker	*Dicaeum erythrorhyncus*	*	*	*
Plum-headed Parakeet	*Psittacula cyanocephala*	*	*	*
Purple-rumped Sunbird	*Nectarinia zeylonica*		*	
Puff-throated Babbler	*Pellornium ruficeps*	*	*	*
Purple Sunbird	*Nectarinia asiatica*		*	
Rose-ringed Parakeet	*Psittacula krameri*			*
Red-vented Bulbul	*Pycnonotus cafer*			*
Red-whiskered Bulbul	*Pycnonotus jocosus*		*	*
Scarlet Minivet	*Pericrocotus flammeus*	*	*	*
Shikra	*Accipiter badius*	*	*	*
Small Minivet	*Pericrocotus cinnamomeus*			*
Spotted Dove	*Streptopelia chinensis*		*	*
Tickell's Blue Flycatcher	*Cyornis tickelliae*	*	*	*
Velvet-fronted Nuthatch	*Sitta frontalis*		*	*
Vernal Hanging Parrot	*Loriculus vernalis*		*	*
White-browed Fantail	*Rhipidura auriola*		*	
White-bellied Treepie	*Dendrocitta leucogastra*	*	*	*
White-bellied Woodpecker	*Dryocopus javensis*	*	*	*
White-cheeked Barbet	*Megalaima viridis*	*	*	*
White-eyed Buzzard	*Butastur Teesa*	*	*	
White-throated Kingfisher	*Halcyon smyrnensis*		*	*
White-browed Wagtail	*Motacilla maderaspatensis*		*	
Yellow-browed Bulbul	*Iole indica*	*	*	*
Yellow-footed Green Pigeon	*Treron phoenicoptera*		*	*

APPENDIX 6

DISTRIBUTION OF MACROFUNGAL MORPHO-SPECIES

Note: An asterisk indicates the occurrence of the species in the land use type.

Name of morpho-species	Forest reserve	Sacred forests	Coffee plantations
Agaricus sp. (brown)		*	*
Agaricus sp.	*	*	*
Amanita sp.		*	
Ascomycetes (yellow-coloured, ball-like sporocarps)	*	*	*
Ascomycetes (elephant dung)	*	*	*
Ascomycetes (spoon)		*	
Ascomycetes (thread)		*	
Astraeus sp.		*	
Auricularia sp.	*	*	*
Boletus sp.		*	*
Callocybe sp.	*	*	
Unknown (Chilli-red)	*		
Clavaria sp. (brown)		*	
Clavaria sp.	*	*	*
Clavaria sp. (orange)	*	*	*
Clavaria sp. (purple)		*	
Clavaria sp. (*Ramaria* like)	*		
Clavaria sp. (tree)	*	*	
Clavaria sp. (white)	*	*	
Clitocybe sp.	*	*	*
Collybia sp. (brown in colour)	*	*	*
Collybia sp. (*Hygrophorus* like)		*	*
Collybia sp. (unknown)	*	*	*
Coprinus sp.	*	*	*
Cordyceps sp. (club)		*	
Cordyceps sp. (orange)	*	*	
Cordyceps sp. (tree)		*	
Coriolus sp. (*Foemes* like)		*	
Coriolus sp. (hirsute)	*	*	*
Coriolus sp.		*	
Cortinarius sp.	*	*	*
Cortinarius sp. (brown)			*
Cortinarius sp. (cyboid)	*	*	
Cortinarius sp. (*Hebeloma* like)		*	
Cortinarius sp. (*Inocybe* like)		*	
Cortinarius sp. (rusty)		*	
Crepidotus sp.	*	*	*
Daedalopsis flavida	*	*	*
Daldinia sp.	*	*	*
Dictyophora sp.		*	
Entolomataceae		*	*
Entolomataceae (pink)		*	

	Reserve	Sacred	Coffee
Entolomataceae (white)		*	
Ganoderma sp. (big fruiting body)	*	*	*
Ganoderma sp. (black)	*	*	*
Ganoderma sp. (brown)	*	*	*
Ganoderma sp. (orange coloured)	*	*	*
Ganoderma sp. (pink)	*		*
Ganoderma sp. (velvet)		*	*
Ganoderma sp. (white)	*		
Geastrum sp.	*	*	*
Hexagonia sp.	*	*	
Hygrocybe sp.	*	*	
Hygrophorus citratus		*	
Hygrophorus sp.	*	*	*
Hygrophorus sp. (red)		*	
Leotia bulgaria		*	*
Leotia sp. (*Peziza* like)		*	
Leotia sp. (brown)	*	*	*
Leotia sp. (orange)	*	*	*
Lycoperdon sp.	*	*	*
Marasmius sp. (bells)		*	
Marasmius sp. (black)		*	
Marasmius sp.	*	*	*
Marasmius sp. (oyster)	*	*	*
Marasmius sp. (small)		*	
Marasmius sp. (white)	*		
Microporus sp. (black)	*	*	
Microporus sp. (brown)	*	*	*
Microporus sp. (black and white)			*
Microporus sp. (brick)	*	*	*
Microporus sp. (orange)	*		*
Microporus sp.	*	*	*
Microporus sp. (oyster)			*
Microporus sp. (pink)	*	*	
Microporus sp. (spoon-like)	*	*	*
Microporus sp. (trumpet-shaped)	*	*	*
Microporus sp. (velvet)	*	*	
Microporus sp. (white)	*		
Morchella sp.	*		*
Mycena sp. (brown)	*	*	
Mycena sp. (leaf)	*	*	*
Mycena sp. (rufous)	*	*	*
Mycena sp.	*	*	*
Mycena sp. (white)	*		
Myxomyceteae	*	*	*
Nidularia sp. (*Cyathus* like)		*	
Otidia sp. (black)			*
Otidia sp. (*Marasmius* like)	*	*	*
Otidia sp. (orange)		*	
Otidia sp. (*Scutellinia* like)		*	
Peziza sp. (disk)	*	*	
Phallus sp. (*Mutinus* like)		*	
Phallus sp. (stalk)		*	
Phellinus sp.	*	*	*
Physarium sp.		*	

	Reserve	Sacred	Coffee
Physarium sp. (white)		*	
Pleurotus sp. (orange)		*	
Pleurotus sp. (oyster)			*
Pleurotus sp.	*	*	*
Polyporus sp. (black)			*
Polyporus sp. (brown)	*	*	*
Polyporus sp. (button)	*	*	*
Polyporus sp. (club)		*	
Polyporus sp. (*Coriolus* like)	*	*	*
Polyporus sp. (grey)	*	*	*
Polyporus sp. (*Hexagonia* like)	*	*	*
Polyporus sp. (orange)	*	*	*
Polyporus sp. (oyster)		*	
Polyporus sp. (pink)	*	*	
Polyporus sp. (polyporous)	*	*	*
Polyporus sp. (resupinate)	*	*	*
Polyporus sp. (rusty)	*	*	*
Polyporus sp. (scars)		*	
Polyporus sp. (spoon)	*		
Polyporus sp.	*		
Polyporus sp. (sulphur)	*	*	
Polyporus sp. (trumpet)	*	*	*
Polyporus sp. (tubes)	*	*	*
Polyporus sp. (velvet)		*	*
Polyporus sp. (white)	*		*
Rhizopogon sp.	*	*	
Russula sp. (brown)		*	
Russula sp. (pink)		*	
Russula sp. (purple)		*	
Russula sp. (red)		*	
Sarcocypha sp. (orange)	*	*	
Sarcocypha sp.	*	*	*
Schizophylum sp.	*	*	*
Tephrocybe sp.		*	
Termitomyces sp.	*	*	
Tremella sp. (jelly)	*	*	*
Tremella sp. (orange)	*	*	*
Tricholomataceae (*Armillaria* like)		*	*
Tricholomataceae (black)		*	
Tricholomataceae (brown)	*	*	*
Tricholomataceae (campanulate)		*	
Tricholomataceae (gills)		*	
Tricholomataceae (grey)	*		
Tricholomataceae (huge)		*	
Tricholomataceae (large, edible)	*	*	*
Tricholomataceae	*	*	*
Tricholomataceae (mucilage and brown)	*	*	*
Tricholomataceae (orange)		*	*
Tricholomataceae (*Oudemansiella* like)		*	
Tricholomataceae (oyster)	*		*
Tricholomataceae (pink)	*		
Tricholomataceae (purple)	*	*	*

	Reserve	Sacred	Coffee
Tricholomataceae (red)			*
Tricholomataceae (ring)		*	
Tricholomataceae (small, edible)		*	*
Tricholomataceae (silver)		*	
Tricholomataceae (trumpet)	*	*	*
Tricholomataceae (white)	*	*	*
Xylaria sp.		*	*
Xylaria sp. (balls)	*	*	
Xylaria sp. (carrot)		*	
Xylaria sp. (club-shaped)	*	*	*
Xylaria sp. (long)		*	
Xylaria sp. (monstrous)		*	
Xylaria sp. (rod-like)	*	*	*
Xylaria sp. (short)	*	*	*
Xylaria sp. (thin)	*	*	*

APPENDIX 7

PARKS AND FACTORS IN THEIR SUCCESS

This is a letter, published in the Science Magazine, written in response to:
Bruner A.G., Gullison R.E., Rice R.E. & da Fonseca G.A.B. (2001) Effectiveness of parks in protecting tropical biodiversity. *<u>Science</u>, 291, 125-128.*

Bruner and co-authors conclude that most tropical nature reserves are effectively protecting biodiversity. However, we find several shortcomings in their study.

First, a park does not prevent exploitation and degradation of ecosystems, but merely displaces them. The presence of a park might increase pressure on the surrounding area as a source of land and resources (1). Hence, Bruner *et al.*'s comparison of a park and its surrounding 10-kilometer zone is flawed. A more meaningful comparison would be with a similar (climate, topography, geology) landscape far enough away from the park to escape from its negative impacts on resource use.

Second, parks are located in areas that are typically in better condition than the surrounding areas (2). Any differences between parks and their surrounding areas might reflect pre-existing differences in habitat quality.

Third, Bruner *et al.*'s study is entirely based on questionnaire surveys. Of the total respondents, 70% are directly involved in park management and therefore have a vested interest in promoting its effectiveness. A similar questionnaire survey by the International Union for Conservation of Nature and Natural Resources (IUCN) of local experts' views on the status of forest protected areas found that only 1% were regarded as secure and many were suffering serious degradation and loss (3). Rigorous fieldwork is essential to cross-check the accuracy of information such as this.

Protected areas and community management are not alternatives; rather, they are complementary (4). Bruner and colleagues maintain that parks are better for habitat integrity than no protection; however, parks should still be compared with community-based conservation systems to determine the conditions under which strict protection is preferable and those where other IUCN categories (5) or community-based conservation approaches would work better.

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References and Notes

1. See, for example, H. Raven, *Science* **280**, 1507 (1998).
2. J. MacKinnon, K. MacKinnon, G. Child, J. Thorsell, Eds., *Managing Protected Areas in the Tropics* (IUCN, Gland, Switzerland, 1986), p. 34.
3. "Threats to forest protected areas" (report from IUCN The World Conservation Union for the World Bank/WWF alliance for forest conservation and sustainable use, IUCN, 1999), also available at http://www-esd.worldbank.org/wwf/paperreport.pdf
4. A. R. E. Sinclair, D. Ludwig, C. W. Clark, *Science* **289**, 1875 (2000).
5. M. Hockings, S. Stolton, N. Dudley, A. Phillips, *Evaluating Effectiveness: A Framework for Assessing the Management of Protected Areas* (World Commission on Protected Areas, IUCN, Cardiff University, Wales, UK, 2000).