EFFECT OF FOREST FRAGMENTATION ON VASCULAR PLANT DIVERSITY IN KHANGCHENDZONGA BIOSPHERE RESERVE, SIKKIM WITH EMPHASIS ON REGENERATION OF SOME IMPORTANT TAXA

BY

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Contents

Chapter 1
General Introduction

The global environment has been changing and the living organisms have adapted continuously to these changes. Natural landscapes have constantly changed and the new habitats have been created. This process has resulted in evolution of new species and extinction of many others. Biodiversity represents the variability in nature and relates to the differences within and between species and their surroundings i.e. ecosystems. The UN Convention on Biological Diversity (CBD 1992) defines biodiversity as "The variability among living organisms from all sources including, *inter alia*, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are a part; this includes diversity within species, between species and of ecosystems". Essentially, there are three levels of biodiversity that come from the definition of CBD viz. 1) diversity between and within ecosystems and habitats; 2) diversity of species and 3) genetic variation within individual species.

The diversity of species within a habitat is measured and expressed as α-diversity. β-diversity measures the rate of replacement of species along a gradient of habitats or communities. The changing conditions within a habitat can change the diversity of species within the habitat, and vice versa. So, by monitoring the numbers and types of species present, it is possible to determine whether any adverse changes, beyond those of natural variability, are occurring. Human induced environmental changes are occurring at a rapidly increasing pace and have severely impacted species diversity and composition. The conservation of rare species is guided by the biological attributes of the taxon. However, lack of basic biological data for many plant species has led to the failure of many recovery plans (Pavlik 1994; Shemske et al. 1994; Schultz & Gerber 2002).

India, with a geographical area of 2.4 per cent of the world, has about 8 percent of the world's total biodiversity. The country is very rich in biodiversity with 45,000 plant

and 75,000 animal species (Lal 1995). Hence, it is called as a mega-diversity country. Of the reported 45,000 plant species of India, 11 per cent are endemic. The country has been divided into a number of biogeographic zones based on biodiversity value and environmental realms (Rodgers & Panwar 1988). Three biodiversity hotspots viz., Himalayas, Indo-Burma and Western Ghats are located in India (Mittermeier et al. 2005). Forest is the most important natural resource of north-east India fulfilling the diverse requirements of human populations. Over-exploitation of forest produces has caused serious damage to natural forest ecosystems and rich biodiversity of the region. Loss of diversity in plant communities limits plant recruitment processes and decreases the ecosystem productivity, thereby affecting the overall ecosystem functioning (Symstad & Tilman 2001).

Determining the mechanisms that maintain the species diversity of different plant growth forms such as trees, lianas, shrubs, herbs and epiphytes is essential to our understanding of the maintenance of species diversity in different forests (Schnitzer & Carson 2000). Lianas and epiphytes are two distinct groups of life form that contribute to the bulk of forest plant diversity. Lianas are woody climbing plants and are often a large component of the canopy in tropical forests. It is often one-third or more of the total leaf area but only a small component of basal area and biomass (Schnitzer & Bongers 2002). In tropics woody climbers make their most important contributions to the physiognomy, productivity and species richness of forests (Putz 1984; Schnitzer & Bongers 2002; Phillips et al. 2005). For example, lianas compete with trees for both above and below-ground resources, substantially decreasing the growth rates and fecundity of adult trees, retarding regeneration of tree seedlings and saplings, and increasing the number of trees damaged and killed in treefalls (Stevens 1987; Schnitzer et al. 2005; Pérez-Salicrup et al. 2001; Kainer et al. 2006). Lianas can also have decidedly positive effects on forests, providing valuable food resources, habitat, and connections among tree canopies that are

used as pathways by arboreal animals (Emmons & Gentry 1983; Ødegaard 2000). Lianas also play a role at the ecosystem level by contributing to the carbon budget of tropical forests, representing as much as 10% of fresh aboveground biomass (Putz 1984); however, when lianas become abundant they may displace trees and actually reduce the ability of forests to sequester carbon (Laurance et al. 2001; Phillips et al. 2002). Determining the abundance and dynamics of lianas in tropical forests is particularly timely because lianas appear to be increasing in abundance, possibly due to global climate change (Phillips et al. 2002; Wright et al. 2004). However their importance decreases with elevation, the average percentage of lianas in woody floras falling to about 10 % in temperate forests (Gentry 1991). This has been attributed to poor compatibility of liana life-form with cold climates (Gentry 1991).

Lianas differ from other structural parasites such as epiphytes and hemiepiphytes because they remain rooted to the ground throughout their lives (Putz & Mooney 1991). Epiphytes on the other hand accounts for about 10% of global vascular plant diversity (Kress 1986). The epicentre for vascular epiphyte diversity is the neotropics (Madison 1977; Benzing 1990) which is also home to the largest proportion of the world's plant species (Gentry 1982b; Henderson et al. 1991; Phillips et al. 1994; Myers et al. 2000). The vast majority of neotropical vascular epiphyte species are concentrated in montane forests (Gentry & Dodson 1987) where the epiphyte component can represent up to 30% (Gentry and Dodson 1987) or 50% (Kelly et al. 1994; Bussmann 2001) of the total vascular plant flora.

Tropical montane forests are characterised by a cool and humid atmosphere. This appears to contribute to the growth of epiphytic plants in the canopy at high densities. Epiphyte communities play an influential role in montane forest ecosystem processes by contributing stripped rain water and nutrients that would otherwise remain unavailable to the forest (Nadkarni 1986; Coxson 1991; Clark et al. 1998; Coxson et al. 1992; Coxson &

Nadkarni 1995). Epiphyte communities are important habitat for insects (Floater 1995) and amphibians (Giaretta et al. 1999; Pounds et al. 1999; Pounds 2000) and are food source for birds (Nadkarni & Matelson 1989) and canopy mammals. Epiphyte communities can also directly benefit host trees, as evidenced in some tropical species by the evolution of adventitious canopy roots to harness the nutrient pool held by epiphyte communities upon their upper branches (Nadkarni 1981, 1994).

Forest fragmentation is one of the most important factors threatening biodiversity of natural ecosystems of the world. Forest fragmentation is a process that leads to conversion of continuous forests into fragments of forest separated by non-forested lands. The process of forest fragmentation has been increasing alarmingly throughout the world, especially in tropical forests that has the bulk of biodiversity and, hence a major concern for the conservationists. Forest fragmentation is a dynamic process in which the habitat is progressively reduced into smaller fragments that become more isolated and increasingly affected by edge effects (Forman & Godron 1986; Reed et al. 1996; Franklin 2001; McGarigal 2002).

The forest fragmentation can be explained in two phases. The first phase results in the reduction of total amount of forest areas whereas the second phase leads to the isolation of smaller patches (Wilcove & Mclellan 1986; Saunders et al. 1991). There are many physical and biological changes associated with forest fragmentation, such as habitat loss and insularization (Lovejoy et al. 1986; Laurance 1990).

Some of the important consequences are reduction in the number of species, interference in dispersal and migration processes, altered ecosystem inputs and outputs, and exposure of isolated core habitats of the forest. Forests may be fragmented by a number of activities or events, such as road construction, logging, conversion to agriculture, or wildfire, but ultimately, the fragmenting cause is either anthropogenic or natural in origin (Wade et al. 2003). All these mechanisms are responsible for the

progressive erosion of biodiversity (Terborgh & Winter 1980; Tilman & Downing 1994). The environment of the fragments becomes conducive for weedy/exotic species. In some cases, the weedy species are incorporated into the remaining plant community and are responsible for the elimination of the species confined to the forest interior (Janzen 1986). Fragmentation affects seedling populations in forest communities through increased tree mortality (Williams-Linera 1990; Laurance 1991) and altered plant-animal interactions (Benitez-Malvido 1998). Fragmentation reduces animal-mediated seed dispersal and therefore, regeneration of plants that depend on animals to carry their seeds is seriously affected. As a result isolated patches are not colonized by many plant species that could potentially live there (Primack 1992, 1993). The survival of the saplings is more affected by fragmentation than that of adult trees, which are less sensitive to changes in environmental conditions (Gibson et al. 1998). Marcelo et al. (2004) observed that forest fragmentation increases tree sapling mortality by accelerating competition with lianas, vines and ruderal species. Lianas are more associated with fragmented forests (Laurance & Cochrane 2001) than regenerating forests (Dewalt et al. 2000; nabe-Nielsen 2002).

Remote sensing and GIS tools have been successfully employed to monitor the fragmented ecosystems. Various satellite sensors with different spatial resolution have been utilized in the study of forest fragmentation. Remote sensing so far is the only feasible way to map forest fragmentation at regional and global scales (Lambin & Ehrlich 1997). The conventional methods of biodiversity assessment mainly focus on species richness, abundance and similarity (Beals 1985). Of late, remote sensing and GIS tools are being used to assess biodiversity at landscape level (Fuller et al. 1998; Nagendra & Gadgil 1999; Roy & Tomar 2000).

An understanding of regeneration process that ensures maintenance of community structure and ecosystem stability is essential for the development and management of mixed plantations, uneven-aged stands as well as natural forests (Moravie et al. 1997). In

natural forest, regeneration occurs both by vegetative and reproductive means. Regeneration by vegetative mode can be either through sprouting of stems and roots or through vegetative propagules. The regeneration through reproductive means involves several phases such as seed production, dissemination, germination and survival of seedlings, each phase has got a bottleneck that restricts reproduction of a particular species (Jones et al. 1994). Many lianas propagate vegetatively as well as by seeds (Putz 1984), enhancing their ability to proliferate under favourable conditions. Most lianas are light loving and respond positively to forest fragmentation (Webb 1958; Putz 1984). Regeneration through sexual reproductive means depends on the availability of viable seeds. In the tropical rainforest the majority of main canopy trees may flower annually, or biannually, but the seed is not always set. Most of them produce good seed in fair quantity at least once in three years (Barik et al. 1996). Therefore, the quantity and frequency of seed production differ in different tree species (Richards 1996). The seeds of tropical-rain forest trees show tremendous variation in size, mass and germination requirements. The dispersal mechanism also influences their germination and seedling establishment. Some of them may get dispersed by mammals, some by birds and some by wind. With the help of these dispersal agents the seeds are able to find a site suitable for their germination instead of falling below the parent plants where they have to compete with the later for both below and above ground resources.

The seeds of tropical-rain forest trees show large inter-specific variation in the time, which they take to germinate. Seeds of some species germinate after few weeks from sowing, while others may take more than 20 weeks. A number of factors may cause delayed germination. These include low water content of seed at maturity, presence of hard seed coat, small size, early stage of development of the embryo and the presence of chemical inhibitors. Besides, environmental conditions may also induce dormancy; shade

appears to be the most important inducer of dormancy in the seeds of tropical trees (Turner 2001).

A large proportion of seeds reaching the ground is destroyed by insects or small mammals or become infected by fungi. The chances of survival of seeds on the forest floor are increased if they get buried into the soil. (Richards 1996). In spite of enormous losses and an array of internal and external factors which influence germination, a heavy seed fall may result in an abundant crop of seedlings. Their survival on the forest floor is influenced by interaction of various abiotic and biotic factors, which in turn restrict the regeneration of the species (Jones et al. 1994). Natural or man-induced disturbances such as gap formation, herbivory, landslides, and logging also affect abundance and composition of seedlings in the forest under-storey (Benitze-Malvido 1998). Differences in dispersal mechanisms and physiological tolerances of seeds and seedlings bring about spatial differentiation of regeneration niches of species. The aggregation of seedlings, therefore, is determined by microsites distribution on the forest floor (Barker & Kirkpatrick 1994). Thus an examination of the fate of seeds and study of seedling growth and their population dynamics on the forest floor are helpful in interpreting regeneration strategies of forest tree and liana species.

The Eastern Himalaya with more than 3000 endemic species is one of the 34 biodiversity hotspots of the world and spreads over an area of 1500 km^2 in Sikkim, West Bengal, Arunachal Pradesh and Nagaland. The region spans a wide spectrum of ecological zones and contains parts of three global Biodiversity Hotspots. The five countries traversed by the Eastern Himalayas viz., Bhutan, China, India, Myanmar and Nepal have very different geo-political and socioeconomic systems, and contain diverse cultures and ethnic groups. The region is the meeting place of three realms, namely, the Indo-Malayan, Palearctic, and Sino-Japanese. The meeting of these realms has created one of the most biologically rich areas on Earth. The region's complex topography and

wide elevational gradients i.e. from floodplain to more than 8000 m in high mountains have contributed to the highly varied vegetation patterns. The complex mountain topography has created diverse bioclimatic zones such as tropical, subtropical, lower temperate, upper temperate, subalpine evergreen, alpine evergreen, and alpine shrubs and meadows and 'island-like' conditions for many species and populations, making them reproductively isolated. This isolation has given rise to genetic differences among populations, thereby contributing to the exceptionally rich genetic, species and ecosystem diversity of the region. This area has been in the spotlight as it contains Crisis Ecoregions, Biodiversity Hotspots, Endemic Bird Areas, Mega Diversity Countries, and Global Ecoregions (Brooks et al. 2006).

The rich biodiversity of Eastern Himalayas has been under threat due to increasing biotic pressure. The traditional protection measures adopted by the government have not been adequate to withstand the anthropogenic/biotic pressures such as grazing and forest fire. The forest ecosystems of the Eastern Himalaya have been affected by various human activities posing serious threats to the existence of several taxonomically and Ethnomedicinally important plant species.

A large portion of the plant diversity of Himalayas, particularly lesser known groups such as lianas and epiphytes, remained unexplored. In general, our understanding of forest fragmentation pattern and its impact on Himalayan plant diversity is poor. One phenomenon that has received very little attention is the regeneration of woody plants in the forest and its relationship with the forest fragments. The present thesis embodies the works carried out in Khangchendzonga Biosphere Reserve (KBR) over a period of five years to bridge the above mentioned knowledge gap. The study has following objectives:

1) To prepare an inventory of plant species with special emphasis on lianas and epiphytes.
2) To study causes and pattern of forest fragmentation in KBR.

3) To study regeneration ecology of a few taxonomically and ethnomedicinally important tree and liana species.

This dissertation has been divided into 8 chapters. This introduction chapter is followed by a brief review of literature and description of study sites. Chapter 4 analyzes the plant diversity and attempts to relate diversity with various forest microenvironmental factors. The pattern of fragmentation has been analyzed in chapter 5 and the impact of fragmentation on tree diversity has been presented in chapter 6. Regeneration ecology of four ethnobotanically important plants has been described in chapter 7 and impact of fragmentation on regeneration process has also been analyzed in this chapter. Chapter 8 discusses the results of all the chapters and attempts to bring out a synthesis on the impact of fragmentation on plant diversity in KBR.

Chapter 2
Review of Literature

Studies on the effects of habitat fragmentation on biodiversity are diverse, and workers have measured fragmentation in different ways. As a consequence, conclusions on magnitude and direction of its effects varied (Fahrig 2003). A substantial portion of world's biodiversity is found in tropical rain forests. The current rate of tropical deforestation i.e. about 15.4 million ha per year suggests that a majority of the world's remaining rain forests will be fragmented into areas of < 100 km^2 within the next 30 years (Whitmore 1997). Increased forest fragmentation poses a great threat to the biodiversity, increases edge effect and reduces interior habitat, which alters the region's biota as a whole. Apart from anthropogenic factors, non-anthropogenic causes are also important factors affecting understorey diversity in forested landscapes (Huebner & Randolph 1995). Biological Dynamics of Forest Fragmentation Project (BDFFP) is the largest running experimental study of forest fragmentation in the world (Debinski & Holt 2000). Experiments have revealed that the diversity of edge effects in fragmented rainforest affects the plant communities and various ecosystem processes. Laurance et al. (1998) concluded that fragmentation causes important changes in the dynamics of Amazonian forest, especially within~100 m of habitat edges. Vellend (2003) through his study at ancient and recent forests from 10 regions of Europe and eastern America assessed that habitat loss inhibits recovery of plant diversity as forest regrows. In a recent study based on a review of 17 empirical studies ranging from small-scale experimental studies to continental-scale analyses, Fahrig (2003) pointed out that the effects of fragmentation on diversity were ambiguous and could be positive or negative. As a result of fragmentation, it was observed that the average population size of forest species is on the decline and they face danger of accelerated rate of exploitation (MacArthur & Wilson 1967).

Fragmentation threatens species in different ways, depending on species-specific characteristics, life stages and the type of environment. Many species are specialized according to the microclimatic conditions of the forest, and such species are affected most by fragmentation. Since no suitable habitat is available for them as continuous forests are fragmented, these species often face the threat of extinction. Economically and commercially important species undergo higher degree of poaching and extraction, e.g. for food, fuel, timber and medicinal uses. Many forest fragments are readily accessible to humans due to high edge-interior ratios (Goparaju et al. 2005).

Satellite imageries were used as a tool for monitoring diversity richness and such information gathered, can be used to organize a programme of monitoring biodiversity (Nagendra & Gadgil 1999). Kushwaha et al. (2005) applied geospatial modelling approach for the assessment of plant richness in Barsey Rhododendron Sanctuary in Sikkim, which is very close to KBR. They noted that assessment of plant richness at ecosystem level presents a more realistic picture than at landscape level. The study demonstrated that remote sensing data coupled with landscape analysis, ground inventory data and geospatial modelling hold good potential for rapid and operational assessment of plant richness. In India, very few studies on fragmentation and anthropogenic disturbances in Himalayan forests have been conducted using remote sensing and GIS (Palni et al. 2000). Some of the workers observed that human dependency on natural vegetation appears to be the main cause of forest fragmentation. A study on impact of forest fragmentation on phytodiversity by Goparaju et al. (2005) in Vindhyan highlands concluded that the community structures are completely different in small and large fragments. It was observed that the three levels of biodiversity operate differently with changing fragment size classes. Recently, Page et al. (2009) studied the effect of forest fragmentation on different plant lifeforms in Western Ghat, India and concluded that different life forms respond differentially to the degree of fragmentation.

Liana typically constitutes about 25% of the woody stem density and species diversity in many tropical forests (Gentry 1991). Liana abundance and diversity, however, can be quite variable among different forests. In an analysis of 32 moist, wet and pluvial neotropical forests, Gentry (1991) reported high variation in liana abundance among forest and soil types. He found no strong trend in liana abundance with soil fertility. Kusumoto et al. (2008) studied the diversity of lianas in subtropical forest of Okinawa in south-western Japan, and Cai et al. (2009) in subtropical forests of Xishuangbanna, South-west China. Campanello et al. (2007) reported high diversity of lianas in lower montane (3010 stems ha^{-1}), followed by montane (2760 stems ha^{-1}) and subtropical Atlantic (1237 stems ha^{-1}) forest. Pioneer studies in lianas in India were done by Chittibabu and Parthasarathy (2001) in Eastern Ghat, Reddy and Parthasarathy (2003) in Coromandel Coast and Parthasarathy et al. (2004) in peninsular India. However, the lianas in the Eastern Himalayas have never been studied.

Epiphytic flora of the Himalayas was first studied by Schimper (1888). The species of *Ribes*, *Euonymus*, *Thalictrum*, *Rhododendron* and epiphytic orchids were widely represented in the flora. The epiphytic orchids were by and large represented by large tropical genera such as *Bulbophyllum*, *Coelogynae*, *Dendrobium*, *Eria* and *Oberonia* (Mehra & Vij 1974; Hajra 1996). The composition of the epiphytic lichen flora is strongly influenced by the vertical variation in microclimate, which in turn is determined by the interactions between regional climate, tree architecture and bark properties, such as chemistry, texture and moisture holding capacity (Halonen et al. 1991; Campbell & Coxson 2001; Lowman & Rinker 2004). Epiphytes tend to occur in different strata of the forest. A differential vertical distribution and partitioning of the available space on the phorophyte is commonly observed in closed canopy forests (Johansson 1974; Sanford 1974; Kelly 1985; Brown 1990; Freiberg 1996) in which micro-environmental conditions vary markedly from canopy to understory (Kelly 1985).

In tropics, study on woody flora regeneration was carried out by Whitmore (1984), Richards (1996) and Turner (2001). However, only a limited number of studies have been carried out to understand the impact of fragmentation on seed germination, predation and dispersal. Effect of fragmentation on seed dispersal, and regeneration dynamics in Kakamega forest, Kenya was studied by Bleur and Gaese-Bohning (2004). They concluded that Kakamega peripheral fragments lost several forest bird species, which in turn, altered dispersal of several plant species. Benitez-Malvido (1998) studied the impact of forest fragmentation on seedling abundance in a tropical rain forest and concluded that reduction in seedling density due to forest fragmentation may be a consequence of a complex interplay of factors of two kinds: (i) those that reduce seedling establishment rate within fragments and (ii) those that increase mortality of seedling within fragments.

Tree regeneration studies in the sub-tropical broad-leaved forests of north-east India have been carried out by Khan et al. (1986), Barik et al. (1992, 1996). Khan et al. (1986) studied the survival of seedlings and sprouts survival in three forest stands and observed that seedlings of *Quercus* spp., *Schima khasiana*, *Schima wallichii* and *Shorea robusta* showed 100% mortality in the dense stand, and only some sprouts could survive in the forest near the periphery. Barik et al. (1992) examined the role of tree fall gaps in maintaining composition and patchy distribution of tree species. Barik et al. (1996) studied the effect of disturbance on natural regeneration of *Schima khasiana, Lithocarpus dealbatus* and *Quercus griffithii*. They found an increase in seed production and germination of *Lithocarpus dealbatus* in a mildly disturbed stand and better performance of *Schima khasiana* in the highly disturbed stand.

Study of tree regeneration in forest is an important and challenging area of research in the field of tropical forest ecology. Studies carried out on this aspect in tropical moist forest have been synthesized and discussed by Whitmore (1984), Richards (1996) and Turner (2001). Recent studies have focused on the role of seed size, forest

microenvironment and disturbance on tree regeneration. The effect of seed mass on seedling emergence and seed removal by animals in 7 species of *Psychotria* in neo-tropical forest was studied by Paz et al. (1999). They found that the effect was both species and habitat-specific. Walters and Reich (2000) studied the effect of seed size, nitrogen supply and growth rate on seedling survival in deep shade of a cold-temperate forest. They found that the seedling survival was positively related to relative growth rate (RGR), but relationship between RGR and survival differed from species to species. Large-seeded, shade-tolerant species had higher survival than small-seeded, intolerant ones. Suresh et al. (2001) studied the influence of age of the tree on seed weight, germination and seedling quality in *Acacia nilotica*. They observed that seeds collected from trees more than 8 years old yields better quality seeds which exhibit better germination percentage. Khan and Uma Shankar (2001) studied the effect of seed weight, light regime and substratum quality on germination and seedling growth of *Quercus semiserrata*. They concluded that heavy seeds germinated early and achieve greater germination percentage than small seeds. Moss was a better substratum than litter or soil surfaces. The species is light-dependent for germination and heavy seeds result in greater seedling survival and dry mass production.

Besides seed size, forest microenvironment also plays an important role in regeneration of plant species by influencing germination and seedling establishment. Hyat and Casper (2000) studied the effect of vegetation on seed bank dynamics of tree species in a temperate deciduous forest. They observed that *Rubus allegheniensis*, *Phytolacca americana* and *Paulownia tomentosa* dominated the seed bank, and the presence of *Rubus* reduced the seed input of *Phytolacca* and increased its seed mortality. Coomes and Grubb (2000) have demonstrated how root competition may influence regeneration. Cater and Chapin (2000) determined the relative importance of competition and microenvironmental changes through which understorey vegetation influences the

establishment of woody plant seedlings. They found that understorey vegetation competed with tree seedlings and their establishment was different in different fragment types due to difference in the micro-environment. Kitzberger et al. (2000) investigated the effects of nurse shrubs and water availability on tree seedling emergence and survival and observed that shrubs were favourable for tree seedling establishment by providing protection from direct sunlight. Myster and Everham (1999) studied the germination requirements of rain forest trees and found that most trees had specific microsites where they grow well, while some were independent of microsites. Rey and Alcantara (2000) studied seedling establishment of *Olea europaea* shrub and found that water stress was responsible for 70% seedling loss. An interactive effect of temperature and light on tree seedling establishment in frost-prone areas was investigated by Egerton et al. (2000). Lewis and Tanner (2000) studied the effects of above and below-ground competition on growth and survival of seedlings of tropical rain forest trees. They transplanted seedlings of *Aspidosperma carapanauba* (shade tolerant) and *Dinizia exelsa* (light demanding non-pioneer) into a two-factor factorial experiment namely trenching and found that gaps reduced mortality rates and trenching increased growth in both the species. Khurana and Singh (2001), in their review article have explained how seed variability and seedling traits help the species to cope up with various abiotic factors and disturbance. Disturbance and soil degradation also affect plant regeneration. Some species show better performance in highly disturbed stand than in mildly disturbed ones. Influence of soil degradation on the rate of secondary succession and forest composition has been examined by Lafon et al. (2000). Their finding was that the canopy tree species diversity was highest in the least degraded sites. The effect of disturbance levels and associations on the regeneration of *Taxus baccata* was studied by Rikhari et al. (2000). They found that the seedlings of the species required shade environment for growth and survival. Tree or liana species respond to disturbance or stress by showing vegetative mode of

regeneration such as sprouting either through roots or stems. Coppicing is an important means of vegetative regeneration where large-scale disturbance occurs as a result of clearing, burning and extensive damage due to storm. Bellingham and Sparrow (2000) in their comprehensive model have explained the response of tree species to the disturbance and its influence on seeding and sprouting. Paciopek et al. (2000) studied the importance of sprouting in forest dynamics and found that resprouting rate varied between species and families in moist tropical forest.

In India, Rajwar et al. (1999) studied the regeneration status of an Oak forest in Garhwal Himalayas and observed that some tree species showed good regeneration while others failed to regenerate in the forest. Ilorkar and Totey (1999) studied the regeneration status of Navegaon National Park in Maharashtra. They found that different species had maximum regeneration at different altitudes. Kadavul and Parthasarathy (2000) studied forest regeneration pattern of woody species in tropical semi-evergreen forest and found that regeneration of a species depended both on the internal factors of the community and the external disturbances. Uma shankar (2001) studied regeneration in a sal dominated lowland forest and found that out of 93 species, 20.4% showed good regeneration, 10.8% fair, 30.1% poor and 17.2% lack regeneration and the remaining 21.5% were either reappearing or immigrating species. Khan and Tripathi (1987) studied seed germination, growth and survival of *Albizia lebbeck* and found that germination was favoured by alternating temperature treatment (25-35°C) and the seedlings emerging from the seeds buried at greater depth showed better survival. Barik et al. (1992) examined the relationship between microenvironment, size of tree fall gaps and pattern of species establishment in tree fall gaps along a size gradient, and concluded that the pioneer species had large gap size preference while primary species occupy small gaps.

The wealth of Sikkim flora was first revealed by Sir J. D. Hooker in 1872-97 in the form of seven voluminous books entitled “The flora of British India”. Sir J. D. Hooker, was the greatest authority on the vegetation of Sikkim, later on G. King and his colleagues explored the flora and made collections of plants species. Afterwards extensive collections were made by Smith and King (1911), especially in Lhonak and Zemu valley from North district of Sikkim. Reorganization of Botanical Survey of India in 1979 (Anonymous 2000) onwards and setting up of separate circle as Sikkim Himalayan Circle, Botanical Survey of India, Gangtok, has triggered the floristic activity of the region and numerous valuable work from their assessment have appeared in the form of articles in journals, fascicles, floras etc. Maity (2004) assessed the vascular plant diversity of KBR and added more new species, varieties, new report to the previous collection. Most of the works on forests ecology in northeast India have been done by the scholars from North-Eastern Hill University (NEHU), Shillong. They mostly focused on community structure and dynamics, gap dynamics and regeneration ecology in sacred groves as well as in tropical forests in Meghalaya, Manipur and Arunachal Pradesh. In KBR Chettri et al. (2002) studied the impact of firewood extraction on tree structure, regeneration and woody biomass productivity in a trekking corridor. Singh (2000) studied the grazing impact on plant diversity and productivity along a tourist trekking corridor in KBR.

Studies on plant diversity and regeneration dynamics in KBR are extremely limited. The above review of literature clearly indicates that our understanding of plant diversity and regeneration ecology in relation to forest fragmentation is poor.

The study was conducted in Khangchendzonga Biosphere Reserve (27°06 -28°05′ N, 88°02′-88°47′ E) in the Eastern Himalayan state of Sikkim in north-eastern India (Figure 3.1).

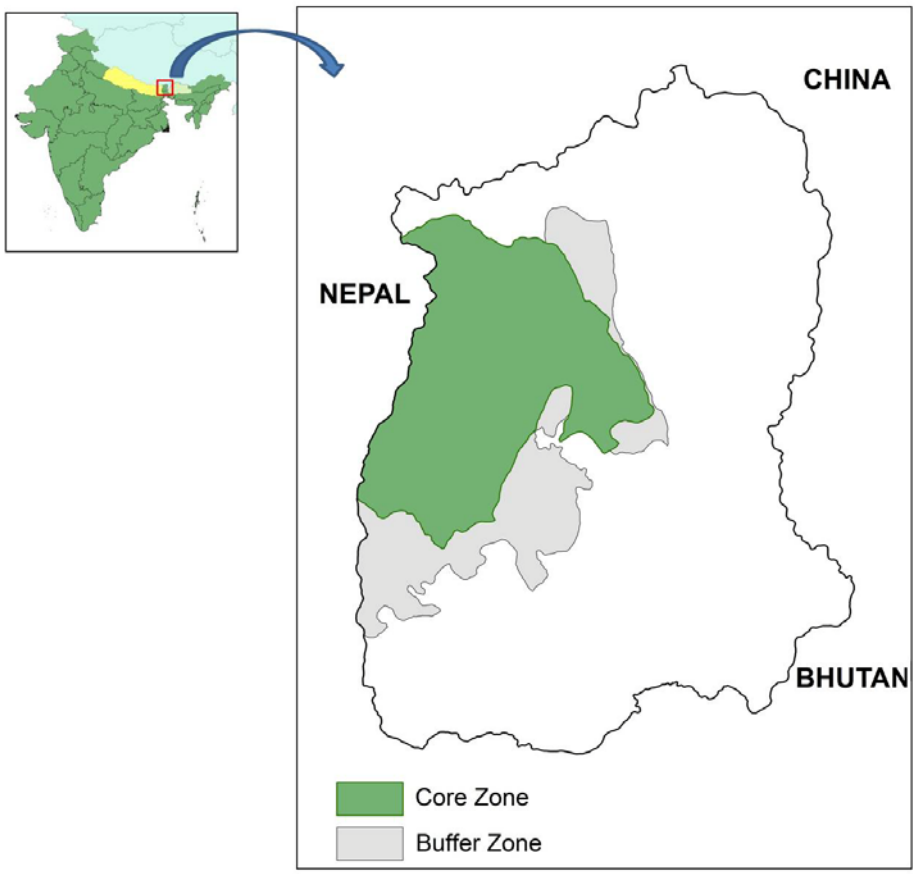


Figure 3.1. Map of Sikkim showing the location of Khangchendzonga Biosphere Reserve in India.

Khangchendzonga Biosphere Reserve (KBR) with a total area of 2619.92 km^2 was notified by the Ministry of Environment and Forests, Government of India as a Biosphere Reserve vide notification No. J-22016/76/91-BR on 7th February, 2000. KBR has 1784 km^2 of core zone and 835.92 km^2 of buffer zone. The Biosphere reserve is a part of Eastern Himalayas, often called as Sikkim Himalayas being in the state of Sikkim. The BR covers two districts of the State, viz., North and West districts. The Biosphere Reserve falls in the elevation range of 1220 m to > 8000 m a.s.l. It lies along the Sikkim-

Nepal border and occupies about 40% of the state's geographical area. Yambung-Singalila range forms the transboundary corridor with Nepal in the West. In the North, the KBR is bound by Lungnak La (5537 m a.s.l) ridge and the Teesta river forms the eastern boundary. In the south, the KBR boundary touches various reserved forests of the South and West Forest Division. It also touches a short stretch of International boundary with the Tibet Autonomous Region (TAR) of China in the North West of the State (Figure 3.1).

Climate

Due to wide altitudinal variation within the BR, a wide variety of climatic conditions are experienced in the BR. The climatic conditions vary from subtropical in the southern part of the BR to cold desert and permanent snow areas in the north. The rainfall pattern is also influenced greatly by the elevation. Sikkim is the most humid place in the whole of the Himalayan range because of its proximity to the Bay of Bengal and direct exposure to the moisture laden southwest monsoon. Three seasons are distinguishable in a year viz., winter (October-March), summer (March-May) and monsoon (June-September) seasons.

Continuous climatic data for the BR are not available because the two closest meteorological stations had data for limited periods and parameters. The Geyzing meteorological station (1533 m a.s.l) is close to the BR (7-10 km distance) from the western direction while Chungthang meteorological station (1606 m a.s.l) is close to the BR (1-5 km distance) from northern direction. The total annual rainfall recorded at Geyzing (Lower montane) during September 2004-August 2006 was 7861.5 mm, 70 % of which was received during April to September and a maximum of 2051.6 mm rainfall was received during the month of August (Figure 3.2). The maximum average daily temperature of 24.8°C was recorded during April and the minimum 5.7 °C was recorded in January at Geyzing. The rainfall data for the four rainy months in each year during 2006-2009 at Chungthang (Montane) revealed that the maximum rainfall occurred during

the months of July and August (Figure 3.2). The continuous data for the Upper montane forests were not available.

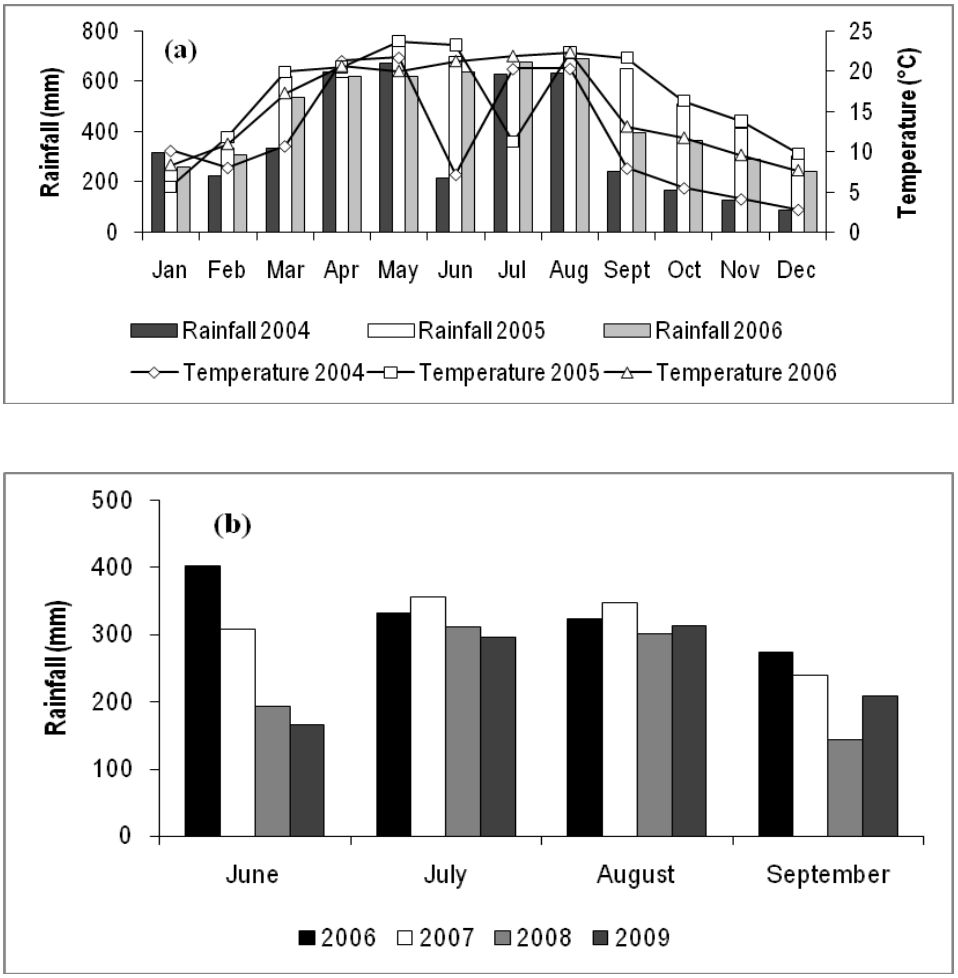


Figure 3.2. Climatic data for (a) Geyzing (Lower montane forests), and (b) Chungthang (Montane forests) recorded during the study period at the nearest two meteorological stations from KBR.

Geology, Geomorphology and Drainage

The state of Sikkim falls in the upper part of the Teesta basin. The BR landscape owes much to the drainage network of the river Teesta. The structural slope of the land is from North to South. Hence all the rivers and the streams in the BR have southern flow. The north-western part of the BR reaches an elevation of > 8000 m a.s.l and therefore remains under snow cover almost throughout the year. The resultant topography is that of the typical glaciated one, characterized by cirques, aretes, glacial trough, and morainic deposits. Besides, there are numerous glacials such as Zemu and Talung glacier in the North, and Rathong glacier in the West, which get frozen during the winter. Geologically,

the BR constitutes hard massive gneissose rocks capable of resisting denudation. The main ridges viz., the Singalila and the Chola ridges within the BR run in a north-south direction. Another north-south ridge runs through the central portion of the BR separating the Rangit from the Teesta valley. The Rangit and the Teesta which form the main channels of drainage, run nearly north-south. Teesta originates from a glacial lake Chho Lhamo located at the north-eastern corner of the BR.

Soil

The soils of BR were in general acidic in reaction due to heavy rainfall and leaching of bases from surface soil to low horizons. They were excessively drained and sandy-loam in texture. According to Harmonised World Soil database, the soil of KBR consists of three main dominant soil types i.e. Cambisols, Leptosols and Glaciers.

Land use

Forest is the dominant land use in KBR. The analysis of imagery pertaining to the year 2002 revealed that more than 43.4% of the total geographical area of the BR was under forest cover or scrub (Figure 3.3). The forest cover/scrub of the BR is 1115.4 km^2, followed by barren land (23.1%), glaciers (12%), meadow (9.3%), snow cover (9.8%). As such, forestry is the major land use in Sikkim and nearly 84% of the total geographic area of the state is under the administrative control of the forest department. The forest cover of the state is 3129 km^2, which is 42% of the total geographic area, followed by barren land 25.4%, pasture and grazing land 17.0%, and the net sown area is 8.9%.

Forest types

Because of wide elevational variation, the BR has diverse forest types ranging from lower montane (subtropical) to Alpine scrubs. Among the forest types described by Lepcha (1998), broad-leaved dense forests occupy the maximum area of 478.2 km^2 in the BR (Table 3.1).

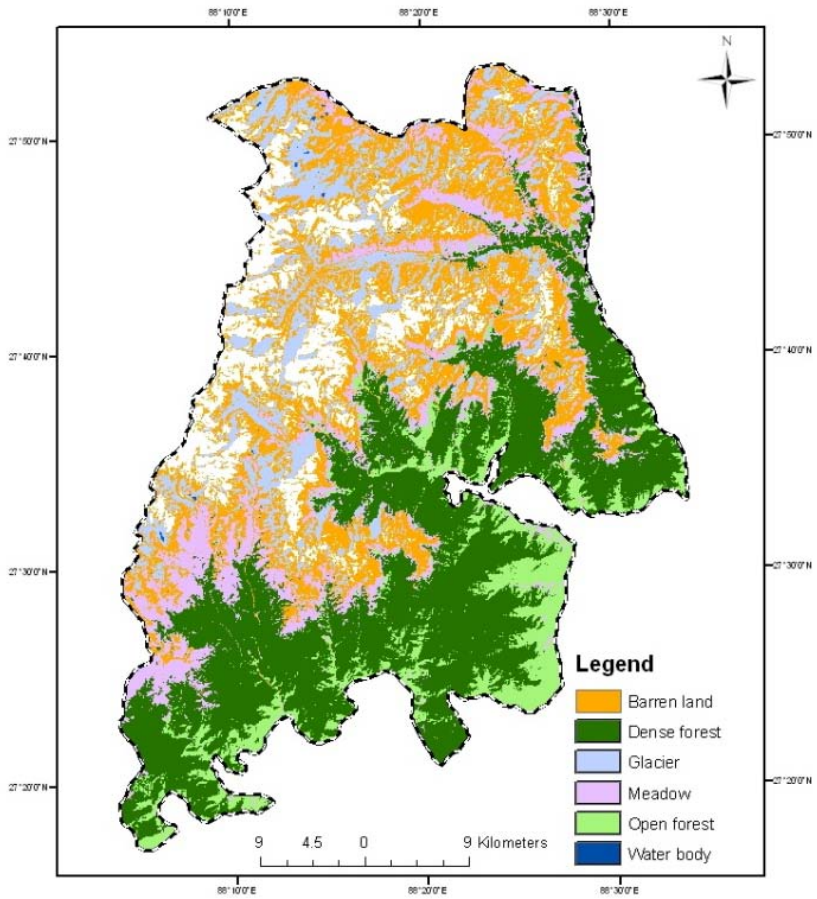


Figure 3.3. Land use of KBR as analysed from the imagery of 2002.

Table 3.1. Area (km^2) statistics of Khangchendzonga Biosphere Reserve.

Forest type	**Area**
Mixed dense forest	478.24
Mixed open forest	180.84
Mixed degraded	172.32
Dense conifer forest	135.81
Open conifer forest	228.12
Degraded conifer forest	140.83
Oak-Rhododendron forest	62.81
Scrubs	28.59
Forest blanks	84.28
Alpine scrubs	244.79
Alpine pastures	20.62
Alpine barren	216.78
Snow	480.93
Glaciers	152.68
Lakes	4.28
River/major streams	13.42
Dry river bed	7.98
Total	**2653.32**

Biodiversity

The Eastern Himalayan region is a biodiversity hotspot of the Indian subcontinent that harbours more than 3000 endemic species (McGinley 2008). The great varieties of environmental conditions found in Himalayas have resulted in diverse ecosystems types, which are rich in species and genetic diversity. Therefore, the Eastern Himalayan region is one of the richest zones of biological diversity in the world. Takhtajan (1969) had considered this region as the "Cradle of flowering Plants". The flora of Sikkim was first described by Sir J. D. Hooker in 1872-97 in the form of seven voluminous books entitled "The flora of British India". Besides Sir J. D. Hooker, G. King and his colleagues explored the flora of Sikkim latter (King & Pantling 1898).

A part of Khangchendzonga BR falls within the biogeographic province of Trans-Himalaya-Tibetan plateau biogeographic zone with biota of Palaearctic affinity (Rodgers et al. 2002). The rests of the BR is a part of Indo-Malayan Biogeographic region. The BR has the richest biodiversity in the Himalayan region, being home to about 140 endemic plant species spread over 41 families (Sharma et al. 2001). The topography, elevational variation, high peaks, glacial lakes, and forest wilderness in biosphere reserve has enriched the KBR's biodiversity. Singh and Chauhan (1997, 1998) reported the presence of 16 species of gymnosperms belonging to 12 genera under seven families from BR. Maity (2004) reported 11 species under 9 genera belonging to five families. Maity (2004) also reported the presence of 1463 species of angiosperms belonging to 138 families. However, the taxa found at higher elevations still remain to be explored.

BR is also equally gifted with high faunal diversity. Ali (1960) reported as many as 430 bird species. Chettri (2000) compiled a list of rare and endangered birds under different schedules of Wildlife Protection Act, 1972. Examples of these birds are *Lophophorus impejanus* (state bird), *Budo nepalensis*, *Ithagenus cruentus* etc. Nearly 150

Plate 3.1. (a) An overview of Khangchendzonga Biosphere Reserves; (b) Upper montane forests; (c) Montane forests, and (d) Lower montane forests.

species of mammals belonging to 28 families have been recorded from BR. Some of the important ones are *Uncia uncia*, *Canis lupus*, *Pseudois nayaur* and *Ailurus fulgens*.

Study sites

Six sites located in three forest types were selected for detailed plant diversity study (Figure 3.4). These three forest types are located along the two trekking paths. The first trekking path passes through Topung, Ngom and Thaprang. While, the second trekking path runs across the places like Yuksum, Bakhim and Kibeck, along the three elevational ranges within the BR. These three forest types differ in topographic characteristics (Table 3.2).

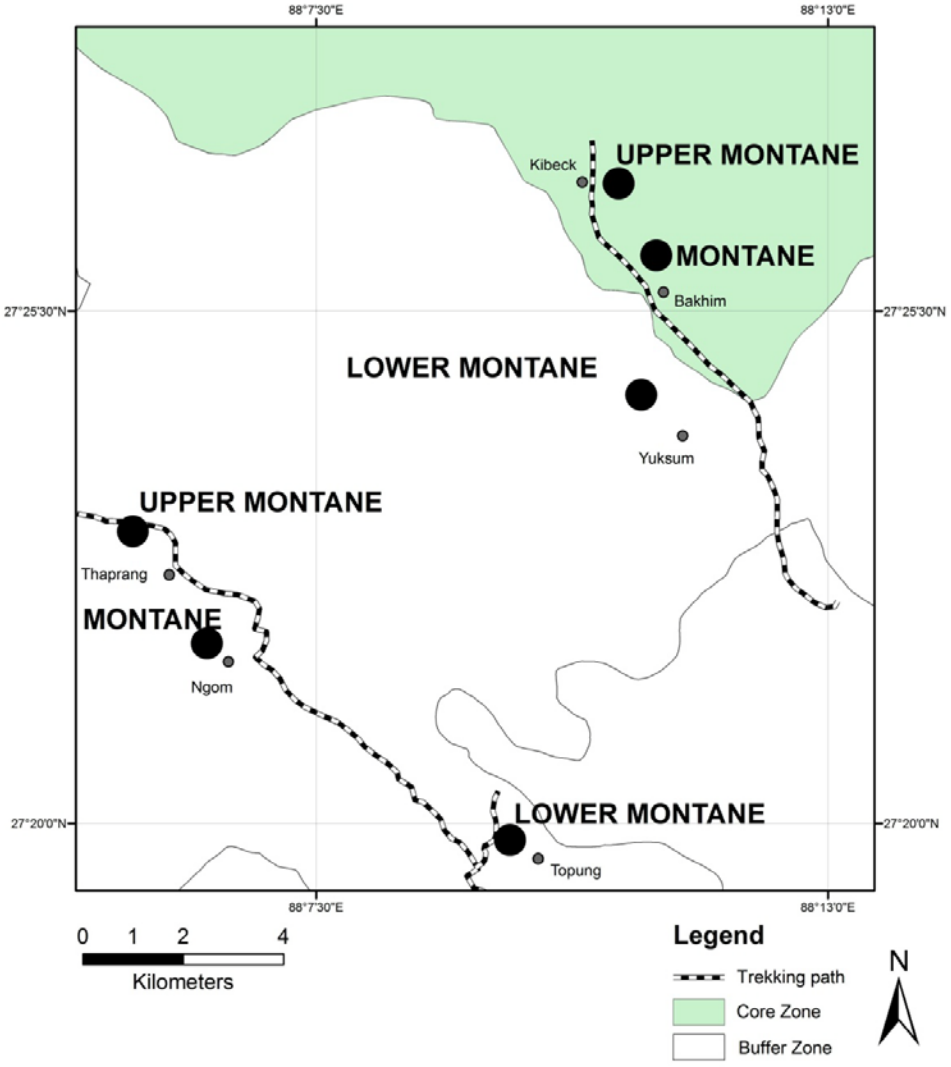


Figure 3.4. Map showing the locations of the study sites for plant diversity assessment in KBR.

Table 3.2. Topographical features of six study sites in three forest types of KBR.

Site characteristics	**Lower montane**		**Montane**		**Upper montane**	
Sites	Topung	Yuksum	Ngom	Bakhim	Dungdang	Kibeck
Coordinates	27° 19' N, 88° 09' E	27° 23' N, 88° 12' E	27° 26' N, 88° 10' E	27° 25' N, 88° 11' E	27° 23' N, 88° 04' E	27° 26‘N 88°10' E
Aspects	North	North	South	South	South	South
Slope (degree)	20 – 45	10-25	10 – 30	15-30	10 – 40	15-35
Elevation range (m)	1200 – 1910	1350-1900	1930 – 2560	1900-2500	2680 – 3000	2900-3100

The three forest types are, Lower montane, Montane, and Upper montane. As per the classification of Champion and Seth (1968), the forest at an elevation of 1500 m a.s.l. is classified as East-Himalayan subtropical wet-hill forest ($8BC_1$) under the group subtropical broad leaved hill forests. The forest at an elevation of 2000 m a.s.l. is classified as East-Himalayan moist temperate mixed coniferous forest (12C3a) under the group Himalayan moist temperate forest. The forest at an elevational range of 3000-3050 m a.s.l is classified as East-Himalayan subalpine-birch/fir forest ($14C_2$) under the group subalpine forest. These three montane forests are referred to as subtropical, temperate and subalpine forests, respectively based on their group name. The following three forest types representing six sites were selected for detailed plant diversity study: (1) Topung-Yuksum was the site at mid-elevation within the buffer zone representing Lower montane forests. The dominant tree species were *Alnus nepalensis* D. Don, *Castanopsis tribuloides* DC., *Engelhardtia spicata* Bl., *Ficus semicordata* Sm., and *Lyonia ovalifolia* Drude. (2) Ngom-Bakhim was the site at higher elevation within the buffer zone representing Montane forests. The dominant tree species in the broadleaved temperate forest were *Acer campbellii* Hiern, *A. nepalensis*, *Betula alnoides* D. Don, *Lithocarpus pachyphylla* Rehder, *Magnolia campbellii* Hk. f. & Thom., and *Rhododendron arboreum* Sm., and (3) Kibeck-Dungdang was the site at highest elevation representing Upper montane forests and falls in the core zone. The dominant tree species were *Abies densa* Griffith ex Parker, *Buddleia colvilei* Thom., *Rhododendron* spp., and *Tsuga dumosa* Eichler.

The composition of soil types of KBR in three forest types were of mainly Leptosols and Glaciers. The textural class of soil was sandy to loamy sandy in the Lower montane forests; sandy soil in Montane and sandy loam to loamy sandy in the Upper montane forests in BR. Soil pH ranged from 5 to 5.2 in the Lower montane forests, 4.2 to 4.9 in Montane and 4.2 to 4.3 in the Upper montane forests. Average soil organic carbon and nitrogen were higher in the Montane forests, while in the Upper montane forests, soil phosphorus, potassium, and moisture were highest. Overall soil temperature was maximum in the Lower montane forests in comparison to Montane and Upper montane forests (Table 3.3).

Table 3.3. Mean of edaphic characteristics of the three forest types in Khangchendzonga Biosphere Reserves (after Chettri et al. 2009).

Parameters	**Lower montane**	**Montane**	**Upper montane**
Soil temperature (°C)	18.56	13.1	7.43
Soil organic carbon (%)	4.29	4.76	4.21
Soil potassium(µg g-1)	16.79	19.41	22.30
Soil moisture content (%)	35.90	44.61	59.27
Soil Nitrogen (%)	0.33	0.67	0.60
Soil pH	5.18	4.58	4.34
Soil Phosphorus (µg g-1)	15.56	26.77	42.71

Chapter 4
Plant diversity-environment relationship in KBR

4.1 Introduction

The Eastern Himalaya with more than 3000 endemic species is one of the 34 biodiversity hotspots of the world and spreads over an area of 1500 km^2 in Sikkim, West Bengal, Arunachal Pradesh and Nagaland. Being at the meeting point of Indo-Malayan and Indo-Chinese biogeographical realms as well as Himalayan and peninsular Indian region, it contains the floristic elements from all these biogeographical zones.

KBR is a part of Eastern Himalayan region and consists of diverse ecosystem types. Each of these ecosystems is characterized by great variations in elevation, climate, landscape, habitat, species composition and vegetation types. The following seven types of ecosystems were described within the KBR (Table 4.1): (i) Lower montane forests (ii) Riverine forests (iii) Grassland (iv) Montane broad-leaved forests (v) Upper montane forests (vi) Meadow, and (vii) *Rhododedron* scrubs. The Lower montane region of KBR is found above 1200 m elevation in the buffer zone and comprises of Lower montane forests, riverine and grassland ecosystems. Montane region of KBR falls in the core zone and comprises of montane broad-leaved forest ecosystem between 2000-3500 m elevations. Upper montane zone, above 3500 m elevation also is a part of the core zone and comprises the upper montane forests, meadows and *Rhododendron* scrubs. The meadows are widely distributed between the treeline and snow line. The *Rhododendron* scrubs are located above the treeline upto 4000 m a.s.l. The meadow represents the natural grassland ecosystem.

The existence of such a wide range of ecosystems in KBR, diverse edapho-climatic and physiographic conditions, and presence of floral elements from a number of species-rich biogeographic realms due to its locational advantage have resulted in having the richest plant diversity in the Himalayan range. KBR is home to at least 140 endemic

plant species (Sharma et al. 2001). KBR is extremely rich in vascular epiphytes and lianas that provide greater level of complexity to the ecosystems of which they are a part. In addition to the high level of species diversity at all level of life forms and endemism, the BR also plays an important role in maintaining elevational connectivity between the habitat and ecosystem types that make up the larger Himalayan ecosystem.

The plant diversity in KBR has not been fully explored, particularly those belonging to such interesting groups of plants as vascular epiphytes and lianas. The factors contributing to high plant diversity in general or to the prevalence of a specific plant group in different forest ecosystems hitherto remained unexplained. An attempt to explain these underlying factors would help to answer one of the most important ecological questions related to biodiversity i.e. why some ecosystems are so species-rich? This chapter therefore documents the plant diversity in the KBR and relates to various microenvironmental factors to identify the factors those contribute most to the high plant diversity in different forest types.

Table 4.1. Terrestrial ecosystem types identified along the elevational gradient in KBR, Sikkim.

Elevational range (m)	Forest type	Ecosystem type	Location (Core zone/ Buffer zone)
1200-1900	Lower montane	Lower montane forests	Buffer
		Riverine forests	Buffer
		Grassland	Buffer
2000-2500	Montane	Motane broad-leaved forests	Core
2600-3100	Upper montane	Upper montane forests	Core
		Meadows	Core
		Rhododendron Scrub	Core

4.2 Methods

For studying the community structure of the Biosphere Reserve (BR), six sites were selected in three forest types in different elevational ranges viz., Lower montane, Montane and Upper montane forests along the two trekking routes. These two trekking routes differ in species composition, as well as amount of human impact and tourism

flow. Both the routes cut across the core and buffer zones. Each of the six sites identified for detailed study along these two trecking routes is spread over an area of at least 1 km^2 with continuous vegetation. An area of 1 km^2 was thus demarcated at each site for detailed sampling, which henceforth has been referred to as forest stand.

Species composition

The species composition in each forest stand was studied by collecting the specimens and preparing the herbaria. The specimens were identified with the help of existing herbarium records of Botanical Survey of India, Himalayan Circle, Sikkim and Eastern Circle, Shillong. The available floras (Cowan & Cowan 1929; Hara 1966, 1971; Polunin & Stainton 1984; Maity & Maiti 2007) were referred that ensured the correct identification of the species.

Community structure and plant diversity

Tree, Shrub and Herb

Ten quadrats were laid randomly at each site for sampling trees and shrubs. Sample plots of 20 x 50 m (0.1 ha) were used for sampling trees and shrubs. A total of 60 quadrats were laid in the six forest stands. Various life forms in the forest vegetation were defined as follows: individuals having DBH (diameter at breast height) $\geq$ 5 cm and having a distinct trunk and crown were considered as trees. Shrubs were distinguished from trees by the absence of a distinct trunk. The herbs (< 1 m in height with no woody stems) were enumerated in 20 randomly placed 1 m x 1 m subplots within each sample plot.

Epiphytes

The vascular epiphytes were sampled on 10 selected old growth canopy host trees in each of the six forest stands that covered all the three forest types. Some host trees belonged to the dominant species e.g. *Abies densa* and some trees belonged to the common species e.g. *Rhododendron* spp., in Upper montane forests. All the host trees selected were highly loaded with epiphytes and it was not diificult to find such trees at each site. In the Upper

montane forests, the host trees belonged to *Abies densa*, *Acer campbellii*, *Betula alnoides*, *Ilex dipyrena*, *Rhododendron falconeri*, *Prunus* spp., *Rhododendron campanulatum* and *Viburnum nervosum* species. In the Montane forest stands, *Elaeocarpus lanceaefolius*, *Ilex fragilis*, *Ilex dipyrena*, *Lithocarpus pachyphylla*, *Magnolia campbellii*, *Rhododendron arboreum*, *R. falconeri*, *Quercus lineata*, *Quercus lamellosa*, and *Sorbus cuspidata* were selected. In the Lower montane forests trees belonging to *Alnus nepalensis*, *Acer thomsonii*, *Castanopsis tribuloides*, *Engelhardtia spicata*, *Ficus auriculata*, *Ficus semicordata*, *Lyonia ovalifolia*, *Prunus cerasoides* and *Schima wallichii* were selected. Each host tree was divided into 10 m vertical intervals from ground to the canopy top following Johansson (1974). The epiphytes on tree trunk at different height intervals were counted by using rope access techniques (Perry 1978b) and by use of binoculars. The species occurring in dense patches like most of the ferns, some orchid species and the members of Piperaceae were counted as stands, and one stand meaning one 'individual'. Host tree DBH was taken at breast height (1.37 m from the ground level) and it ranged from 0.41m to 0.84 m in the Upper montane, 0.87 m to 0.36 m in the Montane and 0.81m to 0.31m in the Lower montane forest stands.

Lianas

All the individuals of adult liana (≥ 0.2 cm DBH and > 1.3 m length/height from the point of emergence from the ground) were identified and enumerated in 10 randomly located replicate plots of 50 m x 20 m size in each of the three forest types. The diameter of adult lianas was measured at 1.37 m from the ground level (DBH) with a cloth diameter tape following the protocol described in Gerwing et al. (2006). For stems that were excentric, flattened or elliptical rather than cylindrical, the diameter was measured at the widest and narrowest points and the mean was calculated.

Dominance

In order to assess the relative share of each species in plant community, importance value index (IVI) for a total score of 300 was calculated (Rao et al. 1990; Barik et al. 1992). Frequency (number of quadrats in which species occurred/total number of quadrats studied), basal area (basal area of the species per quadrat) and density (total number of individuals of the species/total number of quadrats studied) values for each species were calculated. The following formula was used to calculate IVI: IVI = RF + RBA + RD, Where RF, relative frequency (%) = (frequency of the species/frequency of all the species) x 100; RBA, relative basal area (%) = (basal area of the species in all the quadrats/ basal area of all the species in all the quadrats) x 100; and RD, relative density (%) = (density of the species/ density of all the species) x 100. The relative basal area values were derived either from stem DBH or basal diameter values depending upon the category of plant lifeform.

Diversity

All the diversity indices used in this study were computed using Pisces Conservation SDR version (Seaby and Henderson 2007). Shannon's diversity index (*H*), Pielou's evenness index (*J*), Simpson's Dominance index (*D*) and Fisher's Alpha diversity (α) were calculated using the Species Diversity and Richness package 4.1.2 (PISCES Conservation Ltd. 2007).

Fishers's alpha

This is a parametric index of diversity which assumes that the abundance of species follows the log series distribution, αx, $\alpha x^2/2$, $\alpha x^3/3$ ... $\alpha x^n/n$, where each term gives the number of species predicted to have 1, 2, 3....n individuals in the sample. The index is the alpha parameter. A number of authors argue strongly in favour of this index (Kempton & Taylor 1976).

Whittaker's βw

β-diversity measures the increase in species diversity along sample size and is particularly applicable to the study of environmental gradients:

βw = (S/α)-1, where, S = the total number of species encountered in the two sites counting each species only once and 'α'the average species richness of the samples. All samples must have the same size (or sampling effort).

Shannon-Weiner diversity Index

Species diversity was calculated using Shannon-Weiner index of diversity (Shannon and Wiener 1963) as: $H = -\sum (Ni/N) \ln (Ni/N)$ where, 'Ni' is the IVI of i^{th} species and 'N' is the total IVI.

Simpson's dominance Index

Simpson's dominance Index (D) was estimated using Simpson index (Simpson 1949) and was calculated as: $D = \sum (Ni/N)^2$, where, 'Ni' is the IVI of the i^{th} species and 'N' is the total IVI of all species.

Pielou J (All samples)

This measure of equitability compares the observed Shannon-Wiener index against the distribution of individuals between the observed species which would maximise diversity. If H is the observed Shannon-Wiener index, the maximum value this could take is log(S), where S is the total number of species in the habitat. Therefore, the index is: J = H/log(S).

Analysis of similarity (ANOSIM)

This test was developed by Clark (1988, 1993) as a test of the significance of the groups that had been defined *a priori*. The idea is simple, if the assigned groups are meaningful, samples within groups should be more similar in composition than samples from different groups. The method uses the Bray-Curtis measure of similarity. The null hypothesis is therefore that there are no differences between the members of the various groups.

Clark (1988, 1993) proposed the following statistic to measure the differences between the groups: $R = \frac{\bar{r}_B - \bar{r}_W}{N(n-1)/4}$ where, $\bar{r}_B$, $\bar{r}_W$ are the means of the ranked similarity BETWEEN groups and WITHIN groups respectively and n is the total number of samples (quadrat). R scales from +1 to -1. +1 indicates that all the most similar samples are within the same groups. R = 0 occurs if the high and low similarities are perfectly mixed and bear no relationship to the group. A value of -1 indicates that the most similar samples are all outside of the groups. While negative values might seem to be a most unlikely eventuality it has been found to occur with surprising frequency.

To test for significance, the ranked similarity within and between groups is compared with the similarity that would be generated by random chance. Essentially the samples are randomly assigned to groups 1000 times and R calculated for each permutation. The observed value of R is then compared against the random distribution to determine if it is significantly different from that which could occur at random. If the value of R is significant, one can conclude that there is evidence that the samples within groups are more similar than would be expected by random chance.

Similarity percentage (SIMPER)

This analysis breaks down the contribution of each species to the observed similarity (or dissimilarity) between samples. It will allow us to identify the species that are most important in creating the observed pattern of similarity. The method used the Bray-Curtis measure of similarity, comparing in turn, each sample in forest type 1 with each sample in forest type 2. The Bray-Curtis method operates at the species level and therefore the mean similarity between forest 1 and forest 2 can be obtained for each species.

Floristic Ordinations

The objective of the ordination is to help generate hypotheses about the relationship between the species composition at a site and the underlying environmental gradients

(Digby & Kempton 1987). Any inherent pattern that the data may possess becomes apparent for visual inspection (Pielou 1984). It summarizes community data such as species abundance by producing a low dimensional space in which similar species and samples are plotted close together, and dissimilar species and samples are far apart. All ordination was undertaken using Community Analysis Package Version 4.1.3 (2007).

Non-Metric Multidimensional scaling

Multidimensional scaling (MDS) is a technique for expressing the similarities between different objects in a small number of dimensions. This allows a complex set of inter-relationships to be summarised in a simple figure. The method attempts to place the most similar objects (samples) close together. The starting point for the calculation is a similarity or dissimilarity matrix between all the sites or quadrats. These can be non-metric distance measures for which the relationships between the sites/objects/samples (columns) cannot be plotted in a Euclidean space. The aim of non-metric MDS is to find a set of metric coordinates for the sites which most closely approximates their non-metric distances. CAP (PISCES) has been used for MDS analysis that employs Kruskals's least squares monotonic transformation to minimise the stress (Kruskal 1964; Kruskal & Wish 1977).

Epiphyte lifeform and taxonomic groups

All epiphytic species were classified by lifeform i.e. growth habit and by taxonomic group. The lifeform classification in the present study is a modified version of original classification of Hosokawa (1943). The lifeform groups are as follows:

Ascending: A plant where the main stem is erect, plant stems curve upwards from the node.

Caespitose: A plant with a tufted growth form where stems arise from a basal node or rhizome.

Climber: A plant that climbs and attaches vertically.

Closed Tank: A bromeliad with tightly tubular enclosed rosette.

Filmy Fern: A filamentous fern group.

Lepanthid: A pleurothallid orchid with lepanthiform sheaths.

Long Creeping: A fern with a long creeping rhizome.

Long Repent: A plant with a long rhizome/stem that spreads along the stem and sends out roots from nodes.

Pendant: A plant that has drooping stems and leaves.

Short Creeping: A fern with a short creeping rhizome.

Short Repent: A plant with a very short rhizome/stem that spreads along the stem and sends out roots from nodes.

The Taxonomic groups are as follows:

Aroid: All members of the Araceae.

Bromeliad: All members of the Bromeliaceae.

Herb: All non-woody dicotyledonous angiosperms.

Orchid: All members of the Orchidaceae excluding the subtribe Pleurothallidinae.

Pleurothallid: All members of the Pleurothallidinae (Orchidaceae).

Fern: All members of the Polypodaceae.

Woody Dicot: All woody dicotyledonous angiosperms.

Measurement of microenvironmental factors

Climatic (light intensity, relative humidity and air temperature) and edaphic (soil temperature, moisture, pH, total organic carbon (C), total Kjeldahl nitrogen (N), available phosphorus (P) and potassium (K)) microenvironmental variables were studied at seasonal intervals for three seasons in each forest types. The microclimatic variables were measured in every 20 1 m x 1 m subplots within each 50 m x 20 m sample plot. The composite soil sample for each of these plots was prepared by mixing soils collected from the 20 subplots of 1 m x 1 m. The mean values for the microclimatic parameters were

calculated for each of the 50 m x 20 m sample plots based on the values obtained from the respective 20 1 m x 1 m subplots and were used for comparing the variables among the forest types and relating to density. The measurements were taken at 1 m above ground level, three times a day at 3 hourly intervals, i.e. at 10 a.m., 1 p.m. and 4 p.m. for five consecutive days each in August (for rainy season), January (for winter) and April (for summer) during the study period.

Statistical analysis

The variation in environmental factors, among the forest types and seasons was analysed by using two-way ANOVA (fixed effect model). The assumptions of the ANOVA were met through tests of normality of variables (Kolmogorov–Smirnov test) and homogeneity of group variances (Levene's test). Constrained weighted average ordination technique, Canonical Correspondence Analysis (CCA) (ECOM II 2.1.3.137 of PISCES Conservation Ltd. 2007) was used to explore how species respond to specific environmental variables across the forest types (McCune 1997). CCA was appropriate for studying the relationship across the forest types since the variation was large and over a wider range, and thus represented a unimodal response model (Ter Braak & Prentice 1988). The mean values of the three seasonal microenvironmental data sets in the three forest types were used for CCA. To avoid multicollinearity among the environmental variables, a test for collinearity was carried out before performing CCA. Monte Carlo randomisation (ECOM II) test was performed with 100 trials to confirm the statistical significance of the CCA. Realizing the canopy habitat of the epiphyte species, edaphic variables were not correlated with density data during CCA analysis. To identify the most important environmental variables related to all the adult lifefom densities in each forest type, forward stepwise multiple regression analysis was performed considering environmental parameters as explanatory variables and liana density as dependent variable. The analysis was performed by adding parameters sequentially starting from no

variable in the model, and then adding the most significant explanatory variable, i.e. the one with the lowest *P*-value, at each step until all variables were added (ECOM II). The data were standardised using log (x+1) transformation before regression analyses.

4.3 Results

Trees

Seventy-eight tree species belonging to 47 genera and 30 families were recorded from the three forest types. The number of species was highest in the Lower montane forests (49) followed by the Montane (28) and Upper montane (11) forests (Table 4.2). Tree species richness decreased with increasing elevation (R = -0.53; $P < 0.05$). The diversity indices also decreased with increasing elevation (H = 3.74, 3.26, 2.26 and J = 0.96, 0.98 and 0.95). The dominance index (D) also followed the same trend (D = 38.5, 25.8, 8.87). Fisher' alpha diversity for trees was highest in the Lower montane (11.04), compared to Montane (6.50) and Upper montane (1.97) forests (Table 4.3).

β-diversity was highest between the Lower and Upper montane (0.98), Montane and Upper montane (0.95) forest stands. Lower Montane and Montane forests had the lowest β-diversity value of (0.77).

Table 4.2. Species richness, density and basal area of trees, shrubs and herbs, in KBR.

Parameters	Lower montane	Montane	Upper montane
Trees			
Species richness	49	28	11
Density (ha^{-1})	463	239	256
Basal area ($m^2 ha^{-1}$)	92.6	49.9	58.1
Shrubs			
Species richness	33	06	06
Density (ha^{-1})	319	101	234
Herbs			
Species richness	61	52	39
Density (ha^{-1})	609500	711500	625000

Table 4.3. General diversity patterns for different Lifeforms in three forest types (values in parenthesis indicate Jackknife standard error).

Parameters	Lower montane	Montane	Upper montane	Total
Trees				
Fisher's alpha diversity (α)	11.04(0.2)	6.5 (0.15)	1.97(0.05)	19.01(0.25)
Shannon-Wiener index (H)	3.74(0.02)	3.26(0.03)	2.26(0.06)	4.27(0.02)
Simpson's dominance index (D)	38.56(1.61)	25.76(1.40)	8.87(0.92)	59.68(2.84)
Pielou J	0.96(0.01)	0.98(0.01)	0.95(0.02)	0.95(0.01)
Shrubs				
Fisher's alpha diversity (α)	6.54(0.17)	1.20(0.03)	0.97(0.03)	8.29(0.16)
Shannon-Wiener index (H)	3.28(0.06)	1.55(0.02)	1.72(0.06)	3.48(0.07)
Simpson's dominance index (D)	24.38(3.28)	3.37(0.04)	5.36(0.67)	26.28(4.64)
Pielou J	0.44(0.96)	0.80(0.01)	0.96(0.03)	0.93(0.02)
Herbs				
Fisher's alpha diversity (α)	13.53(0.27)	10.61(0.61)	7.66(0.29)	31.55(0.60)
Shannon-Wiener index (H)	3.93(0.03)	3.52(0.08)	3.23(0.13)	4.67(0.05)
Simpson's dominance index (D)	45.39(3.47)	28.11(3.70)	14.28(3.70)	70.84(11.6)
Pielou J	0.95(0.01)	0.91(0.02)	0.88(0.37)	0.93(0.01)

Euphorbiaceae and Fagaceae were the dominant family in the Lower montane forests. Aceraceae, Ericaceae, Fagaceae and Lauraceae, each with 10.7% of the total species dominated the tree species composition in the Montane forests. However, in the upper Montane forests, Ericaceae was the dominant family with 63.6% of the total species share.

The three forest types differed significantly in tree species composition (Clark's R static = 0.95, $P < 0.001$) (Table 4.4). Species dissimilarity between the Lower montane and Montane, Lower and Upper montane, and Montane and Upper montane forests was 99.06, 99.02 and 99.07%, respectively (Table 4.4).

Table 4.4. Similarity test values of ANOSIM and SIMPER in the sampled sites for Lower montane, Montane and Uupper montane forests. The ANOSIM 'R value' is the statistical value of similarity within each forest stands with a probability of 0.001.

All stands together				
R value	P value			
0.95	0.001			
Stand wise test (No. of quadrats)				
1st group	2nd group	P value	ANOSIM (R value)	SIMPER (average dissimilarity)
Lower montane	Montane	0.001	0.94	99.06
Lower montane	Upper montane	0.001	0.98	99.02
Montane	Upper montane	0.001	0.93	99.07

Acer campbellii, *Elaeocarpus lanceaefolius*, *Eurya acuminata*, *Quercus lineata* and *Viburnum nervosum* were confined to Lower montane and Montane forests. *Prunus* spp., *Rhododendron arboreum* and *R. falconeri* were found only in the Montane and Upper montane forests (Annexure 1).

The density of tree decreased with increasing elevation (F = 22.50, P < 0.001). Tree density was highest in the Lower montane (463 stems ha^{-1}) forest followed by Montane (239 stems ha^{-1}), and the Upper montane forests (256 stems ha^{-1}). Basal area was highest in the Lower montane forests (92.6 $m^2 ha^{-1}$) compared to the Montane and the Upper montane forests stands (49.9 and 58.1 $m^2 ha^{-1}$, respectively) (Table 4.2).

With an increase in elevation, the tree species-abundance curves exhibited higher dominance (Figure 4.1). *Alnus nepalensis*, *Castanopsis hystrix*, *Elaeocarpus lanceaefolius* and *Quercus lineata* together shared more than 27% dominance in the Lower montane forests. *Lithocarpus pachyphylla*, *Magnolia campbellii*, *Quercus lamellosa* and *Rhododendron arboreum* were the dominant tree species in the Montane forests. However in the Uupper montane forests, three dominant species viz. *Abies densa*, *Tsuga dumosa* and *Rhododendron arboreum* shared 50% of the total IVI (Annexure I).

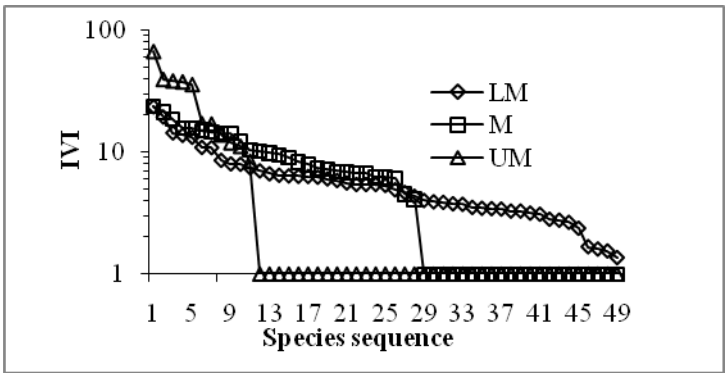


Figure 4.1. Dominance-diversity curves for tree species in Lower montane (LM), Montane (M) and Upper montane (UM) forests in KBR.

The most commonly occurring tree species in the Lower montane forests were *Alnus nepalensis*, *Elaeocarpus lanceaefolius*, *Eurya acuminata* and *Rhus javanica.* Whilst in the Montane forests, *Lithocarpus pachyphylla*, *Quercus lamellosa*, *Q. lineata* and

Rhododendron arboreum were commonly encountered (>45%), whereas *Abies densa* and *Rhododendron* spp., were frequent in the Upper montane forests (>55%).

Density-girth class distribution pattern of tree shows that the Lower montane forests had more number of individuals in the lower 'girth classes' (5-15 cm and 16-25 cm) than in Montane and Uupper montane forests. The number of individuals in the higher girth classes (> 91 cm) was more in Upper montane and Lower montane forests. In the Upper montane forests, middle girth classes (35-60 cm and 61-90 cm) were absent, as no individuals was encountered in these girth classes. Comparatively, stem density decreased with increase in girth classes in all the forest stands (Figure 4.2).

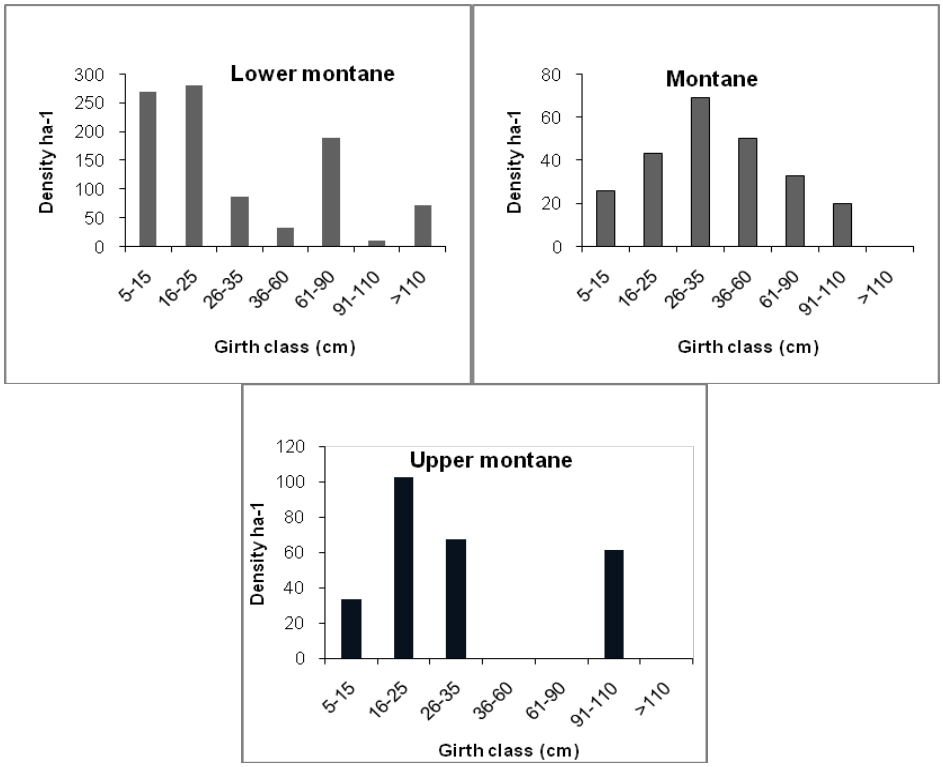


Figure 4.2. Population structure of tree species in Lower montane, Montane and Upper montane forests in KBR.

Microenvironmental factors related to tree abundance

Of the 10 microenvironmental variables studied, air temperature, soil temperature, soil moisture content, Phosphorus (P) and Nitrogen (N) varied significantly (ANOVA $P < 0.01$) among the three forest types. Air temperature, soil temperature, soil moisture

content, soil Carbon (C) and N varied significantly (ANOVA $P < 0.05$) among the seasons (Table 4.5 and Figure 4.3).

Table 4.5. Results of two-way ANOVA of microenvironmental factors to assess the variations due to forest types (Lower montane, Montane, Upper montane) and seasons (winter, summer, monsoon) in KBR. For each environmental variable n = 9 and d. f. = 2 for both forest type and season.

Environmental parameters	**Forests**		**Seasons**	
	F	P	F	P
Light	5.29	0.07	0.51	0.63
Soil pH	2.39	0.20	0.42	0.67
Soil phosphorus	16.27	0.01	3.25	0.14
Relative humidity	0.48	0.64	3.99	0.11
Soil carbon	2.30	0.21	8.07	0.03
Soil potassium	3.79	0.11	4.60	0.09
Soil temperature	35.09	0.00	22.15	0.00
Air temperature	23.74	0.00	7.86	0.04
Soil moisture content	18.28	0.00	9.7	0.02
Soil nitrogen	9774.70	0.00	30.10	0.00

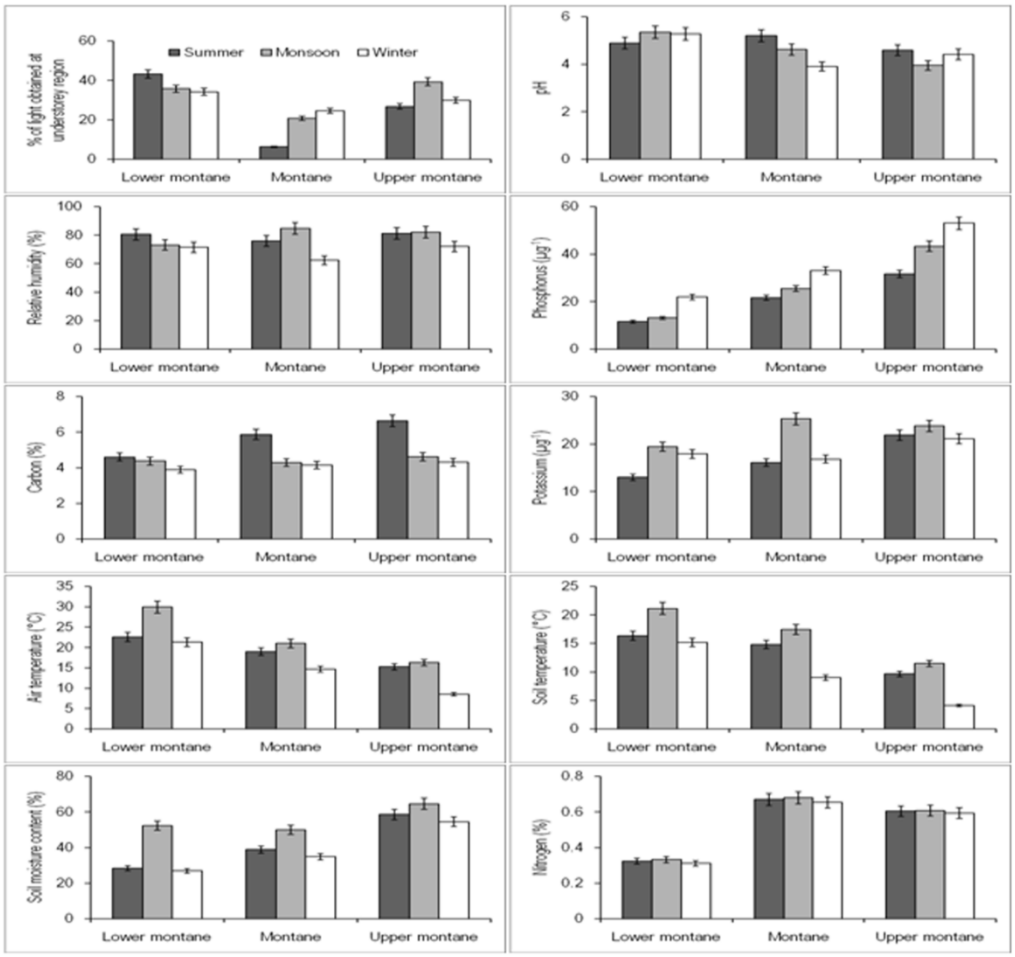


Figure 4.3. Seasonal variation in microenvironmental variables in Lower montane, Montane and Upper montane forests in KBR. Bars represent Standard Error.

The tree species-environment relationship across the three forests was analyzed through CCA. The relationship was weakly explained as the first two canonical axes accounted for 11.5% and 8.3% of the total variance. However, Monte Carlo randomisation test with 100 iterations confirmed the test of significance at $P < 0.009$ level (Table 4.6). N, pH, P and variable 'elevation' were strongly correlated with the first CCA axis and therefore were important determinants of tree species distribution across the forest types (Figure 4.4).

CCA produced an ordination of all 78 tree species that showed the inferred ranking of the species along the four environmental variables. The ordination plot shows the relative position of the species along the line of environmental vectors depicting species environmental preferences. In the Lower montane forests, *Toona ciliata*, *Albizia chinensis*, and *Eurya japonica* with high first axis species scores dominated the areas with high soil pH. Conversely, *Cinnamomum impressinervium, Castanopsis indica*, and *Acer campbellii* occupied low soil pH areas. In the Montane forests, *Acer thomsonii*, *Alnus nepalensis*, *Elaeocarpus lanceaefolius* with high first axis species scores were associated strongly with high soil N level, while *Evodia fraxinifolia*, *Ilex fragilis*, and *Prunus cerasoides* were confined to low soil N areas. In the Upper montane forests, *Abies densa*, *Tsuga dumosa*, and *Ilex dipyrena* were dominant in high soil P and C environment, while *Rhododendron grandiflorum* and *R.campanulatum* were abundant in low soil P and C areas (Figure 4.4). In the Upper montane forests, elevations also influenced the abundance and distribution of tree species.

The relationship between microenvironmental variables and tree density as shown by stepwise forward multiple regression analysis indicated that N ($P > 0.000$) was significantly influencing the overall distribution of trees species along the three forest types. Soil pH and air temperature in the Lower montane, and C in the Montane and Upper montane forests were important environmental factors (Table 4.7).

Table 4.6. Variance explained in the Canonical Correspondence Analysis (CCA) for trees by the first two axes across the forest types in KBR.

Axis		1	2
Total variance (inertia) in the tree species data	8.55		
Sum of the canonical eigen values	2.49		
Sum of non canonical eigen values	6.05		
Canonical eigen value		0.98	0.74
% variance explained		11.49	8.25
Cumulative % variance		11.49	19.73
Probability (Monte Carlo Test)		0.009	0.009
Non-canonical eigen value		0.46	0.36
% variance explained		5.38	4.26
Cumulative % variance		5.38	9.64

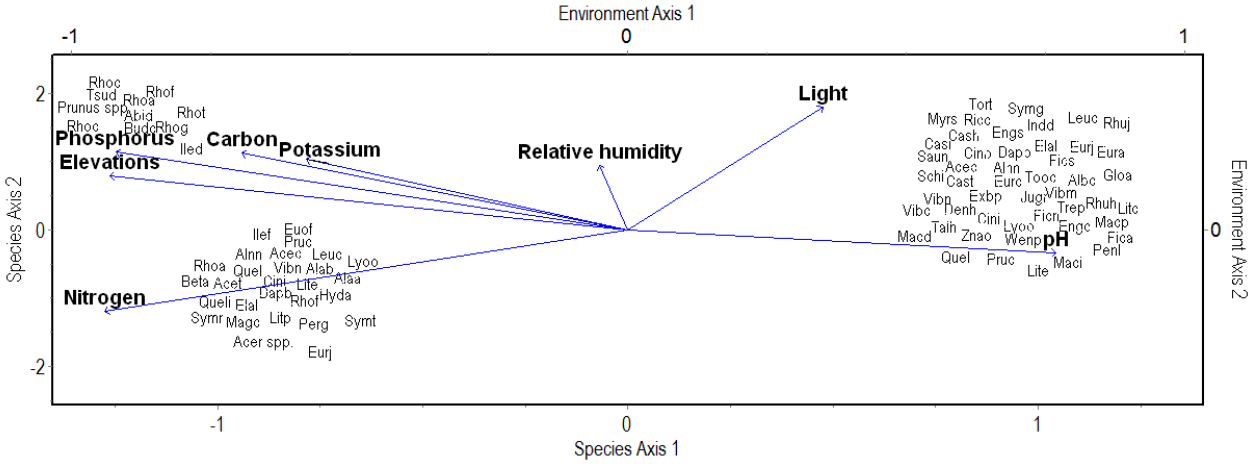


Figure 4.4. CCA ordination diagram using abundance data of 78 tree species and microenvironmental variables from 60 plots across the three forest types in KBR. The environmental variables are indicated by arrow and length of the arrow indicates the strength of the correlation. For clarity, species codes have been used which consist of the first three letter of the genus and the first letter of the species name; Acet-*Acer thomsonii*; Alaa-*Alangium alpinum*; Alab-*Alangium begoniaefolium*; Albc-*Albizia chinensis*; Alnn-*Alnus nepalensis*; Beta-*Betula alnoides*; Budc-*Buddleia colvileii*; Cash-*Castanopsis hystrix*; Casi-*Castanopsis indica*; Cast-*Castanopsis tribuloides*; Cinb-*Cinnamomum bejolghota*; Cini-*Cinnamomum impresssinervium*; Dapb-*Daphne bholua*; Dapp-*Daphne papyracea*; Denh-*Dendrocalamus hamiltonii*; Elal-*Elaeocarpus lanceaefolius*; Engc-*Engelhardtia colebrookianum*; Engs-*Engelhardtia spicata*; Evof-*Evodia fraxinifolia*; Eura-*Eurya acuminata*; Eurj-*Eurya japonica*; Exbp-*Exbucklandia populnea*; Fica-*Ficus auriculata*; Ficn-*Ficus neriifolia*; Fics-*Ficus semicordata*; Gloa-*Glochidion acuminatum*; Hyda-*Hydrangea aspera*; Iled-*Ilex dipyrena*; Ilef-*Ilex fragilis*; Indd-*Indigofera dosua*; Jugr-*Juglans regia*; Leuc-*Leucosceptrum canum*; Lite-*Lithocarpus elegans*; Litp-*Lithocarpus pachyphylla*; Litc-*Litsaea cubeba*; Lite-*Litsaea elongata*; Lyoo-*Lyonia ovalifolia*; Macd-*Macaranga denticulata*; Maci-*Macaranga indica*; Macp-*Macaranga pustulata*; Magc-*Magnolia campbellii*; Myrs-*Myrsine semiserrata*; Penl-*Pentapanax leschenaultii*; Perg-*Persea gammieana*; Pruc-*Prunus cerasoides*; Pruc-*Prunus cornuta*; Prunus spp.; Quel-*Quercus lamellosa*; Quel-*Quercus lineata*; Rhoa-*Rhododendron arboreum*; Rhoc-*Rhododendron campanulatum*; Rhoc-*Rhododendron cinnabarinum*; Rhof-*Rhododendron falconeri*; Rhog-*Rhododendron grande*; Rhot-*Rhododendron thomsonii*; Rhoa-*Rhododendron arboreum*; Rhuh-*Rhus hookeri*; Rhuj-*Rhus javanica*; Ricc-*Ricinus communis*; Saun-*Saurauia napaulensis*; Schi-*Schefflera impressa*; Symg-*Symplocos glomerata*; Symr-*Symplocos ramosissima*; Symt-*Symplocos theifolia*; Tauh-*Taulauma hodgsonii*; Tooc-*Toona ciliata*; Tort-*Toricellia tiliifolia*; Trep-*Trevesia palmata*; Tsud-*Tsuga dumosa*; Vibc-*Viburnum cylindricum*; Vibn-*Viburnum nervosum*; Wenp-*Wendlendia paniculata*; Zano-*Zanthoxyllum oxyphyllum*

Table 4.7. Results of forward stepwise multiple regression analysis of environmental variables with tree density in three forest types in KBR.

Environmental variables	Coefficient	Standard Coefficient	Standard Error	t	Probability>t	Constant
Lower montane						
pH	-3.553	-0.781	0.683	-5.202	0.000	2.688
Air temperature	1.254	0.376	0.501	2.504	0.023	
Montane						
Carbon	4.507	0.564	1.557	2.895	0.010	-2.235
Upper montane						
Carbon	3.414	0.557	1.201	2.842	0.011	-1.331

Shrubs

Thirty-eight species belonging to 35 genera and 17 families were recorded from the three forest types. The number of species was highest in the Lower montane forests (33) followed by the Montane (6) and Upper montane (6) forests (Table 4.2). Shrub species richness decreased with increasing elevation (R = -0.20; $P < 0.05$). The Shannon diversity indices also decreased with increasing elevation (H = 3.28, 1.55, 1.72 respectively in three forests), while, evenness index followed a reversed trend (J = 0.44, 0.80 and 0.96) in the Lower montane, Montane and Upper montane forests. The dominance index (D) also decreased with elevation (D = 24.38, 3.37, 5.36). Fisher's α diversity was greatest in the Lower montane forests, followed by Upper montane and Montane forests (Table 4.3). β-diversity was highest between the Lower and Upper montane (0.84), Montane and Upper montane (0.83) forests stands. Lower montane and Montane forests had the lowest β-diversity value of (0.79).

Rosaceae was the dominant family in the Lower montane (15.6%) and Montane (10%) forests. Ericaceae with 50% of the total species dominated the shrub community in the Upper montane forests.

The three forest types differed significantly in shrub species composition (Clark's R statistic = 0.63, $P < 0.001$). Species dissimilarity between Lower montane and Montane, Lower and Upper montane, and Montane and Upper montane forests was 99.3, 99.1 and 99.5%, respectively (Table 4.8). *Edgeworthia gardneri* and *Aconogonum molle*

were found in the Lower montane and Montane forests only. *Berberis* spp., *Rosa sericea* were confined to Montane and Upper montane forests (Annexure 1).

Table 4.8. Similarity test values of ANOSIM and SIMPER on the sampled sites for Lower montane, Montane and Upper montane forests areas. The ANOSIM 'R value' is the statistical value of similarity within each forest stand with a probability of 0.001.

All stands together				
R value	P value			
0.63	0.001			
Stand wise test (No. of quadrats)				
1st group	2nd group	P value	ANOSIM (R value)	SIMPER (average dissimilarity)
Lower montane	Montane	0.001	0.71	99.29
Lower montane	Upper montane	0.001	0.66	99.34
Montane	Upper montane	0.001	0.54	99.46

The density of shrub decreased from 319 stems ha^{-1} in the Lower montane forests to 101 stems ha^{-1} in the Montane and increased to 234 stems ha^{-1} in the Upper montane forests ($F = 11.82$, $P < 0.001$) (Table 4.2).

With an increase in elevation, the shrub species-abundance curves exhibited higher dominance (Figure 4.5). Three dominant and co-dominant shrub species, *Elsholtzia flava*, *Luculia gratissima* and *Thysaenolaena maxima* shared 37% of the total IVI values in the Lower montane forests while the corresponding figures for Montane forests was much higher at 61% which was shared by *Rosa sericea*. It further increased to 80% in the Upper montane forests which were shared by *Rhododendron anthopogon* and *R. lepidotum* (Annexure 1).

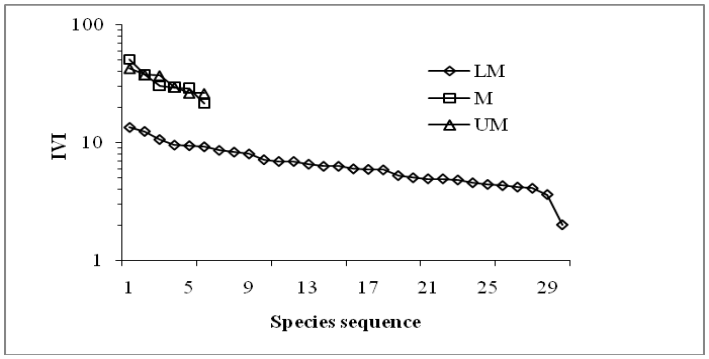


Figure 4.5. Dominance-diversity curve for shrub species in Lower montane (LM), Montane (M) and Upper montane (UM) forests in KBR.

The dominant shrub species in the Lower montane forests were *Elsholtzia flava*, *Melastoma normale*, *Oxyspora paniculata*, *Rubus ellipticus*, *Rubus mollucanus*, *Sambucus adnata* and *Thysaenolaena maxima* (together >55%). *Arundinaria maling*, *Deutzia compacta* and *Rosa sericea* were dominant in the Montane forest stands (together >37%) whereas, *Berberis* spp., *Rhododendron anthopogon*, *R. lepidotum* were dominant in the Upper montane forests (together 60%).

Microenvironmental factors related to shrub abundance

Shrub species-environment relationship across the forests was poorly explained as the first two canonical axes accounted for 10.5 % and 7.02 % of the total variance. However, Monte Carlo randomisation test with 100 iterations yielded a probability level of 0.009 for test of significance (Table 4.9). N, C, pH, P and K were strongly correlated with the first CCA axis and therefore were important determinants of shrub species distribution across the forest types (Figure 4.6).

Table 4.9. Variance explained in the Canonical Correspondence Analysis (CCA) for shrubs by the first two axes across the three forest types in KBR.

Axis		1	2
Total variance (inertia) in the tree species data	9.36		
Sum of the canonical eigen values	2.82		
Sum of non canonical eigen values	6.54		
Canonical eigen value		0.98	0.66
% variance explained		10.45	7.02
Cumulative % variance		10.45	17.47
Probability (Monte Carlo Test)		0.009	0.009
Non-canonical eigen value		0.81	0.75
% variance explained		8.63	8.04
Cumulative % variance		8.63	16.68

CCA produced an ordination of all 38 shrub species that showed the inferred ranking of the species along the environmental variables. In the Lower montane forests, *Boehmeria macrophylla*, *Clerodendrum colebrokianum*, and *Dicranopteris linearis* with high first axis species scores dominated the areas with high soil pH. Conversely, *Boehmeria platyphylla*, *Debregeasia longifolia*, and *Edgeworthia gardneri* occupied low

soil pH areas. In the Montane forests, *Arundinaria maling*, *Berberis sikimensis*, *Edgeworthia gardneri* with high first axis species scores were strongly associated with high soil C, P and K level, while *Rosa sericea* and *Zanthoxylum oxyphyllum* were confined to low soil C, P and K areas. In the Upper montane forests, *Juniperus recurva*, *Rhododendron lepidotum*, and *Rhododendron setosum* were dominant in high soil N environment, while *Arundinaria maling* and *Berberis sikkimensis* were abundant in low soil N areas (Figure 4.6).

The relationship between microenvironmental variables and shrub species density as revealed by stepwise forward multiple regression analysis indicated that N ($P > 0.000$) is significant in influencing the overall distribution of shrub species across the three forests. In addition, light and soil C in the Lower montane, and soil C in the Montane and Upper montane forests were important environmental variables (Table 4.10).

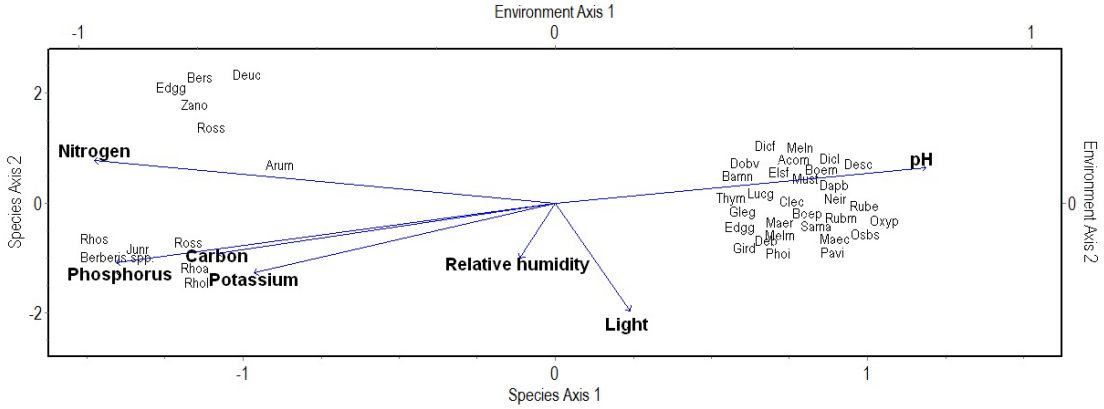


Figure 4.6. CCA ordination diagram using abundance data of 38 shrub species and microenvironmental variables from 60 plots across three forest types in KBR. The environmental variables are indicated by arrow and length of the arrow indicates the strength of the correlation. For clarity, species codes have been used which consist of the first three letter of the genus and the first letter of the species name; Acom-*Aconogonum molle*; Arum-*Arundinaria maling*; Bamn-*Bambusa nutans*; Bers-*Berberis sikkimensis*; Boem-*Boehmeria macrophylla*; Boep-*Boehmeria platyphylla*; Cleb-*Clerodendrum colebrookianum*; Dapb-*Daphne bholua*; Debl-*Debregeasia longifolia*; Desc-*Desmodium confertum*; Deuc-*Deutzia compacta*; Dicf-*Dichroa febrifuga*; Dicl-*Dicranopteris linearis*; Dobv-*Dobinea vulgaris*; Edgg-*Edgeworthia gardneri*; Elsf-*Elsholtzia flava*; Gird-*Girardinia diversifolia*; Gleg-*Gleichenia glauca*; Junr-*Juniperus recurva*; Lucg-*Luculia gratissima*; Maec-*Maesa chisia*; Maer-*Maesa ramentacea*; Melm-*Melastoma malabathricum*; Meln-*Melastoma normale*; Muet-*Mussaenda treutleri*; Nelr-*Neillia rubriflora*; Osbs-*Osbeckia sikkimensis*; Oxyp-*Oxyspora paniculata*; Pavi-*Pavetta indica*; Phoi-*Photinia integrifolia*; Rhoa-*Rhododendron anthopogon*; Rhol-*Rhododendron lepidotum*; Rhos-*Rhododendron setosum*; Ross-*Rosa sericea*; Rube-*Rubus ellipticus*; Rubm-*Rubus mollucanus*; Sama-*Sambucus adnata*; Thym-*Thysaenolaena maxima*; Zano-*Zanthoxylum oxyphyllum*.

Table 4.10. Results of forward stepwise multiple regression analysis of environmental variables with shrub density in three forest types in KBR.

Environmental variables	Coefficient	Standard Coefficient	Standard Error	t	Probability>t	Constant
Lower montane						
Light	-0.356	-0.529	0.068	-5.205	0.000	-2.251
Carbon	-2.503	-0.405	0.586	-4.271	0.001	
Montane						
Carbon	6.380	0.597	2.019	3.160	0.005	-3.820
Upper montane						
Carbon	4.210	0.428	1.820	2.314	0.033	-5.485

Herbs

One hundred and thirty three species belonging to 97 genera and 49 families were recorded from the three forest types. The number of species was highest in the Lower montane forests (61) followed by the Montane (52) and Upper montane (39) forests (Table 4.2). Herb species richness decreased with increasing elevation (R = -0.51; P < 0.05). The species diversity indices also decreased with increasing elevation (H = 3.93, 3.52, 3.23 and J = 0.95, 0.91 and 0.88 in the Lower montane, Montane and Upper montane forests. The dominance index (D) also followed the same trend (D = 45.4, 28.1, 14.3). Fisher's α diversity was greatest in the Lower montane forests, followed by Upper montane and Montane forests (Table 4.3).

β-diversity was highest between the Lower montane and Upper montane (0.94), followed by Montane and Upper montane (0.85) forests stands. Lower montane and Montane forests had the lowest β-diversity value of (0.82).

Asteraceae and Poaceae were the dominant families in the Lower montane forests (21.3% and 6.6%, respectively). Urticaceae, Lamiaceae and Polygonaceae each with 11.5%, 7.7% and 9.6% of the total species composition dominated the Montane forests. Asteraceae, Polygonaceae each with 10.5% and 13.2% of the total species, dominated the herb community in the Upper montane forests.

The three forest types differed significantly in herb species composition (Clark's R statistic = 0.95, P < 0.001). Species dissimilarity between Lower montane and

Montane, Lower and Upper montane, and Montane and Upper montane forests was 99.1, 99.02 and 99.1%, respectively (Table 4.11). *Anaphalis triplivnervis* were found in all the three forest types. *Arundinella bengalensis*, *Athyrium rubicaule*, *Cyanotis vaga*, *Dryopteris barbigera* and *Erigeron karvinskianus* were confined to the Lower montane and Montane forests. *Arisaema griffithii*, *Fragaria nubicola*, *Galium elegans* and *Hemiphragma heterophyllum* were found only in the Montane and Upper montane forests (Annexure 1).

Table 4.11. Similarity test values of ANOSIM and SIMPER on the sampled sites for Lower montane, Montane and Upper montane forests areas. The ANOSIM 'R value' is the statistical value of similarity within each forest stands with a probability of 0.001.

All stands together				
R value	P value			
0.95	0.001			
Stand wise test (No. of quadrats)				
1st group	2nd group	P value	ANOSIM (R value)	SIMPER (average dissimilarity)
Lower montane	Montane	0.001	0.95	99.05
Lower montane	Upper montane	0.001	0.98	99.02
Montane	Upper montane	0.001	0.93	99.07

The density of herbaceous species did not differ significantly across the forests ($F = 0.90$, $P = 0.44$). Highest density was in the Montane forests (711500 individual ha^{-1}), followed by the Upper montane and Lower montane forest stands (625000 individuals ha^{-1} and 609500 individuals ha^{-1} respectively) (Table 4.2).

With an increase in elevation, the herb species-abundance curves exhibited higher dominance (Figure 4.7). Four dominant and co-dominant herb species, *Bidens pilosa*, *B. biternata*, *Carex filicina*, and *Elsholtzia blanda* together shared 27.5% of the total IVI values in the Lower montane forests. *Fragaria nubicola*, *Persicaria runcinata*, *Phlomis bracteosa* together shared 31.2% of the total IVI in the Montane forests while the corresponding figure for Upper montane forests was much greater at 45.5%, which was shared by *Anaphalis triplinervis*, *Juncus* spp., and *Poa alpina* (Annexure 1).

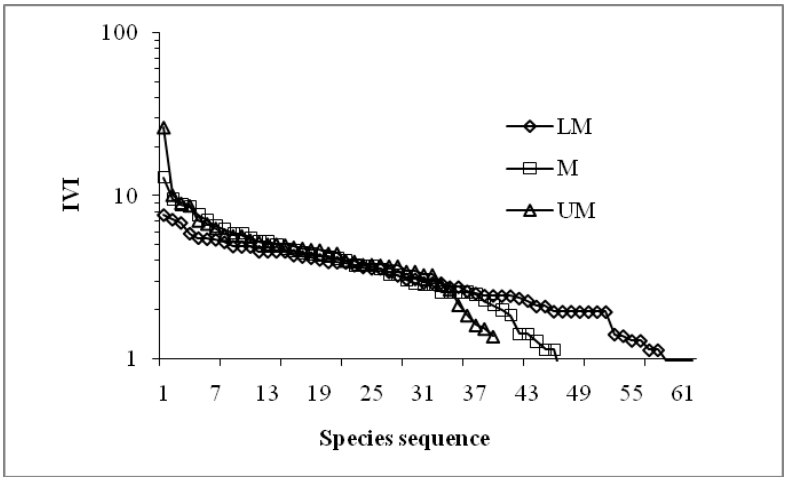


Figure 4.7. Dominance-diversity curve of herb species in Lower montane (LM), Montane (M) and Upper montane (UM) forests in KBR.

Microenvironmental factors related to herb abundance

Herb species-environment relationship across the forests was poorly explained as the first two canonical axes accounted for 8.6 % and 6.1 % of the total variance. However, Monte Carlo randomisation test with 100 iterations has also yielded a strong probability of 0.009 for both the axes indicating that the axes have explained a significant part of the variability in the species abundance data (Table 4.12). N, C, pH, P and K were strongly correlated with the first CCA axis and therefore were important determinants of herb species distribution across the forest types (Figure 4.8).

Table 4.12. Variance explained in the Canonical Correspondence Analysis (CCA) for herbs by the first two axes across the forest types in KBR.

Axis		1	2
Total variance (inertia) in the tree species data	11.25		
Sum of the canonical eigen values	2.74		
Sum of non canonical eigen values	8.52		
Canonical eigen value		0.97	0.68
% variance explained		8.63	6.02
Cumulative % variance		8.63	14.6
Probability (Monte Carlo Test)		0.009	0.009
Non-canonical eigen value		0.55	0.46
% variance explained		4.90	4.09
Cumulative % variance		4.90	9.10

CCA produced an ordination of all 133 herb species that showed the inferred ranking of the species along the above five environmental variables. The ordination plot

shows the relative position of the species along the line of environmental vectors depicting species environmental preferences. In the Lower montane forests, *Boehmeria platyphylla*, *Cyanodon dactylon*, *Paspalum destichum*, *Pilea scripta*, and *Pogonatherum paniceum* with high first axis species scores dominated the areas with high soil pH. Conversely, *Arundinella bengalensis*, *Cuphea balsamona*, *Cyanotis vaga*, and *Desmodium multiflorum* occupied low soil pH areas. In the Montane forests, *Impatiens urticifolia*, *Sanicula elata*, *Oxalis acetosella*, *Viola biflora* with high first axis species scores were associated strongly with high soil N level, while *Ainsliaea aptera* and *Arundinella bengalensis* were confined to low soil N areas. In the Upper montane forests, *Arisaema jacquemontii*, *Poa himalayana*, and *Potentilla eriocarpa* were dominant in high soil P, K and C environment, while *Juncus* spp., *Megacodon stylophorus*, and *Bistorta affinis* were abundant in low soil P, K and C areas (Figure 4.8).

The relationship between microenvironmental variables and herb species density as shown by stepwise forward multiple regression analysis indicated that P ($P < 0.031$) is significant in influencing the overall distribution of herb species along the three forests. Forest wise, pH and elevations in the Lower montane, and N, P in the Montane and P alone in the Upper montane forests were important (Table 4.13).

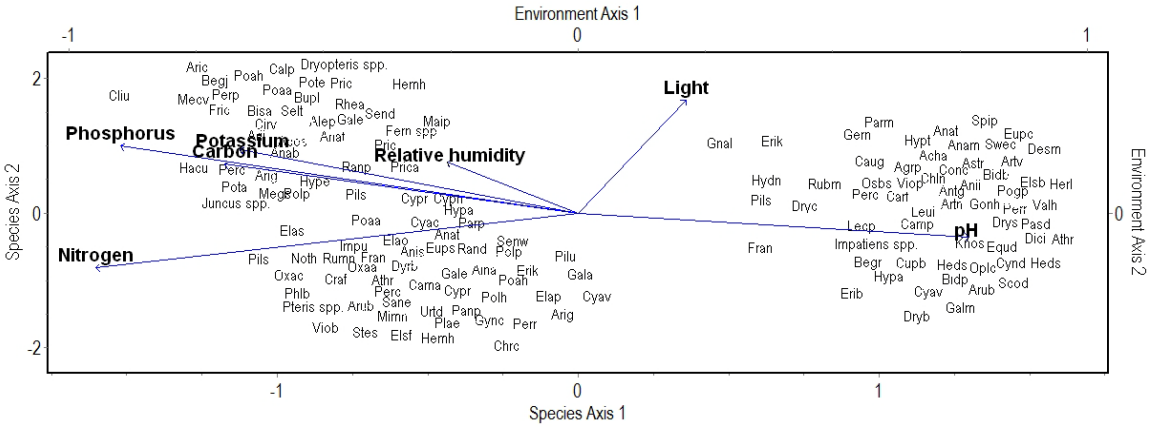


Figure 4.8. CCA ordination diagram using abundance data of 133 herb species and microenvironmental variables from 60 plots across three forest types in KBR. The environmental variables are indicated by arrow and length of the arrow indicates the strength of the correlation. For clarity, species codes have been used which consist of the first three letter of the genus and the first letter of the species name; Acha-*Achyranthes aspera*; Acos-*Aconitum spicatum*; Agrp-*Agrimonia pilosa*; Aina-*Ainsliaea aptera*; Alep-*Aletris pauciflora*; Anab-*Anaphalis busua*; Anam-*Anaphalis margaritacea*; Anat-*Anaphalis triplinervis*; Anas-*Anisadenia saxatilis*; Anii-*Anisomeles indica*; Antg-*Anthogonium gracile*; Aric-*Arisaema concinnum*; Arig-*Arisaema griffithii*; Arij-*Arisaema jacquemontii*; Artn-*Artemisia nilagirica*; Artv-*Artemisia vulgaris*; Arub-*Arundinella bengalensis*; Astr-*Astilbe rivularis*; Athr-*Athyrium rubricaule*; Begj-*Begonia josephii*; Begr-*Begonia rubrella*; Bidb-*Bidens biternata*; Bidp-*Bidens pilosa*; Bisa-*Bistorta affinis*; Bupl-*Bupleureum longicaule*; Calp-*Caltha palustris*; Camp-*Campanula pallida*; Cama-*Campylandra aurantiaca*; Carf-*Carex filicina*; Caug-*Cautleya gracilis*; Chln-*Chlorophytum nepalense*; Chrc-*Chrysosplenium carnosum*; Cirv-*Cirsium verutum*; Cliu-*Clintonia udensis*; Conc-*Coniogramme cautata*; Craf-*Craniotome furcata*; Cupb-*Cuphea balsamona*; Cynv-*Cyanotis vaga*; Cyac-*Cyathula capitata*; Cynd-*Cynodon dactylon*; Cypn-*Cyperus niveus*; Cypr-*Cyperus rotundus*; Desm-*Desmodium multiflorum*; Dici-*Dicrocephala integrifolia*; Dryb-*Dryopteris barbigera*; *Dryopteris* spp.; Elao-*Elatostemma obtusum*; Elap-*Elatostemma platyphylla*; Elas-*Elatostemma sessile*; Elsb-*Elsholtzia blanda*; Elsf-*Elsholtzia fruticosa*; Equd-*Equisetum diffusum*; Erib-*Erogeron bellidioides*; Erik-*Erigeron karvinskianus*; Eupc-*Eupatorium cannabinum*; Eups-*Euphorbia sikkimensis*; Fran-*Fragaria nubicola*; Fric-*Fritillaria cirrhosa*; Gala-*Galium asperifolium*; Gale-*Galium elegans*; Galm-*Galium mullago*; Gern-*Geranium nepalense*; Gnal-*Gnaphalium luteo-album*; Gonh-*Gonostegia hirta*; Gync-*Gynura cusimba*; Hacu-*Hackelia uncinata*; Heds-*Hedychium spicatum*; Heds-*Hedyotis scandans*; Hemh-*Hemiphragma heterophyllum*; Herl-*Herminium lanceum*; Hydn-*Hydrocotyle nepalensis*; Hype-*Hypericum elodeoides*; Hypt-*Hypoestes triflora*; Hypa-*Hypoxis aurea*; *Impatiens* spp.; *Juncus* spp.; Knos-*Knoxia sumatrensis*; Lecp-*Lecanthus peduncularis*; Leui-*Leucostegia immerse*; Miap-*Maianthemum purpureum*; Mecv-*Meconopsis villosa*; Megs-*Megacodon stylophorus*; Mimn-*Mimulus nepalensis*; Noth-*Notochaete hamosa*; Oplc-*Oplismenus compositus*; Osbs-*Osbeckia stellata*; Oxaa-*Oxalis acetosella*; Oxac-*Oxalis corniculata*; Panp-*Panax pseudoginseng*; Parm-*Pardavallodes multidentum*; Parp-*Paris polyphylla*; Pasd-*Paspalum destichum*; Perc-*Persicaria capitata*; Perc-*Persicaria chinense*; Perp-*Persicaria polystachya*; Perr-*Persicaria runcinata*; Phlb-*Phlomis bracteosa*; Pils-*Pilea scripta*; Pils-*Pilea symmeria*; Pilu-*Pilea umbrosa*; Plae-*Plantago erosa*; Poaa-*Poa annua*; Poah-*Poa himalayana*; Pogp-*Pogonatherum paniceum*; Polh-*Polygonum hydropiper*; Polp-*Polygonum plebium*; Polp-*Polystichum prescotianum*; Pota-*Potentilla arbuscula*; Pote-*Potentilla eriocarpa*; Pric-*Primula capitata*; Pric-*Primula caveana*; *Pteris* spp.; Rand-*Ranunculus diffusus*; Ranp-*Ranunculus pulchellus*; Rhea-*Rheum acuminatum*; Rubm-*Rubus mollucanus*; Rumn-*Rumex nepalensis*; Sane-*Sanicula elata*; Scod-*Scoparia dulcis*; Selt-*Selenium tenuifolium*; Send-*Senecio diversifolius*; Senw-*Senecio wallichii*; Spip-*Spilanthes paniculatus*; Stes-*Stellaria sikkimensis*; Swec-*Swertia chirayita*; Urtd-*Urtica dioica*; Valh-*Valeriana hardwickii*; Viob-*Viola biflora*; Viop-*Viola pilosa*.

Table 4.13. Results of forward stepwise multiple regression analysis of environmental variables with herb density in the three forest types in KBR.

Environmental variables	Coefficient	Standard Coefficient	Standard Error	t	Probability>t	Constant
Lower montane						
pH	-3.130	-0.568	0.745	-4.203	0.001	-3.913
Elevations	2.452	0.533	0.622	3.942	0.001	
Montane						
Nitrogen	5.046	0.816	1.048	4.815	0.000	-10.279
Phosphorus	1.801	0.517	0.590	3.052	0.007	
Upper montane						
Phosphorus	3.052	0.485	1.298	2.352	0.030	-4.055

Epiphytes

Ninety two epiphyte species belonging to 57 genera and 31 families were recorded in the three forest types. The number of species was highest in the Lower montane forests (60 species) followed by the Montane (44 species) and the Upper montane forests stands (19 species) (Table 4.14). Epiphytic richness decreased with increasing elevation (R = -0.51, $P < 0.05$). The species diversity indices also decreased with increasing elevation (H = 3.21, 3.07, 2.39) while evenness index remain more or less same in all the forests (J = 0.79, 0.81 and 0.81). The dominance index (D) (D = 13.04, 13.57) was similar in Lower montane and Montane forests while it was least in the Upper montane forests (D = 7.50). Fisher's α diversity was greatest in the Lower montane forests, followed by Upper montane and Montane forests (Table 4.14).

Table 4.14. General epiphytic diversity patterns in three forest types (values in parenthesis indicate Jackknife standard error).

Parameters	**Lower montane**	**Montane**	**Upper montane**	**Total**
Genera	41	34	13	59
Species	60	44	19	92
Family	20	21	8	32
Species/family ratio	3.00	2.10	2.38	2.97
Fisher's alpha diversity	13.83(0.39)	9.51(0.31)	3.93(0.21)	18.80(0.38)
Shannon-Wiener index	3.21(0.84)	3.07(0.09)	2.39(0.11)	3.57(0.06)
Simpson's dominance index	13.04(2.59)	13.57(1.96)	7.50(1.06)	19.47(1.82)
Pielou J (all samples)	0.79(0.02)	0.81(0.03)	0.81 (0.04)	0.78(0.01)

The species: family ratio in the Lower montane (3.00) was also higher compared to Montane (2.10) and Upper montane (2.38) epiphyte species. The Pteridophytic family, Polypodiaceae (21 species, 21.9%) was the dominant family followed by Orchidaceae (19 species, 19.8%) across the forest types. It had 67% of total species in the Lower montane, 50.5% in the Montane and 20.8% in the Upper montane forests. Pteridophytic families were dominant in the Upper montane (66.7%) and the Montane (58.3%) forest stands.

β-diversity was highest between the Lower and the Upper montane (0.90), followed by Lower montane and Montane (0.65) forests. Upper montane and Montane forests had the least β-diversity value of (0.62).

The three forest types differed significantly in epiphyte species composition (Clark's R statistic = 0.47, $P < 0.001$). Species dissimilarity between Lower montane and Montane, Lower and Upper montane, and Montane and Upper montane forests was 95.05, 97.05 and 89.68%, respectively (Table 4.15).

In general, pteridophytic species were dominant epiphytic community in all the forest stands. Dominant species from the Lower montane forests stand were *Hoya linearis*, *Lepisorus nudus*, *Mecodium* spp., *Peperomia tetraphylla* and *Vittaria elongata*. *Lepisorus nudus*, *Pleione humilis*, *Vaccinium retusum* and *Vittaria elongata* were dominant in the Montane forests. While in the Upper montane forests, the dominant species were *Onychium* spp., *Cystopteris sudetica*, *Pleione humilis*, *Phymatopteris malacodon* and *Vaccinium nummularia* (Annexure 2).

The prevalence of epiphytes like *Hoya linearis* (14.1%), *Pleione humilis* (9.3%) and *Vittaria elongataa* (8.1%) were the main contributors to the dissimilarity between the Lower montane and the Montane forests. A high abundance of *Hoya linearis* (15.6%), *Lepisorus nudus* (7.9%) and *Onychium* spp., (1.9%) was the cause of dissimilarity between the Lower montane and the Upper montane forests. Between the Montane and

Upper montane forests, the main cause of dissimilarity was the high abundance of *Pleione humilis* (12.5%), *Onychium* spp., (8.9%) and *Vittaria elongata* (8.8%).

Table 4.15. Similarity test values of ANOSIM and SIMPER in the sampled sites for Lower montane, Montane and Upper montane forests. The ANOSIM 'R value' is the statistical value of similarity within each forest stands with a probability of 0.001.

All stands together				
R value	P value			
0.47	0.001			
Stand wise test (No. of quadrats)				
1st group	2nd group	P value	ANOSIM(R value)	SIMPER(average dissimilarity)
Lower montane	Montane	0.001	0.49	95.05
Lower montane	Upper montane	0.001	0.63	97.05
Montane	Upper montane	0.001	0.25	89.68

The density of epiphytes decreased significantly across the forest types (F = 8.53, P < 0.001). It was highest in the Lower montane forests (5200 individuals 20 tree^{-1}), followed by the Montane and the Upper montane forests (4830 and 2390 individuals 20 tree^{-1} respectively) (Annexure 2).

With an increase in elevation, the epiphyte species-abundance curves exhibited higher dominance (Figure 4.9). Dominance-diversity curves for epiphyte species showed that most IVI values in the Upper montane forests were concentrated in two species viz. *Cystopteris sudetica* and *Onychium* spp. In the Lower montane and Montane forests IVI was distributed more equitably among all the species than the Upper montane forests (Figure 4.9). The three dominant and codominant epiphyte species in the Lower montane forests viz. *Hoya linearis, Lepisorus nudus* and *Vittaria elongata* together shared 57.9 % of the total IVI values. In the Montane forests the three dominants species viz. *V. elongata, L. nudus* and *Pleione humilis* together shared 59 % of the total IVI. In the Upper montane forests, *Cystopteris sudetica, Onychium* spp., and *P. humilis* shared 90.6% of the total IVI. The Pteridophytic family Polypodiaceae (21 species, 21.9 %) was the dominant family followed by Orchidaceae (19 species, 19.8 %) across the forest types. Pteridophytic families were also dominant in the Upper montane (66.7 %) and Montane (58.3 %) forests (Annexure 2).

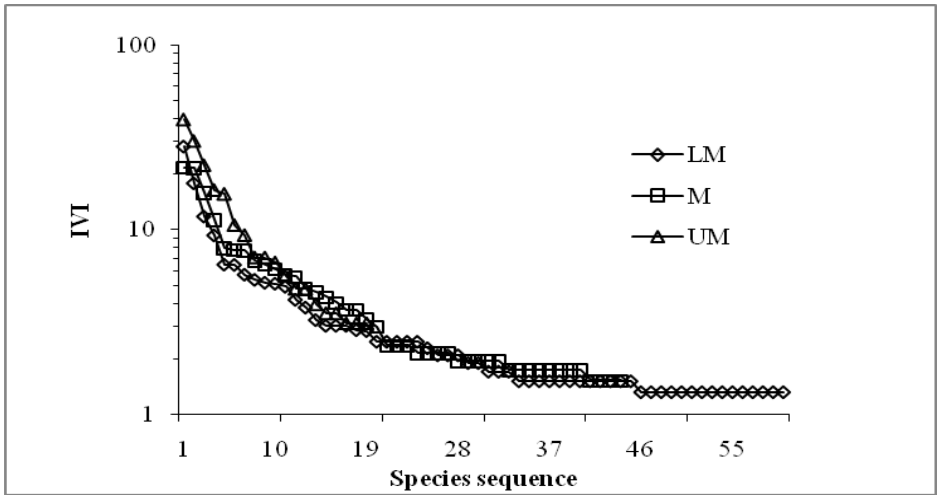


Figure 4.9. Dominance-diversity curves for epiphyte species in Lower montane (LM), Montane (M) and Upper montane (UM) forests in KBR.

Non-Metric Multi Dimensional Scaling

The nMDS ordination resulted in a three dimensional solution with a moderately low stress (2D stress = 0.148) and a small amount of overlap between cluster group scores. The nMDS of three forest types using the abundance data of the species in this case showed that the Montane epiphytic species are more common to both the Lower and the Upper montane epiphytic species (Figure 4.10).

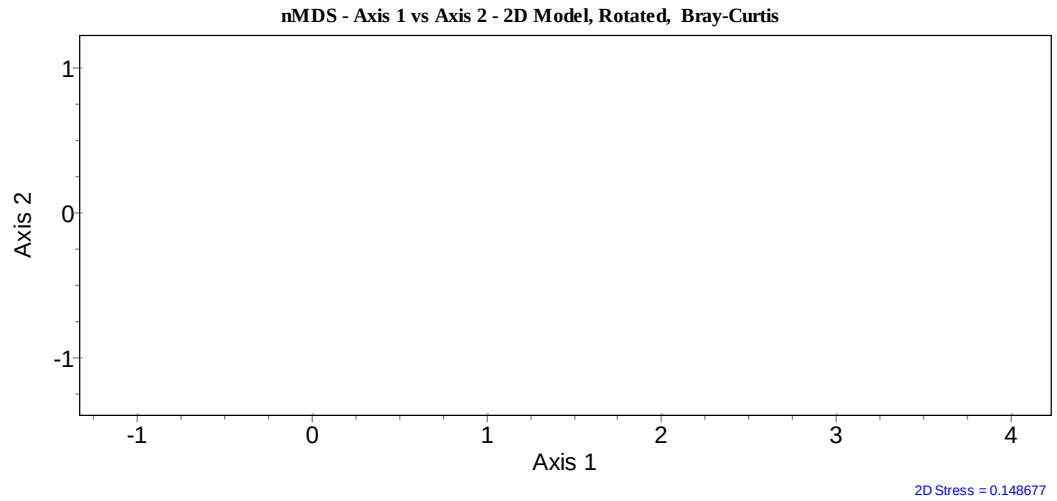


Figure 4.10. Non metric multi dimensional scaling (nMDS) plot for epiphytic species using Bray-Curtis index of similarity of their abundance data from Lower montane (LM), Montane (M) and Upper montane (UM) forests in KBR. The classification showing the distribution pattern of the species in the three forest types is distinct (T-indicates tree host).

All the species were classified by lifeform (growth habit) and by taxonomic group (Figure 4.11). Most epiphytes belonged to Caespitose followed by pendent life form.

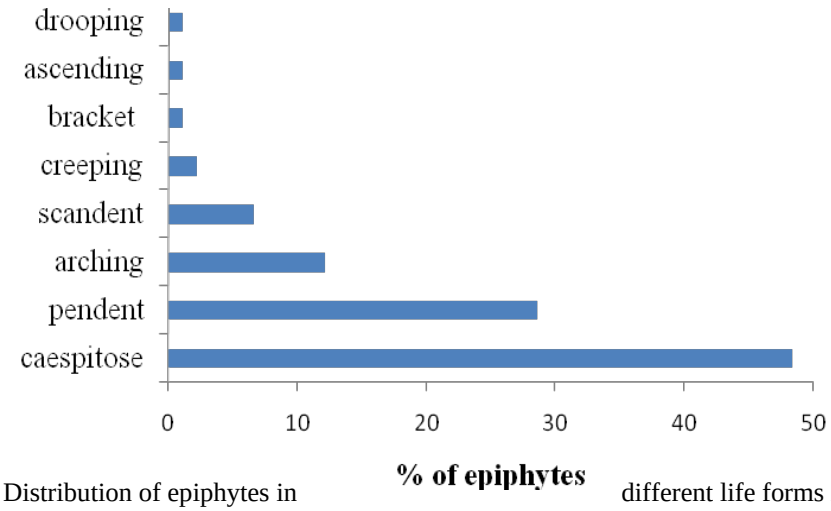


Figure 4.11. Distribution of epiphytes in different life forms in KBR.

According to habit or taxonomic classification, epiphytes consisted of three different groups. The pteridophytic community was the highest with 40.2%, followed by herbaceous epiphytes (33.7%), shrubs and climbers (19.6%) and under tree (5.4%). True epiphytes need hosts where they start and saturate their life cycle and do not destroy or overcast the host. Whereas false epiphytes grow on selective hosts and later on become independent by sending roots to the ground (Table 4.16). True epiphytes were mostly herbaceous epiphytes such as *Peperomia tetraphylla*, *P.heyneana*, *Pilea racemosa*, *P. approximata*, *P. ternifolia*, ferns and *Begonia gemmipara*.

The shrubby epiphytes include *Aeschynanthus* spp., *Hoya* spp., *Hymenopogon parasiticus*, *Hymenodictyon excelsum*, *H. flaccidum*, *Piper mullesua*, *Lysionotus atropurpureus*, *Vaccninium vacciniaceum*, *Agapetes serpens*, *A. sikkimensis*, and *R. vaccinioides*. The climbing epiphytes are *Hoya fusca*, *Piper longum*, *Schefflera benghalensis* and some *Ficus* species.The epiphytic trees are mostly false epiphytes. The examples are *Ficus* spp., *Hymenopogon parasiticus*, *Macropanax undulatus*, *M. dispermus*, *Pentapanax fragrans*, *P. racemosus*, *Rhododendron dalhousiae*, *Vaccinium* spp., and *Wightia speciosissima*.

Table 4.16. Epiphytic flora of KBR according to habit and extent of their dependency and interaction with the host plants.

Nature of epiphytes	Number of species	% of Epiphytes
True (obligatory)	71	77.2
False (accidental)	9	9.8
Intermediate Facultative	11	12

Epiphyte colonization

Regression analysis showed that host tree girth and height class had significant positive correlation with epiphyte proportion (R = 0.48, P < 0.007; R = 0.98 and P < 0.007 for girth and height class respectively). The abundance of epiphytes with increasing girth and height class indicates that large and taller trees supported higher number of epiphytes species in all the forest types (Figure 4.12).

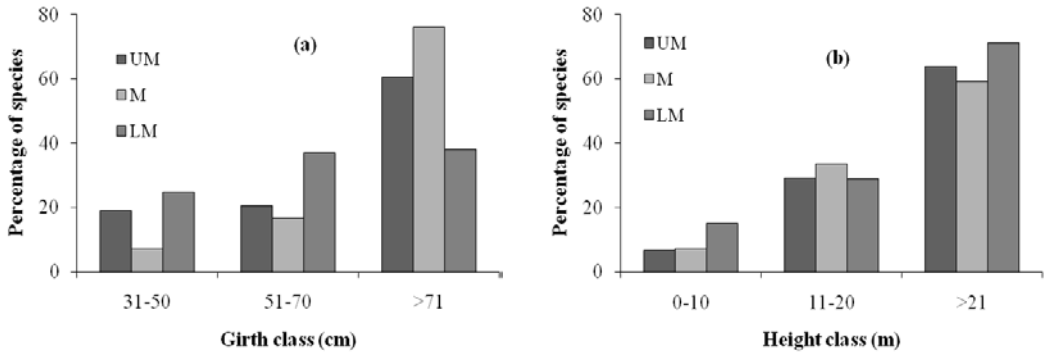


Figure 4.12. Distribution of **e**piphytic species in different girth (a) and height classes (b) of host trees in the Upper montane (UM), Montane (M), and Lower montane (LM) forests in KBR.

Microenvironmental factors related to epiphyte distribution

The epiphyte species-environment relationship across the forests was poorly explained as the first two canonical axes accounted for 6.5% and 3.8% of the total variance. However, Monte Carlo randomisation test with 100 iterations yielded significant (P < 0.009) result indicating that the axes have explained an acceptable proportion of the variability in the species abundance data (Table 4.17). Air temperature and the proxy variable 'elevation' were strongly correlated with the first CCA axis and therefore were important determinants of epiphyte species distribution across the forest types (Figure 4.13).

CCA produced an ordination of all 92 epiphyte species that showed the inferred ranking of the species along the four environmental variables. The ordination plot shows the relative position of the species along the line of environmental vectors depicting species environmental preferences. In the Lower montane forests, *Ficus* spp., *Hoya fusca*, *Remusatia hookeriana*, and *Vandopsis undulata* with high first axis species scores were correlated with elevation. Conversely, *Davallia bullata*, *Medinilla himalayana*, and *Pyrrosia lehmanii* were correlated with RH in second axis. In the Montane forests, *Bulbophyllum reptans*, *Lepisorus kashyapii*, *Lepisorus nudus*, and *Pilea lineolatum*, with high second axis species scores were associated strongly with high light level and RH, while *Codonopsis purpurea*, *Didymocarpus aromaticus*, *Rhododendron pumilum* and *Utricularia multicaulis* were confined to low light and RH areas. In the Upper montane forests, *Cystopteris sudetica*, *Phymatopteris malacodon*, and *Pholidotia* spp., were dominant in high air temperature environment, while *Arthomeris wallichii*, *Lepisorus scolopendrinus* and *Vaccinium nummularia* were abundant in low air temperature (Figure 4.13).

Table 4.17. Variance explained in the Canonical Correspondence Analysis (CCA) for epiphyte by the first two axes across the forest types in KBR.

Axis		1	2
Total variance (inertia) in the epiphyte species data	10.22		
Sum of the canonical eigen values	1.62		
Sum of non canonical eigen values	8.61		
Canonical eigen value		0.67	0.39
% variance explained		6.50	3.77
Cumulative % variance		6.50	10.27
Probability (Monte Carlo Test)		0.009	0.009
Non-canonical eigen value		0.69	0.64
% variance explained		6.77	6.25
Cumulative % variance		6.77	13.02

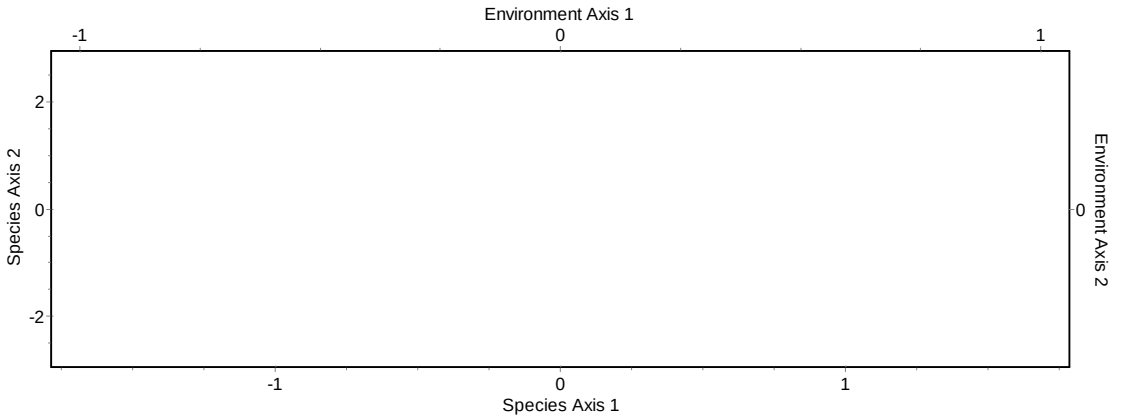


Figure 4.13. CCA ordination diagram using abundance data of 92 epiphyte species and microenvironmental variables from 60 plots across three forest types in KBR. The environmental variables are indicated by arrow and length of the arrow indicates the strength of the correlation. For clarity, species codes have been used which consist of the first three letter of the genus and the first letter of the species name; Aesb-*Aeschynanthus bracteatus*; Aesh-*Aeschynanthus hookeri*; Aess-*Aeschynanthus sikkimensis*; Agah-*Agapetes hookeri*; Agai-*Agapetes incurvata*; Agas-*Agapetes serpens*; Agrb-*Agrostophyllum brevipes* Agrc-*Agrostophyllum callosum*; Arth-*Arthomeris himalayensis*; Artl-*Arthomeris lehmanii*; Artw-*Arthomeris wallichiana*; Aspe-*Asplenium ensiforme*; Bels-*Belvisia spicata*; Bulc-*Bulbophyllum cauliflorum*; Bulr-*Bulbophyllum reptans*; Caug-*Cautleya gracilis*; Chef-*Cheilanthes formosana*; Codp-*Codonopsis purpurea*; Coec-*Coelogynae corymbosa*; Coeo-*Coelogynae ochracea*; Cyss-*Cystopteris sudetica*; Davb-*Davallia bullata*; Dena-*Dendrobium amoenum*; Denl-*Dendrobium longicornu*; Denn-*Dendrobium nobile*; Dida-*Didymocarpus aromaticus*; Dido-*Didymocarpus oblongus*; Dryp-*Drynaria propinqua*; Eric-*Eria coronaria*; Eris-*Eria spicata*; *Ficus* spp.; Gloh-*Globa hookeri*; Gonp-*Gonatanthus pumilus*; Hoyf-*Hoya fusca*; Hoyl-*Hoya lanceolata*; Hoyli-*Hoya linearis*; Hymp-*Hymenopogon parasiticus*; Lepk-*Lepisorus kashyapii*; Lepn-*Lepisorus nudus*; Lepsc-*Lepisorus scolopendrinus*; Leps-*Lepisorus sesquepedalis*; Leui-*Leucostegia immersa*; Lino-*Lindsaea odorata*; Lipp-*Lipparis perpusilla*; Loxi-*Loxogramme involuta*; *Lycopodium* spp.; Lyss-*Lysionotus serratus*; Macu-*Macropanax undulatum*; *Mecodium* spp.; Medh-*Medinilla himalayana*; Micm-*Microsorium membranaceum*; Micp-*Microsorium puctatum*; Nepc-*Nephrolepis cordifolia*; Olew-*Oleandra wallichii*; *Onychium* spp.; Oto-*Otochilus alba*; Penl-*Pentapanax leschenaultii*; Penr-*Pentapanax racemosus*; Peph-*Peperomia heyneana*; Pept-*Peperomia tetraphylla*; Phat-*Phalaenopsis tainitis*; Phlp-*Phlegmariurus phlegmaria*; Phoi-*Pholidota imbricata*; *Pholidota* spp.; Phye-*Phymatopteris ebinipes*; Phym-*Phymatopteris malacodon*; Phyo-*Phymatopteris oxyloba*; Pill-*Pilea lineolatum*; Pleh-*Pleione hookeriana*; Pleh-*Pleione humilis*; Pola-*Polypodiastrum argutum*; Pola-*Polypodioides amoena*; Poll-*Polypodioides lachnopus*; Pyrf-*Pyrrosia flocculosa*; Pyrl-*Pyrrosia lanceolata*; Pyrs-*Pyrrosia stigmosa*; Remh-*Remusatia hookeriana*; Rhop-*Rhododendrom pendulum*; Rhod-*Rhododendron dalhousiae*; Ross-*Roscoea spicata*; *Selaginella* spp.; Smio-*Smilacina oleracea*; Utrm-*Utricularia multicaulis*; Vacn-*Vaccinium nummularia*; Vacr-*Vaccinium retusum*; Vacv-*Vaccinium vacciniaceum*; Vanu-*Vandopsis undulata*; Vite-*Vittaria elongata*; Vitf-*Vittaria flexuosa*; Vith-*Vittaria himalayensis*; Vits-*Vittaria sikkimensis*; Wigs-*Wightia speciosissima*.

The relationship between microenvironmental variables and epiphyte density as shown by stepwise forward multiple regression analysis indicated that elevation ($P < 0.010$) is significant in influencing the overall distribution of epiphyte species along the forest types. In addition, light in the Lower montane, elevation in the Montane and RH in the Upper montane forests were important environmental factors (Table 4.18).

Table 4.18. Results of forward stepwise multiple regression analysis of environmental variables with epiphyte density in the three forest types in KBR.

Environmental variables	Coefficient	Standard Coefficient	Standard Error	T	Probability>t	Constant
Lower montane						
Light	0.415	0.594	0.132	3.134	0.006	1.055
Montane						
Elevation	-4.604	-0.612	1.403	-3.281	0.004	17.076
Upper montane						
Relative humidity	-4.639	-0.541	1.699	-2.731	0.014	10.09

Lianas

Forty-three liana species belonging to 37 genera and 28 families were recorded from the three forest types. The number of species was highest in the Lower montane forests (33) followed by the Montane (19) and Upper montane (15) forests. Liana species richness decreased with increasing elevation (R = -0.57; $P < 0.001$). The species diversity indices also decreased with increasing elevation (H = 3.3, 2.6, 2.4 and J = 0.95, 0.90 and 0.92 in the Lower montane, Montane and Upper montane forests. The dominance index (D) also followed the same trend (D = 35.1, 13.4, 13.5). Fisher's α diversity was greatest in the Lower montane forests, followed by Upper montane and Montane forests (Table 4.19). β-diversity was highest between the Lower and Upper montane (0.71), followed by Montane and Upper montane (0.47) forests. Lower montane and Montane forests had the lowest β-diversity value of 0.57.

Table 4.19. Liana species diversity indices in Lower montane (LM), Montane (M), and Upper montane (UM) forests of KBR. The figures in parentheses are Jackknife standard error.

Diversity indices	**Forest types**		
	Lower montane	Montane	Upper montane
Number of species	33	19	15
Shannon diversity (H)	3.34(±0.08)	2.64(±0.18)	2.44(±0.36)
Simpsons dominance (D)	35.08(±4.08)	13.41(±3.28)	13.54(±5.18)
Pielou's eveness (J)	0.95(±0.02)	0.90(±0.06)	0.92(±0.04)
Fisher's Alpha (α)	20.27(±4.64)	8.66(±3.24)	9.18(±4.83)

Vitaceae was the dominant family in the Lower montane (11.7%) and Montane (10%) forests. Caprifoliaceae, Schisandraceae and Ranunculaceae, each with 13.3% of the total species, dominated the liana community in Upper montane forests.

The three forest types differed significantly in liana species composition (Clark's *R* statistic = 0.637, $P < 0.001$). Species dissimilarity between Lower montane and Montane, Lower and Upper montane, and Montane and Upper montane forests was 61, 99.2 and 99%, respectively. *Clematis buchananiana*, *Embelia floribunda*, *Holboellia latifolia*, *Hydrangea anomala*, *Lonicera glabrata*, *Rubus paniculatus* and *Tetrastigma serrulatum* were found in all the three forest types. *Dicentra scandens*, *Gnetum montanum*, *Hedera nepalensis*, *Micrechites elliptica*, *Parthenocissus himalayana* and *Piper mullesua*, were confined to lower Montane and Montane forests. *Actinidia callosa*, *Schisandra grandiflora* and *Zanthoxylum oxyphyllum* were found only in Montane and Upper montane forests (Table 4.20).

The density of lianas decreased from 83 stems ha^{-1} in the lower montane forests to 73 stems ha^{-1} in the montane and 38 stems ha^{-1} in the upper montane forests ($F = 70.18$, $P < 0.001$). The basal area of lianas also followed a similar trend, i.e. 3.54, 2.25 and 0.13 m^2 ha^{-1} in the Lower montane, Montane and Upper montane forests, respectively.

Table 4.20. List of liana species with density and IVI in three forest types in KBR.

	Lower montane		Montane		Upper montane	
Liana species	density ha^{-1}	IVI	density ha^{-1}	IVI	density ha^{-1}	IVI
Actinidia callosa Lindl.	-	-	6	25	5	39
Aristolochia griffithii Ducharte	1	4	-	-	-	-
Celastrus stylosus Wall.	3	8	-	-	-	-
Cissus repens Lamk.	5	14	-	-	-	-
Clematis acuminata DC.	4	11	-	-	-	-
Clematis buchananiana DC.	1	3	3	9	2	17
Clematis montana DC.	-	-	-	-	4	22
Combretum flagrocarpum Herb.	3	8	-	-	-	-
Dicentra scandens G. Don	2	7	4	15	-	-
Embelia floribunda Wall.	3	8	1	5	1	11
Entada phaseoloides Merr.	1	4	-	-	-	-
Gnetum montanum Markgr.	1	16	1	13	1	12
Hedera nepalensis Koch	3	31	1	10	-	-
Holboellia latifolia Wall.	1	6	13	33	4	22
Hydrangea anomala D. Don	5	10	1	10	1	19
Ipomoea purpurea Roth	4	17	-	-	-	-
Lonicera acuminata Wall.	-	-	-	-	1	14
Lonicera glabrata Wall.	1	3	6	17	2	13
Marsdenia tenacissima Moon	2	16	2	19	-	-
Micrechites elliptica Hk. f.	5	14	4	20	-	-
Mucuna macrocarpa Wall.	1	5	-	-	-	-
Parthenocissus himalayana Planch.	6	17	4	18	-	-
Periploca calophylla Wight	1	3	-	-	-	-
Pericampylus glaucus Moon	3	14	-	-	-	-
Piper mullesua D. Don	4	9	1	4	-	-
Piper peepuloides Roxb.	3	8	-	-	-	-
Rhapidophora decursiva Schott	2	5	-	-	-	-
Ribes takare D. Don	-	-	-	-	3	22
Rubus paniculatus Smith	3	7	7	21	1	11
Sabia campanulata Wall.	-	-	-	-	2	13
Sabia paniculata Edgew.	1	3	-	-	-	-
Schisandra grandiflora Thoms.	-	-	6	20	4	34
Schisandra neglecta Smith	-	-	1	32	-	-
Smilax orthoptera DC.	-	-	4	8	-	-
Solanum jasminoides Paxton.	1	7	-	-	-	-
Stephania glabra Miers	4	10	-	-	-	-
Tetrastigma rumicispermum Planch.	2	6	-	-	-	-
Tetrastigma serrulatum Planch.	2	8	1	4	-	-
Thunbergia coccinea D. Don	2	5	-	-	-	-
Thunbergia fragrans Roxb.	-	-	-	-	6	45
Toddalia asiatica Lamk.	2	7	-	-	-	-
Trachelospermum axillare Hk. f.	1	3	-	-	-	-
Zanthoxylum oxyphyllum Edgew.	-	-	3	16	1	8

With an increase in elevation, the liana species-abundance curves exhibited higher dominance (Figure 4.14). Four dominant and co-dominant liana species, *Cissus repens*,

Clematis acuminata, *Hydrangea anomala* and *Parthenocissus himalayana* together shared 23% of the total IVI values in the Lower montane forests while the corresponding figure for Montane forests was much greater at 42% which was shared by *Actinidia callosa*, *Holboellia latifolia*, *Rubus paniculatus*, and *Schisandra grandiflora*. It further increased to 57% in Upper montane forests which were shared by *A. callosa*, *Clematis Montana*, *H. latifolia*, *Schisandra neglecta*, and *S. grandiflora* (Table 4.20).

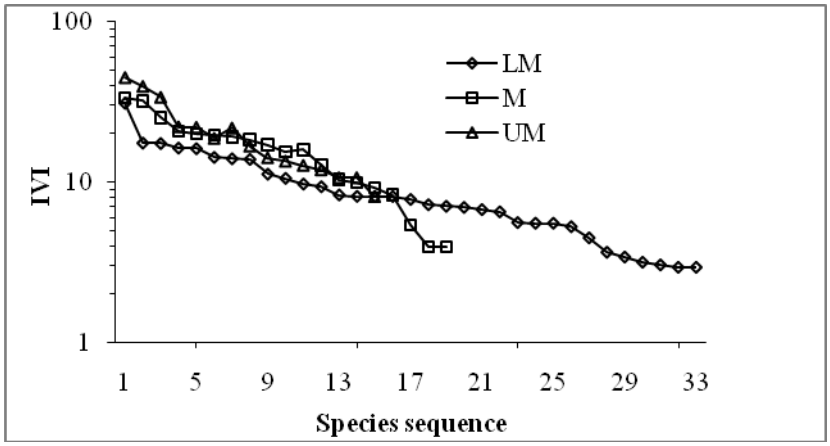


Figure 4.14. Liana dominance diversity curves in Lower montane (LM), Montane (M) and Upper montane (UM) forests in KBR.

Microenvironmental factors related to liana density

The species-environment relationship across the forests was poorly explained as the first two canonical axes accounted for 7.6 % and 6.8 % of the total variance. Nevertheless, Monte Carlo randomisation test with 100 iterations has yielded a probability of 0.009 for both the axes indicating that the axes have explained a significant part of the variability in the species abundance data (Table 4.21).

Light, soil pH, N, P and variable 'elevation' were strongly correlated with the first CCA axis and therefore were important determinants of liana species distribution across the forest types (Figure 4.15). CCA produced an ordination of all 43 species that showed the inferred ranking of the species along the environmental variables. The ordination plot shows the relative position of the species along the line of environmental vectors depicting species environmental preferences. In the Lower montane forests, *Combretum*

flagrocarpum, *Hedera nepalensis*, and *Holboellia latifolia* with high first axis species scores dominated the areas with high soil pH. Conversely, *C. buchananiana*, *Entada phaseoloides*, and *Sabia paniculata* occupied low soil pH areas. In the Montane forests, *A. callosa*, *C. buchananiana*, *L. glabrata*, *S. grandiflora* and *Z. oxyphyllum* with high first axis species scores were associated strongly with high soil N level, while *H. nepalensis*, *Hydrangea anomala*, and *Marsdenia tenacissima* were confined to low soil N areas. In the Upper montane forests, *Actinidia callosa*, *H. latifolia*, *Sabia campanulata* and *Thunbergia fragrans* were dominant in high soil P environment, while *H. anomala*, *L. acuminata* and *Z. oxyphyllum* were abundant in low soil P areas (Figure 4.15).

The relationship between microenvironmental variables and adult liana density as shown by stepwise forward multiple regression analysis, indicated that light in the Lower montane, soil P concentration in the Montane, and both light and soil P in the Upper montane forests were important determinants of liana abundance (Table 4.22).

Table 4.21. Variance explained in the Canonical Correspondence Analysis (CCA) by the first two axes across the forest types in KBR.

Axis		**1**	**2**
Total variance in species data	13.07		
Sum of canonical eigen values	3.85		
Sum of non canonical eigen values	9.21		
Canonical eigen value		0.99	0.89
% variance explained		7.57	6.83
Cumulative % variance		7.57	14.41
Probability (Monte Carlo test)		0.009	0.009
Non-canonical eigen value		0.80	0.77
% variance explained		6.15	5.89
Cumulative % variance		6.15	12.05

Table 4.22. Results of forward stepwise multiple regression analysis of environmental variables with liana density in the three forest types in KBR.

Environmental variables	Coefficient	Standard Coefficient	Standard Error	t	Probability>t	Constant
Lower montane						
Light	0.841	0.955	0.092	9.125	0.000	-0.030
Montane						
Soil phosphorus	0.880	0.836	0.204	4.312	0.003	-0.235
Upper montane						
Light	0.676	0.548	0.214	3.160	0.016	-0.762
Soil phosphorus	0.576	0.554	0.180	3.194	0.015	

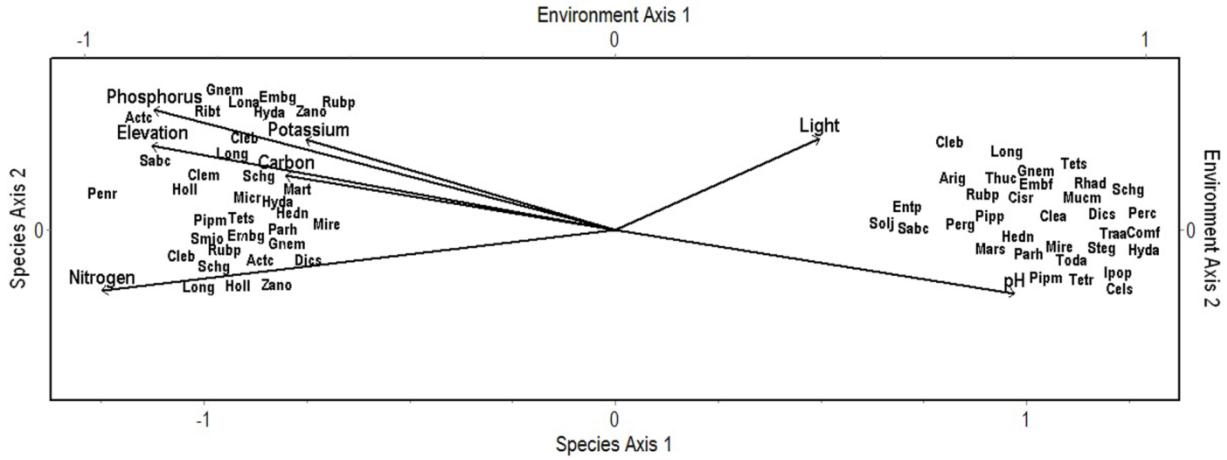


Figure 4.15. CCA ordination diagram using abundance data of 43 liana species and microenvironmental variables from 30 plots across three forest types in KBR. The environmental variables are indicated by arrow and length of the arrow indicates the strength of the correlation. For clarity, species codes have been used which consist of the first three letter of the genus and the first letter of the species name; Actc- *Actinidia callosa*; Arig- *Aristolochia griffithii*; Cels- *Celastrus stylosus*; Cisr- *Cissus repens*; Clea- *Clematis acuminata*; Cleb- *Clematis buchananiana*; Clem- *Clematis montana*; Comf- *Combretum flagrocarpum*; Dics- *Dicentra scandens*; Embf- *Embelia floribunda*; Entp- *Entada phaseoloides*; Gnem- *Gnetum montanum*; Hedn- *Hedera nepalensis*; Holl- *Holboellia latifolia*; Hyda- *Hydrangea anomala*; Ipop- *Ipomoea purpurea*; Lona- *Lonicera acuminata*; Long- *L. glabrata*; Mart- *Marsdenia tenacissima*; Mice- *Micrechites elliptica*; Mucm- *Mucuna macrocarpa*; Parh- *Parthenocissus himalayana*; Perc- *Periploca calophylla*; Perg- *Pericampylus glaucus*; Pipm- *Piper mullesua*; Pipp- *P. peepuloides*; Rhad- *Rhapidophora decursiva*; Ribt- *Ribes takare*; Rubp- *Rubus paniculatus*; Sabc- *Sabia campanulata*; Sabp- *S. paniculata*; Schg- *Schisandra grandiflora*; Schn- *S. neglecta*; Smio- *Smilax orthoptera*; Solj- *Solanum jasminoides*; Steg- *Stephania glabra*; Tetr- *Tetrastigma rumicispermum*; Tets- *T. serrulatum*; Thuc- *Thunbergia coccinea*; Thuf- *T. fragrans*; Toda- *Toddalia asiatica*; Traa- *Trachelospermum axillare*; Zano- *Zanthoxylum oxyphyllum*.

Discussion

The species diversity and richness pattern of different vegetation components in three forests were largely influenced by elevation. Lower montane forests had higher diversity in terms of family, genera and species in comparison to Montane and Upper montane forests. The three forest types were endowed with a number of threatened plant species (Table 4.23 and Plate 4.1). Considerable differences in floristic composition among the plant communities in different forest types indicate the important role of prevailing environmental conditions in determining species composition. A decreasing trend in species richness with elevation has been reported by several earlier workers (Yoda 1967; Odland & Birks 1999; Grytnes et al. 2002). It is obvious that the forests in KBR are

strongly correlated with elevation; which is found to be in accordance with the general elevation pattern of ecosystem formation (Richerson et al. 1980). Elevation gradient produce diverse climates, along with resultant soil differentiation, promoting the diversification of plant species (Brown 2001; Lomolino 2001).

Table 4.23. List of Rare and Threatened plants in three forests.

Plant species	Forest types	IUCN category
Acer hookeri Miq.	Lower montane	Rare
Aconitum ferox Wall.	Upper montane	Endangered
Arisaema griffithii Griffith.	Montane	Vulnerable
Aristolochia griffithii Ducharte	Lower montane/Montane	Vulnerable
Campylandra aurantiaca Baker	Lower montane/Montane	Endangered
Cardiocrinum giganteum (Wall.) Makino	Lower montane	Rare
Ceropegia hookeri Clarke	Montane	Critically rare
Nardostachys grandiflora DC.	Upper montane	Critically rare
Panax pseudo-ginseng Wall.	Upper montane	Lower risk
Rheum nobile Hook. f.	Upper montane	Endangered
Rhododendron anthopogon D. Don	Upper montane	Vulnerable
Rhododendron maddenii Hook.f.	Montane	Rare

Whittaker (1972, 1977), MacArthur (1965), Wilson & Shmida (1984) and Brokaw & Scheiner (1989) have discussed the importance of α- and β-diversity in explaining the species richness of a plant community. Higher α- diversity of trees, herbs and lianas in the Lower montane forests, indicates the existence of a wide range of vegetation formation such as lowland forest, transitional forest, and riverine forests besides the upland mountain forests. These diverse formations have contributed to high species richness in this forest. In comparison to other Lower montane/subtropical forests, the greater diversity of different lifeforms in the Lower montane forests can be attributed to the prevailing monsoon effects in the region, which remains one of the major factors for high vegetation diversity in the main Himalayan region (Singh & Singh 1987). Being at the meeting point of Indo-Malayan and Indo-Chinese biogeographical realms as well as Himalayan and peninsular India, it contains the floristic elements from all the biogeographical zones.

Plate 4.1. Some threatened flowering plants from three forest types in KBR: (1) Left to right - Species from Upper montane forests-*Rheum nobile*, *Aconitum* spp., *Megacodon stylophorus*; (2) species from Upper montane - *Juncus grisebachii*, *Saussurea gossypiphora*, *S. nepalensis*; (3) Species from Montane - *Rhododendron hodgsonii*, *Aconitum elwesii*, *Rhododendron dalhousiae*; (4) Species from Lower montane - *Cardiocrinum giganteum*, *Acer hookeri*, *Aristolochia griffithii*.

The variation in α and β-diversity for epiphytes in three forest types may be due to spatial microhabitats, following a gradient from moist part of the studied forest (LM) to the drier part (UM) and suggests that the distance to moisture sources plays a crucial role in determining richness and composition of epiphyte communities. It has also been argued that epiphyte richness is associated with moisture of the slopes where they grow (Sanford 1968; Sudgen & Robins 1979). β-diversity measures the extent of species replacement or biotic change along environmental gradients (Whittaker 1972; Brokaw & Scheiner 1989). It also reflects the extent of similarity and habitat diversity among the forest types. In the present study, β-diversity for trees and lianas was lower than the shrub and herb components of the vegetation. The β-diversity values for all the components were however less than that obtained for BCI forest studied by Brokaw & Scheiner (1989). Homogeneity in vegetation structure, less diverse microhabitats (Barik et al. 1992) and availability of less treefall gaps could be the reasons of lower β-diversity in the Eastern Himalayan forests.

Similarity test for species composition between the three forest types showed that the forests were significantly dissimilar. It also revealed that the floristic composition had greater similarity among the adjacent forests and had greater dissimilarity among the forests located farther apart. Such differences in species composition may be attributed primarily to the elevational variations (Grell et al. 2005).

As expected, the tree basal area of the Lower montane forests was higher than that of Montane and Upper montane forests which could be attributed to more number of individuals in the higher girth classes. The overall tree density-diameter distribution pattern indicates a rather stable tree population structure in all the three forests (Rao et al. 1990).

The higher value of dominance index for the tree species in Lower montane forests could be due to increased stress on account of the low disturbance level and

extraction of a few trees from the buffer zone of KBR. The low dominance-abundance curve for lianas as obtained in the Lower montane forests indicated more equitable resource distribution pattern among the constituent species than those in the Montane and the Upper montane forests (Crawley 1997). Such equitable resource distribution pattern might have made the Lower montane forests more species rich in comparison with Montane and Upper montane forests.

The dominance of the polypodiaceae is a trend in most humid montane epiphyte communities. Dominance-diversity curves also showed that most IVI was shared by a few species in the Upper montane forests while in the Montane and the Lower montane forests, IVI was equitably distributed. The greater dominance pattern in Upper montane forests indicates that the community is non-equilibrium (Hubbell 1979). Ordination of forest types on the basis of abundance data with respect to epiphyte species compositions resulted into slight overlapping of the Montane forests species with the Lower montane and Montane forests species. The similarities between orchids and fern species are mainly responsible for this overlapping. The long repent lifeform of *Bulbophyllum* spp., and pendant lifeform of *Hoya* spp., allow plant stems to search for light whilst retaining an original attachment, an effective local colonisation mechanism. The abundance of this lifeform was also reported by Freiberg (1996) in montane forest of Ecuador.

The selected host trees for epiphytes ranged from $\geq$ 35 to < 90 cm diameter. The occurrence of large number of species on bigger diameter classes can be explained by the larger area offered with a great variety of host architecture with different microhabitats (Annaselvam & Parthasarathy 2001). The significant relationship between epiphyte species and trunk girth class conforms to the findings of Catling and Lefkovitch (1989) in a Gautemalan forest. The thick humus rich branches and stems are densely covered with epiphytes, accumulating substantial amount of humus, nutrients and moisture. These are appropriate requirements as reported in cloud forest of Veracruz, Mexico (Heitz & Hietz-

Seifert 1995a) and in West African rain forest (Johansson 1974, 1975). The larger epiphytes in KBR such as *Vaccinium nummularia, Rhododendron dalhousiae, R. Pendulum* in the Montane and Upper montane forests and *Macropanax undulatum, Pentapanax leschenaultii, V. vacciniaceum, Wightia speciosissima*, and *Drynaria propinqua* in Lower montane forests therefore, inhabit bigger trees such as, *Lithocarpus pachyphylla, Schima wallichii* and *Elaeocarpus lanceaefolius*.

The number and density of epiphytic species increased with increase in height class of the host tree. Tree tops, branches and twigs together represented a wider range of the microclimatic gradient, from the more shady environments near the trunk to the outermost parts of the tree, where exposure to light and desiccating wind were common. Tree base was poor in epiphytes, only some Araceae inhabited these strata. The richest trunk vegetation was found on the mossy substratum. This was common in the Upper montane and Montane forests on *Abies densa* and *Quercus lamellosa* host respectively. The occurrence of more epiphytes in upper canopy layer than any other part of the tree may be attributed to the bryophyte mats present in larger surface area. The epiphytes also occupied the forks of tree trunk which accumulate litter and humus and provide mechanical support. The high canopy dwelling species are *Eria coronaria*, *Dendrobium* spp., *Lepisorus* spp., *Phalaenopsis tainitis, Vittaria* spp., and *Vaccinium* spp. Most of them are orchids as they are able to withstand drought (Benzing 1976). Long and fine roots of Orchidaceae also seem adaptive to this special habitat.

Seasonal variation in air temperature, soil temperature, moisture content, C and N concentrations as observed in the present study corroborates the findings of Barik et al. (1992) in a subtropical broad-leaved forest of north-east India. However, studies depicting variation in microenvironmental factors among different forest types are rare. In addition to elevation, light, soil pH, C, N and P were correlated with abundance data. Differences in soil properties, elevation, topography and other environmental conditions

in different forest types could explain substantially the observed differences in plant species diversity and abundance in the three forests. An observed gradient in many environmental variables was also related to the differences in structural and functional characteristics of the forest types studied along an elevation gradient in Tierra del Fuego by Frangi et al. (2005).

Relatively lower eigen values of the first two constrained CCA axes and greater eigen values of the first residual (non canonical) axis as obtained in the present study apparently indicate that the environmental variables are not sufficient to predict the main variations on species abundance extracted by CCA, but they do predict a substantial part of remaining variations. Ter Braak and Prentice (1988) opined that terrestrial community data commonly give a residual eigen value as large as the first constrained eigen value, however carefully the environmental variables are chosen. Some of the plant species were confined to specific forest types while others occurred across the whole range.

The strong clustering of Lower montane trees, shrubs, herbs and lianas along the soil pH and light gradients in the CCA ordination plot supported the earlier observations on plant preference for less acidic soil and light (Lowe & Walker 1977; Putz 1984; Whitmore 1989; Phillips & Gentry 1994). The important role of light in determining the density and distribution of many liana species such as *Cissus repens Clematis acuminata* and *Parthenocissus himalayana* is in conformity with the findings of Castellanos (1991), who concluded that liana species thrives well in areas of abundant light in the forest. Study by Laurance et al. (2001) on the effect of forest fragmentation and treefall gaps on liana communities concluded that liana abundance increased considerably near forest edges. However, in the Montane and Upper montane forests, the abundance of trees, shrubs, herbs and lianas was mainly driven by the edaphic variables (N, P and K). As shown by stepwise forward multiple regression analysis, light and soil P either alone or both influenced liana density in different forests. Soil pH in Lower montane and C in

Montane and Upper montane forests influenced tree density. Liana and tree desity in the Montane and Upper montane forests were strongly related to soil nutrients such as N and P, and pH, C, respectively. Light and C either alone or both influenced shrub density in Montane and Upper montane forests. pH, elevation in the Montane, N and P either alone or both influenced herb density in Montane and Upper montane forests respectively. The role of soil nutrients in plant species distribution was emphasised by Dewalt et al. (2000, 2006) and Godefroid et al. (2007) corroborating to the present finding. All these abiotic factors influence the adaptation and survival of the plant species contributing to greater species diversity in the forest communities (Ibarra-Manríquez & Martínez-Ramos 2004; van der Heijden & Phillips 2008).

Epiphytic species had strong clustering around RH, elevations and air temperature. Result of stepwise forward multiple regression analysis also showed the influence of light, elevation and relative humidity on epiphyte density. The roles of elevational gradient and RH in epiphyte richness were also emphasised by Kufer et al. (2004), and Kharkwal et al. (2005) corroborating to the present finding. Upto 107 epiphyte species were reported by Valdivia (1977) on a single tree where dense evergreen canopies feature multiple microclimates and numerous alternatives to earth soil. He attributed such high species richness to relative humidity. But many epiphytes require high exposure and others like certain filmy ferns cannot endure either as much light or the associated aridity (Hietz & Briones 1998). Consequently, epiphytes segregated along the environmental gradients in different forest types (Annexure 2).

To conclude, it is evident that there are considerable differences in species composition among different forests types. This was attributed to diversity in hahitat types, forest types, host structure, and available environmental and forest structural gradients.

Annexure 1. Density (ind ha^{-1}), basal area (m^2 ha^{-1}) and IVI of trees, shrubs and herbs in three forest types.

Name of species		Lower montane			Montane			Upper montane		
	Family	Den.	IVI	BA	Den.	IVI	BA	Den.	IVI	BA
Trees										
Abies densa Griffith.	Pinaceae	-	-	-	-	-	-	37.50	67.41	0.28
Acer campbellii Hk. f. & Thoms ex Hiern.	Aceraceae	10.00	6.39	0.06	-	-	-	-	-	-
Acer spp.	Aceraceae	-	-	-	6.00	7.24	0.09	-	-	-
Acer thomsonii Miq.	Aceraceae	-	-	-	3.50	4.11	0.09	-	-	-
Alangium alpinum (Clarke) Sm. & Cave	Alangiaceae	-	-	-	5.00	6.95	0.11	-	-	-
Alangium begoniaefolium Baill.	Alangiaceae	-	-	-	6.00	6.18	0.04	-	-	-
Albizia chinensis (Osbeck.F) Merr.	Fabaceae	9.00	5.34	0.06	-	-	-	-	-	-
Alnus nepalensis D. Don	Betulaceae	29.00	23.86	0.19	5.50	4.51	0.04	-	-	-
Betula alnoides Buch. - Ham. ex D. Don	Betulaceae	-	-	-	10.50	14.66	0.13	-	-	-
Buddleja colvilei Thom.	Buddlejaceae	-	-	-	-	-	-	11.50	12.02	0.05
Castanopsis hystrix A. DC.	Fagaceae	14.00	14.48	0.28	-	-	-	-	-	-
Castanopsis indica (Roxb ex Lindl.) A.DC.	Fagaceae	6.00	5.59	0.22	-	-	-	-	-	-
Castanopsis tribuloides (Sm.) A. DC.	Fagaceae	10.00	10.95	0.31	-	-	-	-	-	-
Cinnamomum bejolghota (Buch. – Ham.) Sweet	Lauraceae	8.00	4.90	0.06	-	-	-	-	-	-
Cinnamomum impressinervium Meisn.	Lauraceae	4.50	2.75	0.04	9.50	14.25	0.17	-	-	-
Daphne bholua Buch. – Ham. ex D. Don	Thymelaceae	-	-	-	7.50	7.37	0.05	-	-	-
Daphne papyracea Wall. ex Steud	Thymelaceae	7.00	3.40	0.03	-	-	-	-	-	-
Dendrocalamus hamiltonii Nees et Arn. ex Munro	Poaceae	11.00	5.43	0.04	-	-	-	-	-	-
Elaeocarpus lanceaefolius Roxb.	Elaeocarpaceae	24.00	19.61	0.18	6.50	6.25	0.03	-	-	-
Engelhardtia colebrokiana Lindl.	Juglandaceae	8.00	4.66	0.09	-	-	-	-	-	-
Engelhardtia spicata Lesch ex Bl.	Juglandaceae	8.00	6.47	0.15	-	-	-	-	-	-
Euodia fraxinifolia Hk. f.	Theaceae	20.00	10.99	0.06	10.50	14.33	0.13	-	-	-
Eurya cerasifolia (D. Don) Kobuski	Theaceae	16.00	8.59	0.06	-	-	-	-	-	-

Eurya japonica Thunb.	Theaceae	10.50	6.03	0.09	4.00	6.11	0.07	-	-	-
Exbucklandia populnea (R. Br ex Griff.) R. Br.	Hamamelidaceae	9.00	8.03	0.22	-	-	-	-	-	-
Ficus auriculata Lour.	Moraceae	8.50	4.34	0.04	-	-	-	-	-	-
Ficus neriifolia Sm.	Moraceae	6.00	2.80	0.03	-	-	-	-	-	-
Ficus semicordata Buch. - Ham ex Sm.	Moraceae	6.50	3.45	0.04	-	-	-	-	-	-
Glochidion acuminatum Mull.	Euphorbiaceae	6.50	3.27	0.06	-	-	-	-	-	-
Hydrangea aspera Buch. - Ham. ex D. Don	Hydrangeaceae	-	-	-	7.50	7.87	0.05	-	-	-
Ilex dipyrena	Aquifoliaceae	-	-	-	-	-	-	7.00	11.22	0.09
Ilex fragilis Hk. f.	Aquifoliaceae	-	-	-	7.50	6.85	0.04	-	-	-
Indigofera dosua Buch. - Ham. ex D. Don.	Fabaceae	3.00	1.53	0.03	-	-	-	-	-	-
Juglans regia L.	Juglandaceae	17.00	13.34	0.17	-	-	-	-	-	-
Leucosceptrum canum Sm.	Lamiaceae	7.00	4.00	0.07	8.50	9.82	0.09	-	-	-
Lithocarpus elegans (Bl.) Hatus ex Soepadmo	Fagaceae	6.00	5.85	0.27	-	-	-	-	-	-
Lithocarpus pachyphylla (Kurz) Rehder	Fagaceae	-	-	-	6.50	6.72	0.08	-	-	-
Litsea cubeba Bl.	Lauraceae	9.00	5.43	0.07	-	-	-	-	-	-
Litsea elongata (Wall. ex Nees.) Benth. et Hk. f.	Lauraceae	-	-	-	15.50	23.70	0.18	-	-	-
Lyonia ovalifolia (Wall.) Drude	Ericaceae	11.50	5.43	0.05	11.00	12.29	0.07	-	-	-
Macaranga denticulata (Bl.) Muell. – Arg.	Euphorbiaceae	12.00	6.32	0.06	-	-	-	-	-	-
Macaranga indica Wight	Euphorbiaceae	6.50	3.27	0.06	-	-	-	-	-	-
Macaranga pustulata King ex Hk. f.	Euphorbiaceae	14.00	7.03	0.06	-	-	-	-	-	-
Magnolia campbellii Hk. f. & Thoms.	Magnoliaceae	-	-	-	9.50	9.53	0.08	-	-	-
Myrsine semiserrata Wall.	Myrsinaceae	15.00	7.97	0.04	-	-	-	-	-	-
Pentapanax leschenaultii (DC.) Seem.	Araliaceae	4.50	2.36	0.03	-	-	-	-	-	-
Persea gammieana Kosterm. ex Kosterm. & Charter	Lauraceae	-	-	-	12.00	21.40	0.24	-	-	-
Prunus cerasoides D. Don	Rosaceae	7.00	3.76	0.05	-	-	-	-	-	-
Prunus cornuta (Royle) Steud.	Rosaceae	-	-	-	7.00	10.44	0.10	-	-	-
Prunus spp.	Rosaceae	-	-	-	-	-	-	18.00	17.14	0.08
Quercus lamellosa Sm.	Fagaceae	-	-	-	9.50	8.33	0.06	-	-	-

Quercus lineata Bl.	Fagaceae	12.00	13.84	0.32	7.50	15.78	0.24	-	-	-
Rhododendron arboreum Sm.	Ericaceae	-	-	-	8.00	15.72	0.22	-	-	-
Rhododendron campanulatum D. Don	Ericaceae	-	-	-	-	-	-	45.00	36.11	0.05
*Rhododendron cinnabarinum*Hook.f.	Ericaceae	-	-	-	-	-	-	13.00	8.45	0.04
Rhododendron falconeri Hk. f.	Ericaceae	-	-	-	17.50	18.70	0.09	17.50	13.97	0.06
Rhododendron grande Wight	Ericaceae	-	-	-	-	-	-	28.50	39.79	0.06
Rhododendron thomsonii Hk. f.	Ericaceae	-	-	-	-	-	-	20.50	17.26	0.03
Rhododendrum spp. Sm.	Ericaceae	-	-	-	-	-	-	42.50	38.00	0.09
Rhus hookeri Sahni & Bahadur	Anacardiaceae	1.50	1.35	0.09	-	-	-	-	-	-
Rhus javanica Hk. f.	Anacardiaceae	13.00	7.46	0.04	-	-	-	-	-	-
Ricinus communis L.	Euphorbiaceae	13.00	6.34	0.02	-	-	-	-	-	-
Saurauia napaulensis DC.	Saurauiaceae	3.00	1.66	0.05	-	-	-	-	-	-
Schefflera impressa (Clarke) Harms	Araliaceae	6.50	3.15	0.02	-	-	-	-	-	-
Symplocos glomerata King ex Gamble	Symplocaceae	5.50	3.72	0.03	-	-	-	-	-	-
Symplocos ramosissima Wall. ex D. Don	Symplocaceae	-	-	-	13.00	15.00	0.09	-	-	-
Symplocos theifolia D. Don	Symplocaceae	-	-	-	6.00	6.73	0.07	-	-	-
Talauma hodgsonii Hk. f. & Thoms.	Magnoliaceae	3.00	2.64	0.14	-	-	-	-	-	-
Toona ciliata M. Roem.	Anacardiaceae	4.50	3.86	0.15	-	-	-	-	-	-
Toricellia tillifolia DC.	Toricelliaceae	13.50	6.24	0.04	-	-	-	-	-	-
Trevesia palmata (Roxb.) Vis.	Araliaceae	3.00	1.59	0.03	-	-	-	-	-	-
Tsuga dumosa (D. Don) Eichler	Pinaceae	-	-	-	-	-	-	24.00	38.64	0.27
Viburnum cylindricum Buch. - Ham. ex D. Don	Caprifoliaceae	14.00	6.65	0.03	-	-	-	-	-	-
Viburnum mullaha Buch. - Ham. ex D. Don	Caprifoliaceae	6.00	3.08	0.03	-	-	-	-	-	-
Viburnum nervosum D. Don	Caprifoliacea	7.00	3.43	0.03	6.50	8.98	0.06	-	-	-
Wendlandia paniculata DC.	Rubiaceae	6.00	3.51	0.06	-	-	-	-	-	-
*Zanthoxylum oxyphyllum*Edgew	Rutaceae	7.50	3.90	0.03	11.50	10.16	0.03	-	-	-
		Lower montane		**Montane**		**Upper montane**				
Shrub	**Family**	**Den.**	**IVI**	**Den.**	**IVI**	**Den.**	**IVI**			

Aconogonum molle (D. Don) Hara	Polygonaceae	8.00	4.56	-	-	-	-
Arundinaria maling Gamble	Poaceae	-	-	21.50	29.182	-	-
Bambusa nutans Wall. ex Munro	Poaceae	20.00	8.32	-	-	-	-
Berberis sikkimensis Ahrendt	Berberidaceae	-	-	25.00	37.91	-	-
Berberis spp.	Berberidaceae	-	-	-	-	40	37.094
Boehmeria macrophylla D. Don	Urticaceae	7.50	4.41	-	-	-	-
Boehmeria platyphylla D. Don	Urticaceae	17.00	9.44	-	-	-	-
Clerodendron colebrookianum Walp.	Verbenaceae	7.00	6.30	-	-	-	-
Daphne bholua Buch. - Ham. ex D. Don	Thymelaceae	7.00	4.93	-	-	-	-
Debregeasia longifolia (Burm. f.) Wedd.	Urticaceae	10.00	6.56	-	-	-	-
Desmodium confertum DC.	Fabaceae	5.00	4.31	-	-	-	-
Deutzia compacta Craib	Hydrangeaceae	-	-	14.00	29.651	-	-
Dichroa febrifuga Lour.	Hydrangeaceae	6.50	4.09	-	-	-	-
Dicranopteris linearis (Burm. f.) Underw.	Gleicheniaceae	18.50	7.17	-	-	-	-
Dobinea vulgaris Buch. - Ham. ex D. Don	Rubiaceae	12.50	8.03	-	-	-	-
Edgeworthia gardneri (Wall.) Meisn.	Thymelaceae	7.00	4.93	8.50	21.574	-	-
Elsholtzia flava (Benth.) Benth.	Lamiaceae	30.00	13.51	-	-	-	-
Girardina diversifolia (Link) Friis	Urticaceae	9.50	5.03	-	-	-	-
Gleichenia glauca (Thunb.) Hook.	Gleicheniaceae	11.00	4.82	-	-	-	-
Juniperus recurva Buch. – Ham. ex D. Don	Cupressaceae	-	-	-	-	29.50	29.75
Luculia gratissima (Wall.) Meisn.	Rubiaceae	18.00	12.49	-	-	-	-
Maesa chisia Buch. - Ham ex D.Don	Myrsinaceae	9.00	4.19	-	-	-	-
Maesa ramentacea Wall. ex Roxb..	Myrsinaceae	8.00	5.93	-	-	-	-
Melastoma malabathricum L.	Melastomatacae	9.00	6.93	-	-	-	-
Melastoma normale D. Don	Melastomataceae	13.00	9.55	-	-	-	-
Mussaenda treutleri Stapf.	Rubiaceae	10.00	5.87	-	-	-	-
Neillia rubiflora D. Don	Rosaceae	7.00	6.30	-	-	-	-
Osbeckia sikkimensis Craib.	Melastomataceae	2.00	2.00	-	-	-	-

Oxyspora paniculata (D. Don) DC.	Melastomataceae	12.00	9.24	-	-	-	-
Pavetta indica L.	Rosaceae	5.00	3.62	-	-	-	-
Photinia integrifolia Lindl.	Rosaceae	6.00	5.99	-	-	-	-
Rhododendron anthopogon D. Don	Ericaceae	-	-	-	-	59.50	42.57
Rhododendron lepidotum Wall. ex G. Don	Ericaceae	-	-	-	-	55.00	37.79
Rhododendron setosum D. Don	Ericaceae	-	-	-	-	28.50	26.47
Rosa sericea Lindl.	Rosaceae	-	-	22.50	51.22	21.50	26.33
Rubus ellipticus Sm.	Rosaceae	8.00	5.25	-	-	-	-
Rubus mollucanus L.	Rosaceae	9.00	6.93	-	-	-	-
Sambucus adnata Wall. ex DC.	Sambucacaeae	10.00	8.61	-	-	-	-
Thysanolaena maxima (Roxb.) Kuntz.	Poaceae	16.50	10.65	-	-	-	-
Zanthoxylum oxyphyllum Edgew.	Rutaceae	-	-	9.50	30.46	-	-
		Lower montane		**Montane**		**Upper montane**	
Herbs	**Family**	**Den.**	**IVI**	**Den.**	**IVI**	**Den.**	**IVI**
Achyranthes aspera L.	Amaranthaceae	6000	1.96	-	-	-	-
Aconitum spicatum (Burhl) Stapf.	Ranunculaceae	-	-	-	-	17000	6.78
Agrimonia pilosa Ledeb.	Rosaceae	4000	1.95	-	-	-	-
Ainsliaea aptera DC.	Asteraceae	-	-	2000	0.71	-	-
Aletris pauciflora (Klotzsch) Hand. - Mazz.	Melanthiaceae	-	-	-	-	11000	5.37
Anaphalis busua (Buch. – Ham. ex D. Don) DC.	Asteraceae	-	-	-	-	8000	3.99
Anaphalis margaritacea (L.) Benth.	Asteraceae	14000	4.56	-	-	-	-
Anaphalis triplinervis (Sims) C.B. Clarke	Asteraceae	9000	2.45	7000	2.27	34000	9.06
Anisadenia saxatilis Wall. ex Meisn.	Linaceae	-	-	10000	3.55	-	-
Anisomeles indica (L.) O. Kuntz.	Lamiaceae	10000	4.88	-	-	-	-
Anthogonium gracile Lindl.	Orchidaceae	4500	1.39	-	-	-	-
Arisaema concinnum Schott.	Araceae	-	-	-	-	12000	5.08
Arisaema griffithii Schott.	Araceae	-	-	9000	4.27	9500	3.78
Arisaema jacquemontii Bl.	Araceae	-	-	-	-	5500	2.68
Artemisia nilagirica (Clarke) Pamp.	Asteraceae	4000	1.3	-	-	-	-
Artemisia vulgaris L.	Asteraceae	4000	1.95	-	-	-	-
Arundinella bengalensis (Spreng.) Druce	Poaceae	12000	4.56	19000	6.53	-	-
Astilbe rivularis Buch. - Ham. ex D. Don	Hydrangeaceae	6000	2.28	-	-	-	-

Athyrium rubricaule (Edgew. ex C. B. Clarke) Bir	Athyriaceae	2000	0.98	6000	2.56	-	-
Begonia josephii DC.	Begoniaceae	-	-	-	-	11000	4.47
Begonia rubella Buch. - Ham. ex D. Don	Begoniaceae	4000	1.95	-	-	-	-
Bidens biternata (Lour.) Merr. & Sherff.	Asteraceae	28000	5.89	-	-	-	-
Bidens pilosa L.	Asteraceae	19000	7.65	-	-	-	-
Bistorta affinis (D. Don) Greene	Polygonaceae	-	-	-	-	11000	4.02
Bupleurum longicaule Wall. ex DC.	Apiaceae	-	-	-	-	6500	3.30
Caltha palustris L.	Ranunculaceae	-	-	-	-	5000	2.15
Campanula pallida Wall.	Campanulaceae	14000	5.53	-	-	-	-
Campylandra aurantiaca Baker	Liliaceae	-	-	17500	3.75	-	-
Carex filicina Nees	Cyperaceae	22000	6.85	-	-	-	-
Cautleya gracilis (Sm.) Dandy	Zingiberaceae	7500	2.53	-	-	-	-
Chlorophytum nepalense (Lindl.) Baker	Anthericaceae	13500	5.45	-	-	-	-
Chrysosplenium carnosum Hk. f. & Thoms.	Saxifragaceae	-	-	17000	6.25	-	-
Cirsium verutum (D. Don) Spreng.	Asteraceae	-	-	-	-	7500	3.46
Clintonia udensis Trautv. & Mey.	Liliaceae	-	-	-	-	12500	3.81
Coniogramme cautata (Wall. ex Ettingshausen) Ching	Hemionitidaceae	6000	2.93	-	-	-	-
Craniotome furcata (Link) Kuntz.	Lamiaceae	-	-	23500	3.73	-	-
Cuphea balsamona Cham. et Schlechtend	Lythraceae	7000	3.09	-	-	-	-
Cyanotis vaga (Lour.) Schult. & Schult. f.	Commelinaceae	12000	5.21	36500	8.56	-	-
Cyathula capitata Miq.	Lamiaceae	-	-	25000	5.23	-	-
Cynodon dactylon (L.) Pers.	Poaceae	21500	4.17	-	-	-	-
Cyperus niveus Retz.	Cyperaceae	-	-	4000	1.42	-	-
Cyperus rotundus L.	Cyperaceae	4000	1.42	5000	1.99	-	-
Desmodium multiflorum DC.	Fabaceae	10000	4.88	-	-	-	-
Dicrocephala integrifolia (Lf) Kuntz.	Asteraceae	11000	3.75	-	-	-	-
Dryopteris barbigera (Hk.) O. Kuntz.)	Dryopteridaceae	12000	4.56	20500	5.03	-	-
Dryopteris chrysocoma (Christ.) C. Chr.	Dryopteridaceae	8000	3.9	-	-	-	-
Dryopteris sparsa (D. Don) O. Kuntz.	Dryopteridaceae	6000	2.93	-	-	-	-
Dryopteris spp.	Dryopteridaceae	-	-	-	-	16000	7.07
Elatostema obtusum Wedd.	Urticaceae	-	-	4000	1.42	-	-
Elatostemma platyphylla Wedd.	Urticaceae	-	-	6000	2.56	-	-
Elatostemma sessile J. R. & G. Forst.	Urticaceae	-	-	17500	5.89	-	-
Elsholtzia blanda (Benth.) Benth.	Lamiaceae	24000	7.17	-	-	-	-
Elsholtzia fruticosa (D. Don) Rehder	Lamiaceae	-	-	11000	3.26	-	-

Equisetum diffusum D. Don	Equisetaceae	12000	3.59	-	-	-	-
Erigeron bellidioides Benth. ex C. B. Clarke	Asteraceae	9000	4.07	-	-	-	-
Erigeron karvinskianus DC.	Asteraceae	8000	3.25	2000	1.14	-	-
Eupatorium cannabinum L.	Asteraceae	4000	1.95	-	-	-	-
Euphorbia sikkimensis Boiss.	Euphorbiaceae	-	-	18000	5.53	-	-
Fragaria nubicola Lindl. ex Lacaita	Rosaceae	8000	2.61	49000	12.90	18000	8.75
Fritillaria cirrhosa D. Don	Liliaceae	-	-	-	-	12500	4.71
Galium asperifolium Wall.	Rubiaceae	-	-	8000	3.27	-	-
Galium elegans Wall. ex Roxb.	Rubiaceae	-	-	44500	7.11	6500	3.30
Galium mullago Hk. f.	Rubiaceae	14000	3.59	-	-	-	-
Geranium nepalense Sweet.	Geraniaceae	8000	1.96	-	-	-	-
Gnaphalium luteo - album L.	Asteraceae	10500	3.66	-	-	-	-
Gonostegia hirta (Bl. ex Hassk) Miq.	Urticaceae	12000	3.91	-	-	-	-
Gynura cusimbua (D. Don.) S. Moore	Asteraceae	-	-	2000	0.71	-	-
Hackelia uncinata (Royle ex Benth.) C. Fischer	Gentianaceae	-	-	-	-	11000	4.47
Hedychium spicatum Sm.	Zingiberaceae	2000	0.98	-	-	-	-
Hedyotis scandans D. Don	Rubiaceae	12500	4.32	-	-	-	-
Hemiphragma heterophyllum Wall.	Scrophulariaceae	-	-	2000	0.71	16000	5.72
Herminium lanceum (Thunb. ex Sw.) Vuijk	Orchidaceae	2000	0.98	-	-	-	-
Hydrocotyle nepalensis Hk.	Apiaceae	14000	4.24	-	-	-	-
Hypericum elodeoides Choisy	Hypericaceae	-	-	-	-	4500	1.62
Hypoestes triflora Roem. & Sch.	Acanthaceae	5000	2.11	-	-	-	-
Hypoxis aurea Lour.	Hypoxidaceae	3000	1.14	8000	2.84	-	-
Impatien urticifolia Wall.	Balsaminaceae	-	-	24000	4.23	-	-
Impatiens spp.	Balsaminacaeae	7000	2.44	-	-	-	-
Juncus spp.	Juncaceae	-	-	-	-	52000	10.15
Knoxia sumatrensis (Roxb.) Korth	Rubiaceae	5000	2.11	-	-	-	-
Lecanthus peduncularis (Royle) Wedd.	Urticaceae	20000	3.93	-	-	-	-
Leucostegia immersa (Wall.) Presl.	Davalliaceae	25000	5.4	-	-	-	-
Maianthemum purpureum (Wall.) La Frankie	Convallariaceae	-	-	-	-	9500	4.68
Meconopsis villosa (Hk. f.) G. Taylor	Papaveraceae	-	-	-	-	12000	5.08
Megacodon stylophorus (C. B. Clarke) Sm.	Gentianaceae	-	-	-	-	12500	3.81
Mimulus nepalensis Benth.	Scrophulariaceae	-	-	6000	2.13	-	-
Notochaete hamosa Benth.	Lamiaceae	-	-	19000	4.39	-	-
Oplismenus compositus (L.) P. Beauv.	Asteraceae	2000	0.98	-	-	-	-

Osbeckia stellata Ker. – Gawl.	Melastomataceae	6500	2.36	-	-	-	-
Oxalis acetosella L.	Oxalidaceae	-	-	19000	5.25	-	-
Oxalis corniculata L.	Oxalidaceae	-	-	6000	2.56	-	-
Panax pseudo ginseng Wall.	Araliaceae	-	-	11000	4.12	-	-
Paradavallodes multidentatum (Wall.) Ching	Davalliaceae	4000	1.3	-	-	-	-
Paris polyphylla Sm.	Trilliaceae	-	-	2000	0.71	-	-
Paspalum destichum L.	Poaceae	10000	2.94	-	-	-	-
Persicaria capitata (Buch. - Ham. ex D. Don) Gross	Polygonaceae	26000	4.91	-	-	16500	5.80
Persicaria chinense L.	Polygonacea	-	-	17500	5.89	-	-
Persicaria polystachya (Wall. ex Meisn.) Gross	Polygonaceae	-	-	-	-	16000	5.72
Persicaria runcinata (Buch. - Ham. ex D. Don) H. Gross	Polygonaceae	11000	3.42	52500	9.52	-	-
Phlomis bracteosa Royle ex Benth.	Lamiaceae	-	-	53500	8.81	-	-
Pilea scripta (Buch. - Ham. ex D. Don) Wedd.	Urticaceae	7000	3.09	3000	1.28	-	-
Pilea symmeria Wedd.	Urticaceae	-	-	-	-	13500	4.87
Pilea umbrosa Bl.	Urticaceae	-	-	4000	1.85	-	-
Plantago erosa Wall.	Plantaginaceae	-	-	10000	3.98	-	-
Poa alpina L.	Poaceae	-	-	-	-	16000	26.33
Poa annua L.	Poaceae	-	-	2000	1.14	-	-
Poa himalayana Nees ex Steud.	Poaceae	-	-	20500	7.60	20000	6.36
Pogonatherum paniceum (Lam.) Hack.	Poaceae	9000	2.77	-	-	-	-
Polygonum hydropiper L.	Polygonaceae	-	-	13500	4.47	-	-
Polygonum plebeium R. Br.	Polygonaceae	-	-	-	-	14500	5.03
Polystichum prescotianum (Wall.) Moore.	Dryopteridaceae	-	-	8000	3.70	-	-
Potentilla arbuscula D. Don.	Rosaceae	-	-	-	-	13000	4.79
Potentilla eriocarpa Wall. ex Lehm.	Rosaceae	-	-	-	-	13000	5.24
Primula calderiana Balf. F. & Cooper	Primulaceae	-	-	-	-	12000	3.73
Primula capitata Hk. f.	Primulaceae	-	-	-	-	3000	1.38
Primula caveana Sm.	Primulaceae	-	-	-	-	6000	1.86
Pteris spp.	Pteridaceae	-	-	1000	0.57	-	-
Ranunculus diffusus DC.	Ranunculaceae	-	-	6000	2.56	-	-
Ranunculus pulchellus C. Meyer	Ranunculaceae	-	-	-	-	15000	2.86
Rheum acuminatum Hook. f. & Thomson ex Hook.	Polygonaceae	-	-	-	-	4000	1.54
Rubus mollucanus L.	Rosaceae	7000	2.44	-	-	-	-
Rumex nepalensis Spreng.	Polygonaceae	-	-	5500	2.49	-	-
Sanicula elata Buch. - Ham. ex D. Don	Apiaceae	-	-	12000	4.69	-	-

Scoparia dulcis DC.	Scrophulariaceae	3000	1.14	-	-	-	-
Selenium tenuifolium Wall. ex C. B. Clarke	Apiaceae	-	-	-	-	12000	3.73
Senecio diversifolius Wall. ex DC.	Asteraceae	-	-	-	-	7500	3.46
Senecio wallichii DC.	Asteraceae	-	-	6500	3.06	-	-
Spilanthes paniculatus Wall. ex DC.	Asteraceae	13000	2.46	-	-	-	-
Stellaria sikkimensis Hk. f. Edgew. & Hk. f.	Caryophyllaceae	-	-	8500	2.91	-	-
Swertia chirayita (Roxb. ex Flem.) Karst.	Gentianaceae	9000	2.77	-	-	-	-
Urtica dioica L.	Urticaceae	-	-	8000	2.84	-	-
Valeriana hardwickii Wall.	Valerianaceae	14000	4.56	-	-	-	-
Viola biflora L.	Violaceae	-	-	14500	4.61	-	-
Viola pilosa Bl.	Violaceae	6000	1.96	-	-	-	-

Annexure 2. Frequency, density (tree^{-1}) and IVI of epiphytes in three forest stands.

		Upper montane			Montane			Lower montane		
Epiphytes	**Family**	**Frequency**	**Density tree^{-1}**	**IVI**	**Frequency**	**Density tree^{-1}**	**IVI**	**Frequency**	**Density tree-^{1}**	**IVI**
Aeschynanthus bracteatus Wall. ex DC.	Gesneriaceae	-	-	-	10	20	1.73	10	20	1.51
Aeschynanthus hookeri Clarke	Gesneeriaceae	-	-	-	-	-	-	20	40	3.02
Aeschynanthus sikkimensis (Clarke) Stapf	Gesneriaceae	-	-	-	-	-	-	10	20	1.51
Agapetes hookeri(Clarke) Sleumer	Ericaceae	-	-	-	10	50	2.35	10	20	1.51
Agapetes incurvata (Griff.) Sleumer	Ericaceae	-	-	-	10	20	1.73	-	-	-
Agapetes serpens (Wight) Sleumer	Ericaceae	-	-	-	-	-	-	10	70	2.47
Agrostophyllum brevipes King & Pantl.	Orchidaceae	-	-	-	-	-	-	10	10	1.32
Agrostophyllum callosum Rchb. f.	Orchidaceae	-	-	-	-	-	-	10	20	1.51
Arthomeris himalayensis (Hk.) Ching	Polypodiaceae	20	60	7.06	-	-	-	-	-	-
Arthomeris lehmanii (Mett.) Ching	Polypodiaceae	-	-	-	10	30	1.94	-	-	-
Arthomeris wallichiana (Spr.) Ching	Polypodiaceae	10	40	3.95	10	30	1.94	-	-	-
Asplenium ensiforme Wall.	Aspleniaceae	-	-	-	10	20	1.73	-	-	-
Belvisia spicata (L.f.) Mirbel	Polypodiaceae	-	-	-	-	-	-	10	110	3.24
Bulbophyllum cauliflorum Hk.f.	Orchidaceae	-	-	-	-	-	-	10	20	1.51
Bulbophyllum reptans (lindl.) Lindl.	Orchidaceae	-	-	-	10	40	2.14	10	50	2.09
Cautleya gracilis(Smith) Dandy	Zingiberaceae	-	-	-	10	50	2.35	10	70	2.47
Cheilanthes formosana Hay.	Sinopteridaceae	-	-	-	10	20	1.73	-	-	-
Codonopsis purpurea Wall.	Campanulaceae	-	-	-	10	20	1.73	-	-	-
Coelogynae corymbosa Lindl.	Orchidaceae	-	-	-	30	30	4.57	20	30	2.82
Coelogynae ochracea Lindl.	Orchidaceae	-	-	-	10	20	1.73	-	-	-
Cystopteris sudetica A. Br.	Davalliaceae	70	340	30.14	-	-	-	-	-	-
Davallia bullata Wall.	Davalliaceae	-	-	-	-	-	-	10	20	1.51
Dendrobium amoenum Wall. ex Lindl.	Orchidaceae	-	-	-	-	-	-	10	20	1.51
Dendrobium longicornu Lindl.	Orchidaceae	-	-	-	-	-	-	10	10	1.32
Dendrobium nobile Lindl.	Orchidaceae	-	-	-	-	-	-	10	10	1.32
Didymocarpus aromaticus Wall. ex D. Don	Gesneriaceae	-	-	-	20	50	3.67	-	-	-
Didymocarpus oblongus Wall. ex D.Don	Gesneeriaceae	-	-	-	-	-	-	20	80	3.79
Drynaria propinqua (Wall.) J. Smith	Drynariaceae	-	-	-	-	-	-	10	20	1.51
Eria coronaria (Lindl.) Reichb.f.	Orchidaceae	-	-	-	-	-	-	10	10	1.32
Eria spicata (D. Don) Hand. - Mazz.	Orchidaceae	-	-	-	-	-	-	10	10	1.32

Ficus spp.	Moraceae	-	-	-	-	-	-	10	10	1.32
Globba hookeri Clarke ex Baker	Zingiberaceae	-	-	-	10	50	2.35	10	70	2.47
Gonatanthus pumilus (D. Don) Engler & Krause	Araceae	-	-	-	-	-	-	10	40	1.89
Hoya fusca Wall.	Asclepiadaceae	-	-	-	-	-	-	10	30	1.70
Hoya lanceolata Wall. ex D. Don	Asclepiadaceae	-	-	-	-	-	-	20	40	3.02
Hoya linearis Wall. ex D. Don	Asclepiadaceae	-	-	-	10	310	7.73	60	1120	28.28
Hymenopogon parasiticus Wall.	Rubiaceae	-	-	-	10	40	2.14	-	-	-
Lepisorus kashyapii (Mehra) Mehra	Polypodiaceae	10	60	4.78	20	140	5.53	20	220	6.48
Lepisorus nudus (Mehra) Mehra	Polypodiaceae	30	60	9.33	50	440	15.69	60	580	17.90
Lepisorus scolopendrinus (Don) Mehra & Bir	Polypodiaceae	10	20	3.11	30	190	7.88	-	-	-
Lepisorus sesquipedalis (J. Sm.) Fras. - Jenk.	Polypodiaceae	10	30	3.53	-	-	-	-	-	-
Leucostegia immersa (Wall.) Presl	Davalliaceae	-	-	-	-	-	-	10	60	2.28
*Lindsaea odorata*Roxb. ex Griff.	Lindsaeaceae	-	-	-	-	-	-	10	20	1.51
Lipparis perpusilla Hk. f.	Orchidaceae	-	-	-	-	-	-	10	30	1.70
Loxogramme involuta (D. Don) Presl.	Loxogrammaceae	-	-	-	10	230	6.08	-	-	-
*Lycopodium japonicum*Thunb.	Lycopodiaceae	-	-	-	-	-	-	10	50	2.09
Lysionotus serratus D. Don	Gesneriaceae	-	-	-	10	30	1.94	-	-	-
Macropanax undulatum Seem.	Araliaceae	-	-	-	10	40	2.14	-	-	-
Mecodium spp.	Hymenophyllaceae	-	-	-	30	180	7.67	30	90	5.10
Medinilla himalayana Hk.f	Melastomataceae	-	-	-	-	-	-	10	10	1.32
Microsorium membranaceum (D. Don) Ching	Polypodiaceae	-	-	-	-	-	-	10	10	1.32
Microsorium punctatum (Linn.) Copel.	Polypodiaceae	-	-	-	-	-	-	10	10	1.32
Nephrolepis cordifolia (L.) Presl.	Nephrolepidaceae	-	-	-	10	20	1.73	30	80	4.91
Oleandra wallichii (Hk.) Presl.	Oleandraceae	30	90	10.58	30	120	6.43	-	-	-
Onychium spp.	Cryptogrammaceae	50	670	39.40	-	-	-	-	-	-
Otochilus alba Lindl.	Orchidaceae	-	-	-	-	-	-	10	10	1.32
Pentapanax leschenaultii Seem.	Araliaceae	-	-	-	-	-	-	10	10	1.32
Pentapanax racemosus Seem.	Araliaceae	-	-	-	10	10	1.52	-	-	-
Peperomia heyneana Miquel.	Piperaceae	-	-	-	-	-	-	10	70	2.47
Peperomia tetraphylla (Forst. f.) Hk. & Arn.	Piperaceae	-	-	-	-	-	-	30	310	9.33
Phalaenopsis tainitis Christenson & Pradhan	Orchidaceae	-	-	-	-	-	-	10	40	1.89
Phlegmariurus phlegmaria (L.) Sen.et Sen	Huperziaceae	-	-	-	-	-	-	10	10	1.32
Pholidota imbricata Hk.	Orchidaceae	-	-	-	10	30	1.94	20	40	3.02
Phymatopteris ebinipes (Hk.) Ching	Polypodiaceae	10	30	3.53	-	-	-	-	-	-
Phymatopteris malacodon (Hk.) Pichi - Serm.	Polypodiaceae	-	-	-	-	-	-	10	50	2.09

Phymatopteris oxyloba (Wall. ex Ktze) Pic. Ser	polypodiaceae	50	120	16.38	-	-	-	-	-	-
Pilea spp.	Urticaceae	20	60	7.06	10	40	2.14	-	-	-
Pleione hookeriana (Lindl.) O. Kuntze.	Orchidaceae	-	-	-	20	200	6.77	-	-	-
Pleione humilis (Smith) D. Don	Orchidaceae	-	-	-	-	-	-	20	180	5.71
Polypodiastrum argutum (Wall. ex Hk.) Ching	Polypodiaceae	30	370	22.30	40	780	21.41	10	220	5.35
Polypodioides amoena Wall.	Polypodiaceae	-	-	-	10	10	1.52	-	-	-
Polypodioides lachnopus Wall.	Polypodiaceae	-	-	-	20	80	4.29	10	20	1.51
Pyrrosia flocculosa (D. Don) Ching	Polypodiaceae	10	20	3.11	-	-	-	-	-	-
Pyrrosia lanceolata (L.) Farwell	Polypodiaceae	-	-	-	-	-	-	20	100	4.17
Pyrrosia stigmosa (Sw.) Ching	Polypodiaceae	-	-	-	-	-	-	10	210	5.16
Remusatia hookeriana Schott	Araceae	-	-	-	-	-	-	10	70	2.47
Rhododendron dalhousiae Hk. f.	Ericaceae	-	-	-	-	-	-	10	20	1.51
Rhododendron pendulum Hk. f.	Ericaceae	-	-	-	20	50	3.67	-	-	-
Roscoea spicata Seem.	Zingiberaceae	10	20	3.11	20	30	3.25	10	10	1.32
Selaginella spp.	Selaginellaceae	-	-	-	-	-	-	10	30	1.70
Smilacina oleracea (Baker) Hk. f.	Convallariaceae	10	60	4.78	10	210	5.66	-	-	-
Utricularia multicaulis Oliver	Lentibulariaceae	-	-	-	10	20	1.73	-	-	-
Vaccinium nummularia Hk. f. & Thoms. ex Clarke	Ericaceae	-	-	-	10	130	4.01	-	-	-
Vaccinium retusum (Griffith) Hoo. f. ex Clarke	Ericaceae	30	210	15.60	10	10	1.52	-	-	-
Vaccinium serratum Wight	Ericaceae	20	50	6.64	60	160	11.21	-	-	-
Vaccinium vacciniaceum (Roxb.) Sleumer	Ericaceae	-	-	-	30	40	4.78	30	160	6.45
Vandopsis undulata (Lindl.) Smith	Orchidaceae	-	-	-	-	-	-	10	10	1.32
Vittaria elongata Swartz	Vittariaceae	-	-	-	50	720	21.49	40	380	11.80
Vittaria flexuosa Fee	Vittariaceae	-	-	-	10	30	1.94	10	20	1.51
Vittaria himalayensis Ching.	Vittariaceae	10	80	5.62	10	80	2.97	-	-	-
Vittaria sikkimensis Kuhn.	Vittariaceae	-	-	-	-	-	-	10	90	2.85
Wightia speciosissima (D. Don) Merr.	Scrophulariaceae	-	-	-	10	10	1.52	10	10	1.32

Chapter 5
Pattern of forest fragmentation in Khangchendzonga Biosphere Reserve

5.1 Introduction

Fragmentation of continuous forests into smaller patches has serious consequences on the survival of species and ecosystems. Fragmentation alters microenvironment and increases the vulnerability of the forest communities (Lovejoy et al. 1984, 1986; Lord & Norton 1990; Robinson et al. 1992; Matlack 1994). As forest landscapes become increasingly fragmented, populations of forest species are reduced, dispersal and migration patterns are interrupted, ecosystem inputs and outputs are altered, and previously isolated core habitats are exposed to external conditions, all of which result in a progressive erosion of biological diversity (Terborgh & Winter 1980; Tilman et al. 1994). Forest fragmentation is a dynamic process in which the habitat is progressively reduced into smaller patches that becomes more isolated and increasingly affected by edge effects (Forman & Godron 1986; Reed et al. 1996; Franklin 2001; McGarigal 2002). The fragments of irregular shape tend to have increased edge lengths (Echeverria et al. 2007) and therefore, total species richness in smaller fragments is significantly lower than the larger ones (Metzger et al. 1997). Thus, conversion of continuous forests into forest fragments and large fragments into smaller fragments has been described as the most important factor of species and ecosystem loss in tropics (Turner 1996).

Forests may be fragmented by anthropogenic or natural activities or events, such as road construction, logging, conversion to agricultural land, and wildfire. The size of the forest fragment is a function of causes of disturbance, history of forest and disturbance, disturbance intensity and frequency, and often forest management interventions. Changes in land use and land cover could be an effective indicator of forest fragmentation. Land use refers to man's activities on land, whereas land cover denotes natural vegetation cover, water bodies etc. Because of conversion of forest lands into

croplands and pastures, and deforestation, reforestation and afforestation activities, both land use and land cover change in a forested landscape (Kilie et al. 2004). Forest fragmentation due to land use changes is considered to be the most important reason for biodiversity decline in forest ecosystems (Kilie et al. 2004; Matsushita et al. 2006). The ecological consequences of fragmentation may differ depending on the fragmentation patterns, i.e. spatial configuration imposed on a landscape, and their temporal and spatial variations (Echeverria et al. 2006; Cayuela et al. 2006). Therefore, an understanding of the temporal and spatial patterns of fragmentation and its impact on populations is a prerequisite for land use and biodiversity management in any landscape.

Khangchendzonga Biosphere Reserve (KBR) in Eastern Himalayas, Sikkim is facing the increasing incidences of cattle grazing, landslide, forest fire and wind-throw. All these disturbances have brought about discontinuity in forest cover in many parts of the Biosphere Reserve, thereby fragmenting the natural landscape as well as habitats of several plant species. However, no quantitative data on fragmentation pattern, fragmentation dynamics and causes of fragmentation is available for KBR. Therefore, this chapter has been designed (i) to study the spatial and temporal patterns of forest fragmentation, and (ii) to analyze the causes and intensity of disturbances causing forest fragmentation. The consequences of the forest fragmentation on plant species populations have been presented in the next chapter.

5.2 Methods

Fragment spatial variables

Fragment size, and isolation of forest fragments were evaluated to study forest fragmentation in KBR. Both the attributes were estimated on ArcView 3.2 platform using the Spatial Analyst 2.0 extension (ESRI 1999). For estimating Fragment size (*FS*), the surface area (ha) of the fragment was calculated (McGarigal & Marks 1994).

Isolation of forest fragments was examined in terms of their proximity to surrounding fragments. Isolation Index (*II*; adapted from Forman 1997 & Cook 2002):

$$II = \frac{1}{N}\sum dij$$

where *II* = Isolation Index, *N* = all neighbouring fragments within a radius of 1 km from the focal fragment; *dij* = distance (patch *i* and neighbouring patch *j*). Higher *II* values indicate higher isolation.

Land use and land cover mapping

The land use and land cover of the KBR was mapped using multi-dated satellite imageries and through extensive ground truthing for each land use type following stratified random sampling method. Survey of India (SOI) topographical maps were used for geocoding the imageries. The satellite images used in this study are Landsat Multispectral Scanner (MSS) imagery of 1999, Landsat Thematic Mapper (TM) imagery of 2002 and Indian Remote Sensing Satellite (IRS) 1D LISS III imagery of 2008. The satellite images were rectified or geometrically corrected using Ground Controlled Points (GCPs) obtained from topographical sheets and the GPS points collected from the field survey. Points such as intersection of the roads, river junctions and permanent establishments were identified on the topographical sheets as GCPs. By using polynomial equations the scene was geometrically corrected and geo-referenced using a UTM (Universal Transverse Mercator) projection system. The spheroid and datum used were WGS 84 and the UTM Zone 45N. The pixels of the satellite images were re-sampled using a maximum likelihood algorithm and the study area was extracted from the scene using a digitized state boundary, Biosphere Reserve boundary, buffer area boundary and core area boundary from the Survey of India topographical sheets. Sub-pixel image to map accuracy was achieved through repeated attempts. Histogram matching was done to correct the radiometric differences, when present.

Supervised digital image classification method was carried out to delineate different land uses viz., dense forests, open forests, barren land, meadow, snow cover area, glacier beds and water bodies. Supervised method of classification can be defined normally as the process of samples of unknown identity and are those pixels located within the training sites. A brief description of each of the land cover classes is given in Table 5.1.

In this type of classification, spectral signatures are developed from specified locations in the image. These specific locations are given a generic name 'training sites' and are defined. These training sets help in developing the outline areas. Multiple polygons are created for each land category to delineate relevant land use type. These signatures will then be used to classify all pixels in the scene. Sufficient Ground Controlled Points (GCP) were taken to confirm the different land use types. Nearest Neighbourhood Analysis was done for post classification smoothening. The steps followed for classification have been summarised in figure 5.1.

Table 5.1. Land cover classes of KBR.

Land cover classes	Description
Barren land	Soil less than 1 cm in thickness, devoid of any vegetations/exposed rocks
Snow/glacier	Huge mass of ice originating from an accumulation of snow
Meadows	Mainly the high elevational alpine meadow/pasture land
Open forests	Interrupted forest due to forest gaps, whose canopy cover is less than 10%
Dense forests	Uninterrupted forest cover, whose canopy cover is greater than 10%
Water bodies	Lakes/rivers/streams/wet lands

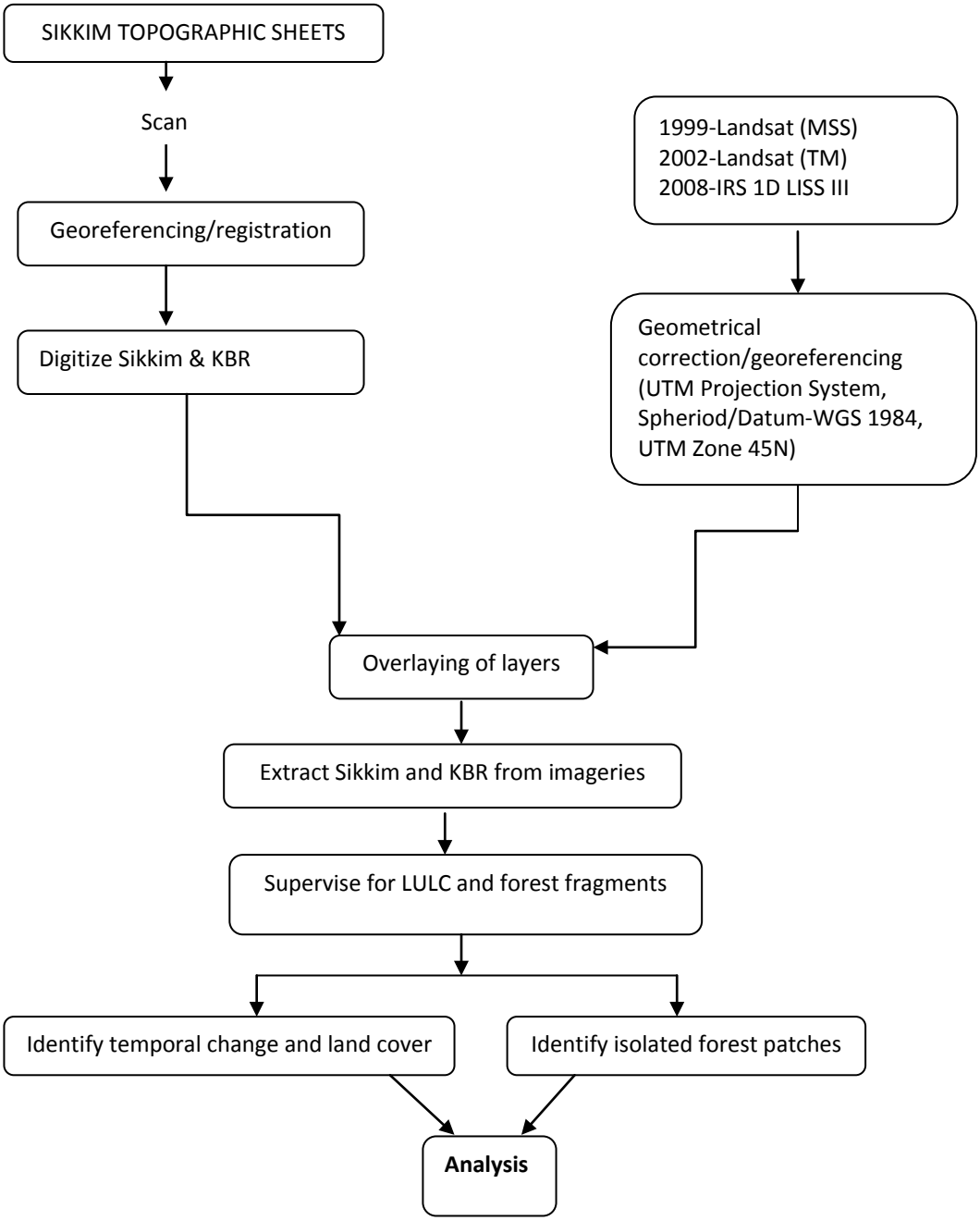


Figure 5.1. Flow chart for studying land use/ land covers changes and fragment dynamics in KBR.

Collection of Ground Control Points for fragmentation studies

KBR was extensively surveyed for locating the forest fragments. Twenty five fragments were located in the KBR spread over three forest types. One fragment was located in Lower montane forests, 14 in Upper montane and 10 in Montane forests. With the help of a Garmin Global Positioning System (GARMIN model *Map* 76), geographical coordinates for each of these 25 fragments were collected (Table 5.2) and mapped. The changes in forest fragmentation pattern was studied by comparing the time series data i.e.

imageries of 1999, 2002 and 2008 as described in the earlier section. Visual interpretation technique was used for mapping the fragment locations and sizes in the entire KBR.

Table 5.2. Characteristics of the 25 forest fragments in Khangchendzonga Biosphere Reserve, Sikkim.

Fragment number (FF)	Aspect	Slope (°)	Elevational range (m)	GPS Coordinate (at the centre of fragment)	
				Latitude (N)	Longitude (E)
FF1	South	10-15	2525-2550	27°22.020’	88°06.650’
FF2	East	10-25	2560-2565	27°22.356’	88°06.667’
FF3	East	15-30	2740-2770	27°22.457’	88°06.450’
FF4	East	10-20	2800-2900	27°22.500’	88°06.249’
FF5	South-East	30-50	3018-3600	27°22.675’	88°05.850’
FF6	South-east	5-20	3455-3530	27°22.951’	88°05.300’
FF7	South-west	10-20	3500-3570	27°23.249’	88°05.130’
FF8	South-west	10-25	3485-3500	27°23.352’	88°05.010’
FF9	West	20-40	3480-3600	27°23.528’	88°04.921’
FF10	South-west	10-35	3530-3570	27°23.463’	88°04.775’
FF11	South-west	10-45	3584-3700	27°23.600’	88°04.575’
FF12	South	20-40	3700-3855	27°23.415’	88°05.685’
FF13	South-east	10-20	2545-2552	27°25.657’	88°11.288’
FF14	East	10-35	2709-2725	27°25.705’	88°11.135’
FF15	South	10-40	2900-2930	27°25.850’	88°10.950’
FF16	South	15-20	2940-2985	27°25.974’	88°10.826’
FF17	South-east	10-15	3283-3290	27°26.730’	88°10.770’
FF18	North-east	5-10	3800-3840	27°27.025’	88°05.320’
FF19	South-east	15-20	3770-3790	27°27.210’	88°05.325’
FF20	West	20-25	3830-3930	27°27.260’	88°05.630’
FF21	South	25-35	3125-3139	27°46.350’	88°32.600’
FF22	East	30-40	3088-3117	27°46.110’	88°32.565’
FF23	South	30-45	1700-1760	27°19.420’	88°09.672’
FF24	South-west	30-40	3250-3300	27°47.22’	88°33.04’
FF25	South-west	30-45	3179-3380	27°46.65’	88°32.97’

Changes in forest fragments

An annual fragment creation rate was calculated using the following formula (Puyravaud 2003):

$$P = \frac{100}{t_2 - t_1} \ln \frac{A_2}{A_1}$$

where, *P* is percentage of forest loss per year, and A1 and A2 are the amount of forest cover/area under fragments at time t_1 and t_2, respectively.

Causes of forest fragmentation

Grazing activity, NTFPs cultivation and extraction, taungya plantation systems, where crops were grown between rows of trees during early ages of plantation, and wild fire and landslides were the main causes of disturbance causing forest fragmentation in KBR. The causes of fragmentation for each of the 25 fragments, and the occurrence and relative frequency of disturbance related to these 25 fragments in KBR were studied in detail pertaining to the period 1999-2008. Data on the intensity and extent of disturbance within each forest fragment were based on site inspection and information obtained from the interviews of the local people. Interviews from the local residents, cattle herders, and people living in the buffer zone generated extremely useful information and strengthened the ground truth data gathered from the field study.

All the disturbance factors were identified in the field and were given '-' score if that particular disturbance factor (e.g. NTFP collection) was absent in a given year from all the 25 fragments and '+' score if the factor was present during that particular year in any of the 25 fragments. The total score for a particular causative factor over the period 1999-2008 was thus calculated. Depending on the total presence/absence score of a particular causative factor across the ten years period, frequency of occurrence of a particular disturbance causing factor was obtained by dividing the score with the total number of disturbance causing factors i.e. 12. The relative frequency of a causative factor thus obtained was finally expressed in percentage.

Categorisation of intensity of disturbance as 'high' 'low' and 'medium' was done based on the impacts of 12 factors of disturbance on fragments during the survey. Numbers of disturbance factors affecting each fragment were also noted in order to differentiate the severity of disturbance in different fragments.

5.3 Results

Fragment spatial variables

The size of 25 fragments located for detailed ground level studies ranged between 0.1 and 72.2 ha. The mean size of these 25 fragments was 9.3 ± 0.5 ha (SE). Values of isolation index ranged from 0.01 to 8.1 and the mean value being 0.51 ± 0.3 (SE) (Table 5.3).

Table 5.3. Spatial variables for the 25 fragments in KBR.

Fragment number	Fragment size (area in km^2)	Isolation index
FF1	0.02	0.2
FF2	0.03	0.5
FF3	0.04	0.8
FF4	0.05	1.2
FF5	0.06	1.5
FF6	0.07	1.8
FF7	0.08	2.2
FF8	0.09	2.5
FF9	0.10	2.8
FF10	0.11	3.2
FF11	0.12	3.5
FF12	0.13	3.8
FF13	0.14	4.2
FF14	0.15	4.5
FF15	0.16	4.8
FF16	0.17	5.2
FF17	0.18	5.5
FF18	0.19	5.8
FF19	0.20	6.2
FF20	0.21	6.5
FF21	0.22	6.8
FF22	0.23	7.2
FF23	0.24	7.5
FF24	0.72	7.8
FF25	0.72	8.2

Decadal land use changes during 1999-2008

Land use/land cover maps for 1999, 2002 and 2008 of KBR clearly differentiated various land use/cover types. Across the years, major portion of land use was dominated by dense forests (Figure 5.2). Lower montane and Montane forests in KBR were distinguishable.

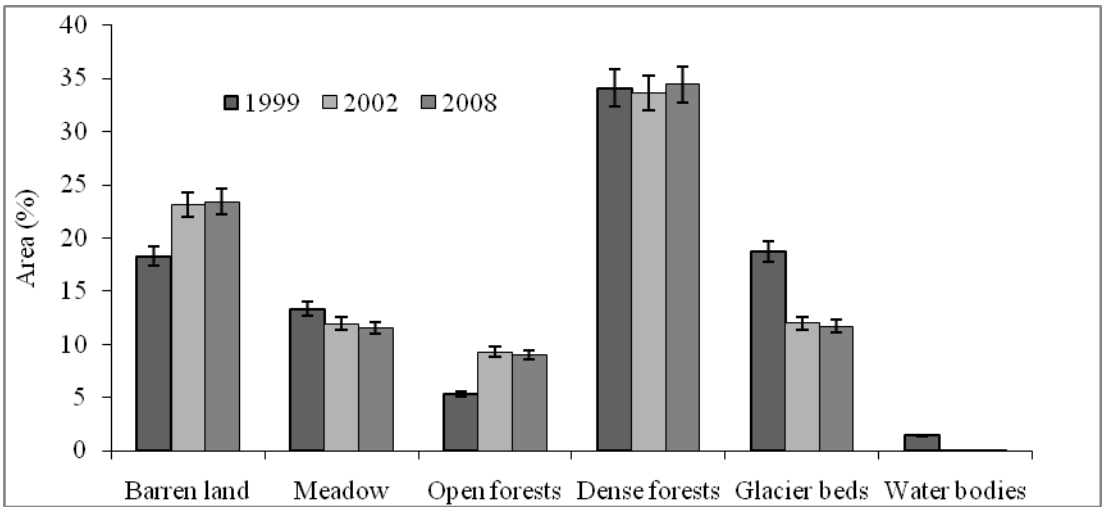


Figure 5.2. Temporal changes in land use and land cover in KBR during 1999-2008.

(Figure 5.3). Areas under dense forests, lakes (water bodies), and snow cover remained more or less same and did not show any remarkable changes during the study period. Barren land, devoid of any vegetation covers and characterised by stony rocky-lands increased during the study period by 5% (132.6 km^2). The open forest areas also increased by 92.6 km^2 during the study period. Contrastingly, glacier beds and meadows that covered 832 km^2 in 1999, decreased to 603.3 km^2 in 2008 (Table 5.4 and Figure 5.2).

Table 5.4. Land use changes in KBR during 1999 – 2008.

	Area (km^2)			Net change (km^2)	
Land use classes	**1999**	**2002**	**2008**	**1999-2002**	**2002-2008**
Barren land	600.6	608.4	475.8	-7.8	132.6
Snow	254.8	254.8	228.8	0	26
Meadow	309.4	299	345.8	10.4	-46.8
Open forests	241.8	234	137.8	7.8	96.2
Dense forests	873.6	894.4	886.6	-20.8	7.8
Glacier beds	312	304.2	486.2	7.8	-182
Water bodies	2.6	2.6	36.4	0	-33.8

Forest fragmentation dynamics during 1999-2008

Forest fragmentation pattern during the ten years of study revealed that there was an increase in number of smaller fragments (Figure 5.4). The number of fragments in 1999 was 875, which reduced to 533 in 2002. However, in 2008, it increased to 615. The number of forest fragments < 1 ha during 1999 was 515 (112 ha), which decreased to 295 (86.3 ha) in 2002 but again increased in 2008 to 341 (108.5 ha) fragments. Forest

fragments in 1-50 ha size class were also more in 1999 with 350 fragments covering an area of 2420.9 ha; while in 2002, it was 231 fragments, covering an area of 1668.7 ha. It increased to 267 fragments with 1764 ha of area by 2008. Forest fragments larger than 50 ha size class were only 10 (1368.6 ha) in 1999, which decreased to seven fragments each in 2002 and 2008, having an area of 688.9 ha and 517.5 ha respectively (Figure 5.5). During 1999, 51.3% of the forest area was in smaller fragments between 0-1 ha; 47.5% in medium size fragments and 1.1% in large fragments. By 2008, the number of fragments under smaller patches decreased to 40.8%, while, the medium size fragments increased to 58% and larger fragments remain stable. During the whole study period, the average annual fragmentation rate was 0.7 ha $year^{-1}$, equivalent to 0.007%. The mean fragment size decreased from 4.4 ha in 1999 to 3.9 ha in 2008 (Table 5.5). This decline in mean patch size was associated with decrease in patch density and a substantial reduction in the size of the large forest fragments during the study period.

Table 5.5. Fragment dynamics in KBR during 1999-2008.

Fragment size (ha)	No. of fragments(Area in ha)			Mean fragment size (ha)			Fragment density (No. of fragments/100 ha)		
	1999	2002	2008	1999	2002	2008	1999	2002	2008
<1	515(112)	295 (86.3)	341(108.5)	0.2	0.4	0.4	0.173	0.088	0.097
1-50	350(2420.9)	231(1668.7)	267(1764)	5.8	5.5	5.0	0.160	0.114	0.137
>50	10(1368.6)	7(688.9)	7(517.5)	136.9	98.4	73.9	0.004	0.003	0.003
Total	875(3901.6)	533(2443.9)	615(2390)	4.4	4.5	3.9	0.337	0.205	0.237

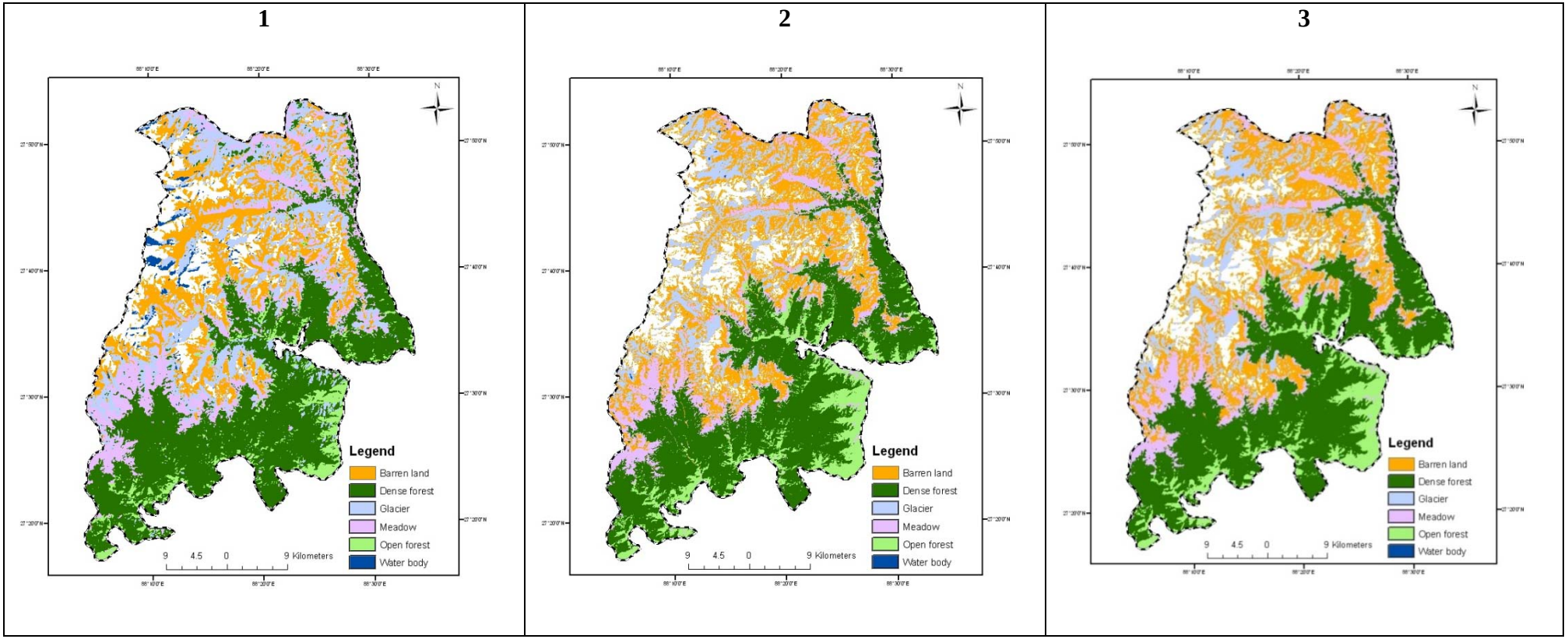


Figure 5.3. FCC depicting the land use/landcover status of the KBR during 1999 (1), 2002 (2) and 2008 (3).

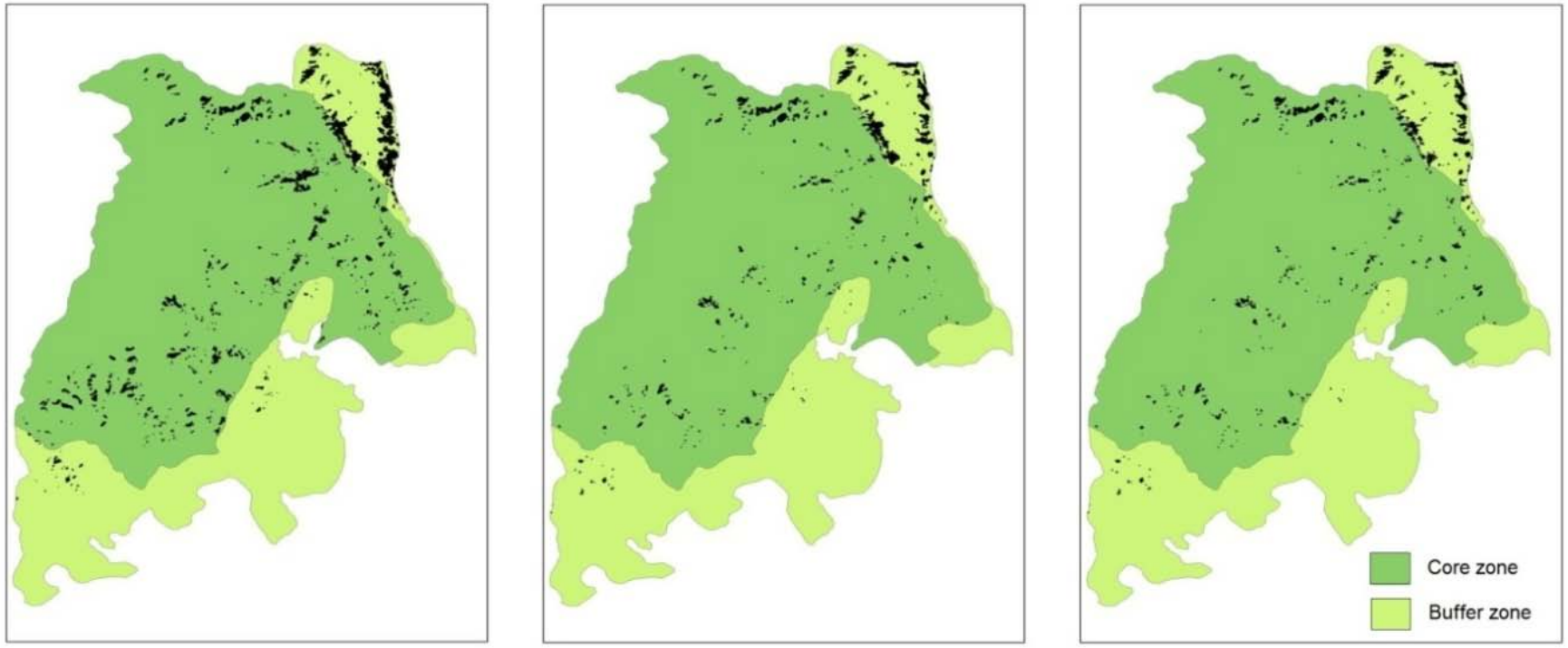


Figure 5.4. Pattern of forest fragmentation in KBR.

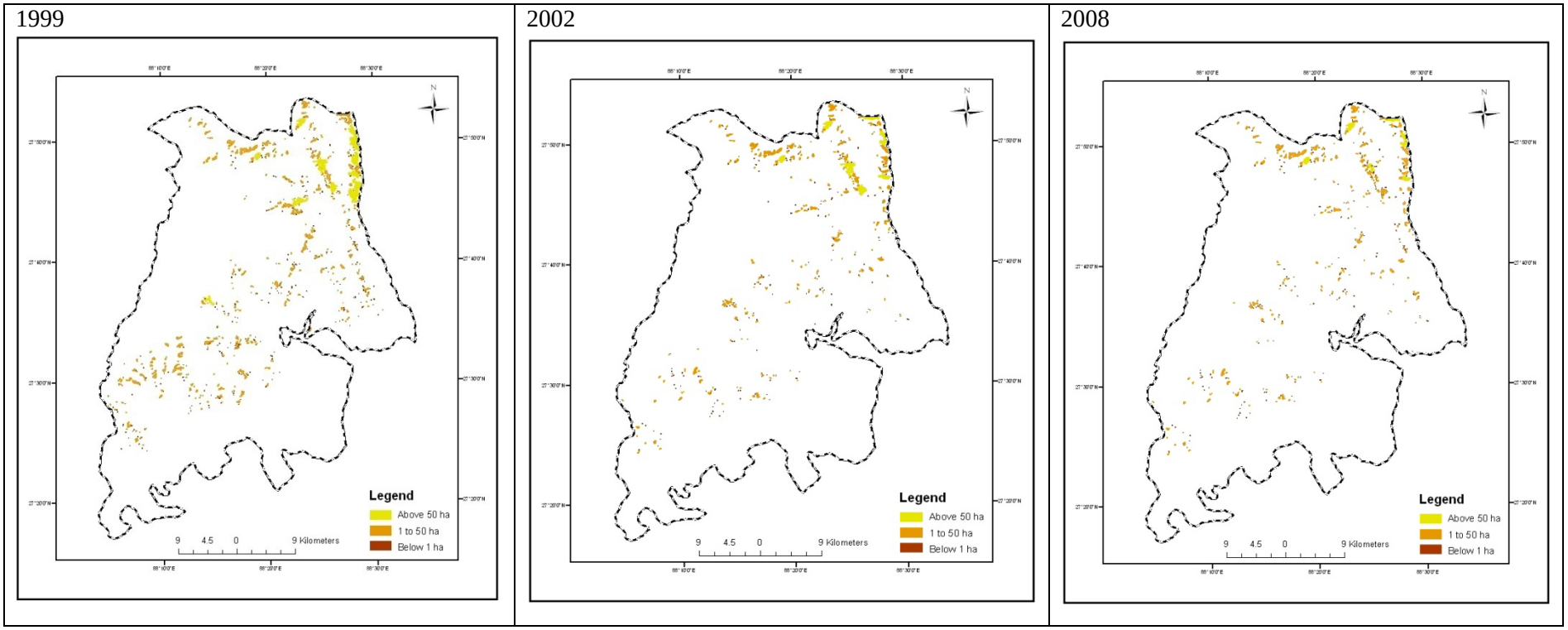


Figure 5.5. Distribution of forest fragments in three size classes in KBR during 1999, 2002 and 2008

Intensity and causes of forest fragmentation

The causes of forest fragmentation were studied in 25 fragments categorized into three fragment size classes i.e. <1, 1-50 and >50 ha. These fragments were created due to anthropogenic (agriculture, grazing, NTFPs cultivation/extraction, timber/poles, trekking routes, settlement, tourism, road) and natural (wind-throw, landslide, snow avalanche, wild fire) disturbances. The mean number of causative factors per fragment reduced from 3 during 1999, to 2 during 2008 (Figure 5.6). In general, anthropogenic causes of disturbances such as agriculture, NTFPs cultivation/extraction, and agriculture decreased by 87% in 2008. But disturbances from the natural causes remained same in KBR (Figure 5.6).

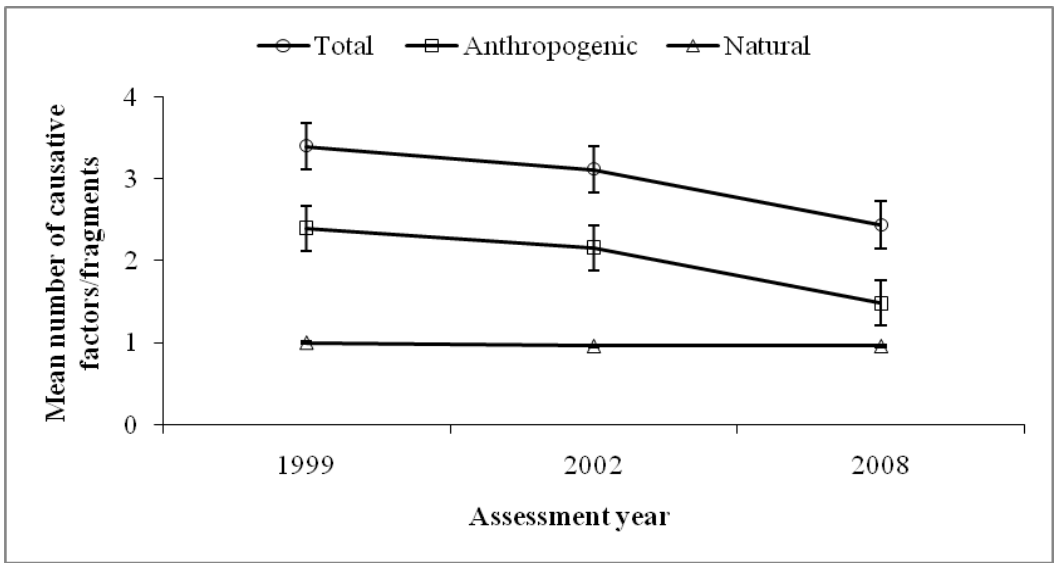


Figure 5.6. Mean number of causative factors per fragment during 1999, 2002 and 2008 in KBR

The percentage of fragments affected due to of trekking route, grazing, tourism, were high during the study period (Figure 5.7). Occurrences of medium and high intensity of disturbances were also common during the study period (Figure 5.8). Fragments created due to high intensity disturbance like wind-throw, landslide, snow avalanche and wildfire though were less in number, the relative frequency of occurrence of such disturbances were greater and often devastating.

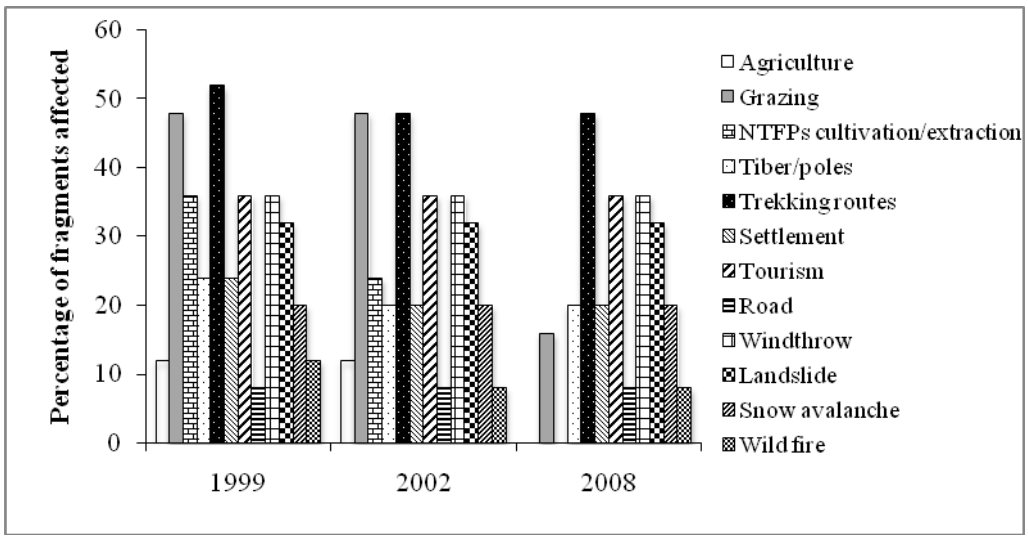


Figure 5.7. Percentage of fragments affected by various causative factors.

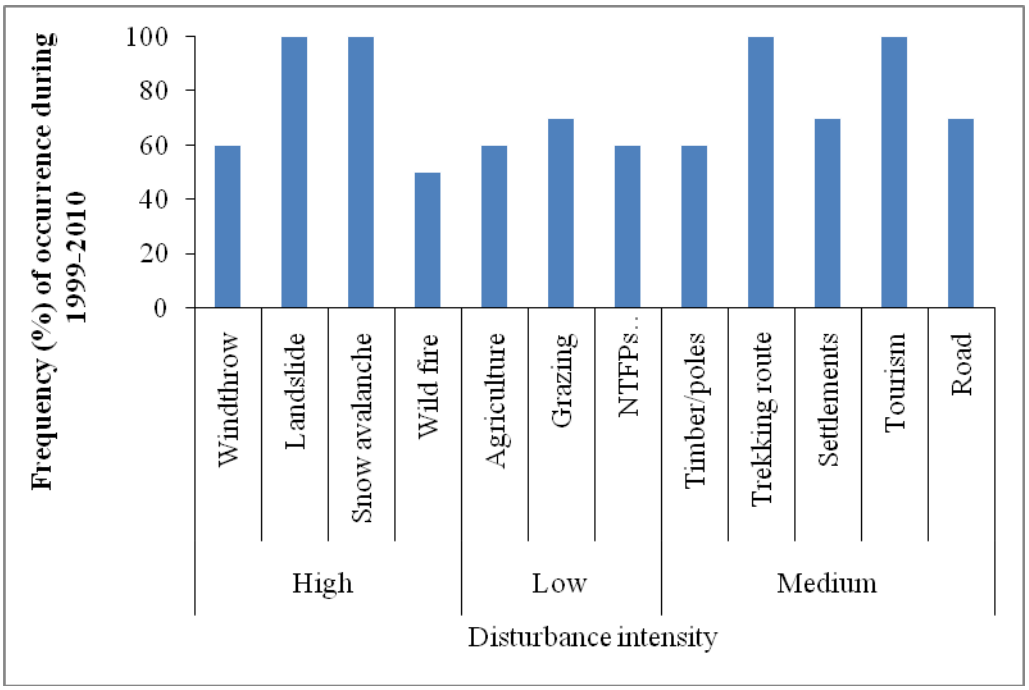


Figure 5.8. Relative frequency of occurrence of various causative factors of disturbance and their intensity in 25 fragments of KBR during 1999-2008.

5.4 Discussion

The results of the study on temporal change in forest fragmentation pattern in KBR helped in detecting the type of change, location of change, and quantifying the changes taking place in KBR. Land cover changes over the study period show conversion of forest category into other classes thereby fragmenting the natural forest cover. Results from the imageries confirm decrease in meadow and open forest areas during the study period,

which might be attributed to extensive grazing by cattle and human disturbances especially in the high elevational belt forest in the past. The decrease in glacier beds on other hand might be related to global climate change phenomena.

Forest fragmentation did not occur as continuous process. The pattern and degree of fragmentation were functions of topology, climatic condition and production activities. In the present study, the forest fragmentation was more during 1999 than in the subsequent time series. Analysis was successful in creating a temporal profile of forest fragmentation for KBR. As evident from the imageries, there has been discontinuity in the formation of forest fragments. On average, 674 (± 103.1) fragments were present in the KBR during the study period and major portion of fragments were in the smallest size classes (< 1 ha). The number of forest fragments during the last decade (1999 Landsat imagery) was at its peak and declined towards 2008. Average annual fragmentation in the KBR was 0.007%, which is comparatively less than others studies around the world. Keles et al. (2008) in their study at Trabzon province reported higher annual rate of forest fragmentation (0.41%), compared to the present finding in KBR. However, Li et al. (2009) in their study in Alabama forest have shown that the rate of forest loss may go as high as 2% annually.

In the higher size classes, fragments were negligible. Lesser number of fragments in higher size classes will have significant effect on the response of some species in the study area. Whereas, abundance of large number of forests fragments in lower size class might be due to landslide, a universal phenomenon in the mountain ecosystem, and hilly areas of Sikkim. There had been events of heavy mud avalanches in 1995, which affected the whole state of Sikkim, which might have created the forest fragments. The prevalence of poor soil conditions, and increasing poverty due to absence of alternative economic options might have also contributed to forests fragmentation in the KBR.

The reduction in the number of forest fragments in KBR during the study period might be due to: (i) complete disappearance of a certain number of forest fragments converting them into continuous forest blanks, and (ii) merger of forest fragments with main continuous forests through regeneration, especially in the montane and upper montane forest matrix (Figure 5.9).

Figure 5.9. Regeneration of Silver fir (*Abies densa*) following disturbance due to fire in KBR.

The former factor contributed more to the reduction of forest fragments in KBR. In effect, smaller fragments are bound to disappear faster than medium or large patches. On the positive side therefore, it is noteworthy that the proportion of medium fragments are higher than smaller fragments. This opens up prospects for reclaiming back forest fragments into the main forests.

The increase in fragmentation is related either to natural or anthropogenic sources (Wade et al. 2003; Geist & Lambin 2001). The forest fragmented by anthropogenic factors is at higher risk of further fragmentation or removal than forest fragmented by natural causes. Identifying only human-cause of forest fragmentation may be a useful tool for policy and decision makers, allowing for improved risk assessments and better targeting of areas for protection or remediation.

Ranking of disturbance parameters according to its visually observed intensity although seems arbitrary, it does characterize various disturbances that took place during the study period. At the forest scale, pattern of disturbance may be strongly influenced by topography, pre-existence of matured forests, time since past disturbance and location in the study sites. But fragmentation due to chronic age old and low intensity disturbances like illegal transboundary grazing and trekking routes have substantially contributed to the creation of forest fragments in KBR.

Cayuela et al. (2006) examined the clearance and fragmentation of tropical montane forests in the Highlands of Chiapas, Mexico using Landsat imagery from 1975, 1990 and 2000 and observed an increasing rate of fragmentation over this region. Echeverria et al. (2006) focused on the rapid deforestation and fragmentation of Chilean temperate forests and they also reported an increasing fragmentation over 25 years (1975, 1990 and 2000). In the current analysis, forest fragmentation was greater in 1999. The study revealed a decelerated rate of forest fragmentation in KBR over the time period 1999-2008. As far as forest cover is concerned, the level of disturbance was not so severe and therefore no remarkable change in the dense forest area was observed during the study period. The decreasing trend in disturbance level and fragment formation with time in KBR might be attributed to strict rules and regulations enforced by state forest department on forest grazing, imparting awareness and knowledge on forestry to local people living in the fringe areas of KBR and also protection measures taken by some local bodies, NGOs (KCC, Himal Rakshak, etc) to safeguard the ecosystems of KBR.

The relationship between forest fragmentation and forest disturbances is important to facilitate future forest landscape management and monitoring actions. The present work has provided useful information to local land use/KBR managers for developing ecologically sustainable forest management strategies and biodiversity conservation practices.

Chapter 6
Tree diversity along the fragment size gradient

6.1 Introduction

Fragmentation of tropical forest has been described as the single greatest threat to global biological diversity (Turner 1996; Laurance 1999). Fragmentation decreases species number and alters community composition because of reduction in forest size, a change in forest shape and an increase in isolation from the remaining forest fragments. Fragment size has been reported to be the major determinant of changes in woody plant composition and guild structure in montane forests (Tabarelli et al. 1999). This process can cause differences not only in diversity but also in composition between fragments and continuous forest. Fragment size is related positively to the presence of primary forest species (Grashof-Bokdam 1997; Bender et al. 1998). These species fail to establish in small forest fragments because they lack the interior conditions necessary for the survival of these species (Kremen et al. 1994; Forman 1999). The large forest fragments support larger plant populations as well as provide required habitats for smaller populations facing the threat of extinction (Rosenzweig 1995).

Various types of human interventions in continuous forests result in different plant diversity patterns (Peres & Barlow 2004; Tilman & Lehman 2001). Once fragments are created, plant dynamics change owing to difference in the forest spatial organization, such as the size, shape, and isolation of forest fragments, edge effects, invasion of foreign species, and other disturbances (Hobbs & Yates 2003; Laurance et al. 2002). Because of changes in local conditions, some plant species may be disproportionately favoured, achieving dominance, while others may face local extinction (Hobbs & Yates 2003). Fragmentation can generate changes in the availability of moisture, nutrients, light, wind, and overall microenvironment. In general, fragment size can limit population size of many species of plants and animals. When a continuous forest is disturbed and

fragmented through a natural or anthropogenic cause, the situation generally leads to a decrease in floristic diversity. However, not many evidences are available to support this hypothesis.

Vegetation in the fragments is more exposed to adverse microclimatic conditions that produce drier and warmer growing conditions than in the original forests (Camargo & Kapos 1995; Chen et al. 1995; de Casenave et al. 1995; Malcolm 1998). As a result, shade tolerant plant species along the edge are replaced by species that are found in open areas (Lovejoy et al. 1986). Edge effects further reduce the area within the fragments occupied by original forest species. As the size of the fragment is reduced, it reaches a critical threshold below which all parts of the fragment become edges, thus eliminating many of the original shade tolerant species and reducing the overall diversity (With & Crist 1995).

Recent research has shown that fragmentation produces severe changes in the demography and community attributes of trees present before disturbance (Turner et al. 1996; Laurance et al. 1997, 1998, 2000; Curran et al. 1999; Gascon et al. 2000). The rich biodiversity of KBR is also facing natural and anthropogenic disturbances, thereby fragmenting the natural habitats of tree species. In this chapter, an analysis has been done to show the impact of fragmentation on tree species composition in the forest. The composition of tree species in different forests fragments in the three forest types has been compared with the diversity in the adjacent continuous forests. The following questions have been addressed in this chapter: (i) Do larger forest fragments have a greater diversity of woody plants than the smaller ones? (ii) How the microclimatic variables differ spatially i.e. fragments vis-a-vis continuous forest, along a fragment size gradient, and from forest edge to the centre within a fragment; (iii) Can such spatial variations in microclimatic conditions be related to the observed pattern of tree diversity?

These findings can be used for the effective management of forest fragments and for the conservation of tree species.

6.2 Methods

Tree species enumeration

The study was conducted in all the three forest types, viz., Lower montane, Montane and Upper montane forests of KBR. Twenty five fragments of varied areas and shapes (details are in chapter 5), adjacent undisturbed continuous forest protected as a part of BR or often a forest corridor, and forest gaps created due to various natural and anthropogenic disturbances were selected for the detailed studies on tree diversity as affected by fragmentation. The ecotone between the forest fragments and the surrounding vegetation were almost absent as the boundaries of the fragments and continuous forests were clearly divided.

Four belt transects of varying sizes (depending upon the size of the fragment) were laid in each of the 25 fragments from periphery to the centre of the fragment. 10 m x 5 m plots were used as sampling units for trees within the transects in all the fragments. The number of sample plots laid in each fragment is listed below. The number of plots laid in the adjacent continuous forest was same with the corresponding fragment:

Fragments No.	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25
No. of plots	16	8	8	8	20	20	20	16	16	20	20	12	8	12	16	8	16	16	16	16	8	12	16	40	40

Measurement of microenvironmental variables

The microenvironmental variables were measured from the periphery of the fragment up to 120 m deep towards the fragment centre. The measurements were undertaken at following distance class intervals: 0-10 m (forest gap boundary/outer edge of the fragment to 10 m deep inside towards the centre of the fragment), 10-20, 20-30, 30-40,

40-50 and 110-120 m. The methods of measurement of various microenvironmental factors were same as described in Chapter 4.

Data analyses

Diversity

Shannon's diversity index (*H*), Pielou's evenness index (*J),* and Fisher's Alpha diversity (α) were analysed using Species Diversity and Richness package 4.1.2 (PISCES Conservation Ltd. 2007). Abundance data on tree species in all the fragments and adjacent continuous forests were used in diversity analysis. Alpha diversity (α) was measured as species richness per plot. 'β' diversity was calculated following Whittaker (1960) as: $\beta = (S/\alpha)-1$, where 'S' is the total number of species encountered in the two sites counting each species only once and 'α' is the mean species richness of two sites.

Ordination of tree species in different fragment sizes

The ordination methods used were Canonical Correspondence Analysis (CCA) and Detrended Correspondence Analysis (DECORANA or DCA). The tree species abundance data in different forest fragments were used to classify the fragments based on species composition using TWINSPAN (Two way Indicator Species Analysis) program of the Community Analysis Package (CAP 4, Version 4.1.3; PISCES Conservation Ltd. 2007). A total of 71 tree species from 25 fragments were classified using the TWINSPAN program. TWINSPAN is a hierarchical, polythetic divisive classification technique that classifies vegetation communities according to their floristic similarity (Hill 1994; Kent & Coker 1996). It can characterize the samples by "differential species" that are prevalent on the one side of the dichotomy (Hill et al. 1975; Hill 1994). In TWINSPAN, fragments are categorized based on the similarities and dissimilarities in species composition. The default options (maximum number of indicators per division = 5, maximum level of division = 6 and maximum size of group to be divided = 5, all pseudospecies were given equal weighting) of the TWINSPAN program were used as recommended (Pisces

Conservation 2007). DECORANA (DCA) was used with TWINSPAN for better interpretation of tree communities.

Identification of indicator species

Indicator species are those that occur predominantly among the samples at one end of an ordination axis. Their distribution should reflect, i.e. indicate the environmental characteristics of the samples at either end of the gradient depicted by the ordination axis. Samples along the ordination axis are divided into two groups about the centre of the axis. Those to the right of the centre are placed in one group, called the positive group (+); those to the left are placed in a different group, called the negative group (-).

6.3 Results

Trees diversity in continuous forest and forest fragments

Correlation between Fishers alpha diversity and the fragment size was insignificant (R = -0.13, P = 0.54) indicating no relationship between alpha diversity and fragment size.

Tree species diversity index was lower than that of adjacent continuous forests for all the fragments. This indicates that diversity of tree decreased with fragmentation of forests. Species composition was independent of fragment size in all the forest types (Table 6.1).

Table 6.1. Fragments size, tree diversity summary for 25 forest fragments and adjacent continuous forests in KBR, Sikkim.

Forests fragments	**Upper montane**														**Lower montane**	**Montane**									
Fragment size (ha)	7.68	3.04	6.72	3.84	6.72	6.4	2.57	3.1	3.78	4.47	1.02	2.07	72.2	72.2	4	3.2	1.92	1.6	1.99	9.92	1.6	2.93	5.86	1.9	0.1
Species richness	8	8	6	6	9	8	7	9	9	10	9	10	8	8	13	22	11	10	9	13	7	8	14	6	5
Species number	42	37	32	33	45	54	25	37	33	39	23	31	101	102	31	49	15	15	15	49	19	29	39	22	10
Shannon-Weiner index	1.91	1.86	1.55	1.48	1.99	1.84	1.67	2.01	2.02	2.11	1.99	2.10	0.98	0.98	2.41	2.95	2.18	2.06	1.85	2.43	1.71	1.81	2.47	1.51	1.28
Evenness index	0.95	0.95	0.96	0.92	0.97	0.94	0.98	0.98	0.97	0.97	0.96	0.97	0.94	0.95	0.98	0.96	0.97	0.95	0.93	0.97	0.99	0.95	0.96	0.94	0.92
Fisher’s Alpha	5.06	5.10	3.60	3.96	5.57	4.61	4.62	5.82	5.65	6.18	6.59	6.49	5.0	4.7	7.49	14.58	9.00	8.08	7.27	7.96	5.72	5.02	8.81	3.88	5.33
Continuous forests																									
Species richness	10	9	7	7	10	9	8	10	10	11	10	10	9	9	15	25	14	11	10	15	8	9	15	7	9
Species number	54	44	42	38	54	59	29	44	38	43	26	35	132	132	41	58	16	19	18	64	20	33	42	26	16
Shannon-Weiner index	2.09	2.01	1.73	1.65	2.12	1.98	1.91	2.16	2.13	2.25	2.16	2.13	2.14	2.14	2.59	3.08	2.52	2.23	2.1	2.56	1.96	1.93	2.54	1.69	1.92
Evenness index	0.95	0.96	0.96	0.92	0.96	0.95	0.98	0.98	0.96	0.97	0.96	0.97	0.97	0.97	0.98	0.96	0.98	0.96	0.95	0.97	0.99	0.92	0.96	0.94	0.95
Fisher's Alpha	6.35	5.47	4.04	4.43	6.16	5.08	5.47	6.32	6.12	6.69	7.07	6.47	1.91	1.91	8.51	15.5	11.6	8.6	7.97	8.3	6.15	5.93	9.24	4.95	7.47

Alpha (α) and Beta (β) diversity of trees in the fragments

The tree 'α' diversity values within the fragments were higher in the Montane forests fragments than in the other two forests (Table 6.1). The 'β' diversity was highest between Upper and Lower montane forests fragment (0.91), followed by Lower montane and Montane (0.81). The Montane and Upper montane forests fragments had the lowest 'β' diversity value of 0.62. Within Montane forests, 'β' diversity between the largest fragment (72.2 ha) and smallest one (0.1 ha) was 0.50 and in the Upper montane forests 'β' diversity between the largest fragment (72.2 ha) and smallest one (1.02 ha) was 0.80.

Microclimatic factors in continuous forest and forest fragments in three forests type

The effect of forest fragmentation on microenvironmental variables was distinct in all the forests types (Figure 6.1). Air and soil temperature, light intensity decreased with increasing distance towards the forest interior, while relative humidity showed an increasing trend. These environmental variables were also influenced by forests type. For instance in Lower montane forests the difference in environmental variables between continuous forest and its fragment was minimal, whereas in Montane forests light intensity, air temperature and relative humidity showed clear difference between the continuous forest and forest fragments. In the Upper montane forests, the difference in microenvironmental variables was minimum, however, relative humidity showed an increasing trend.

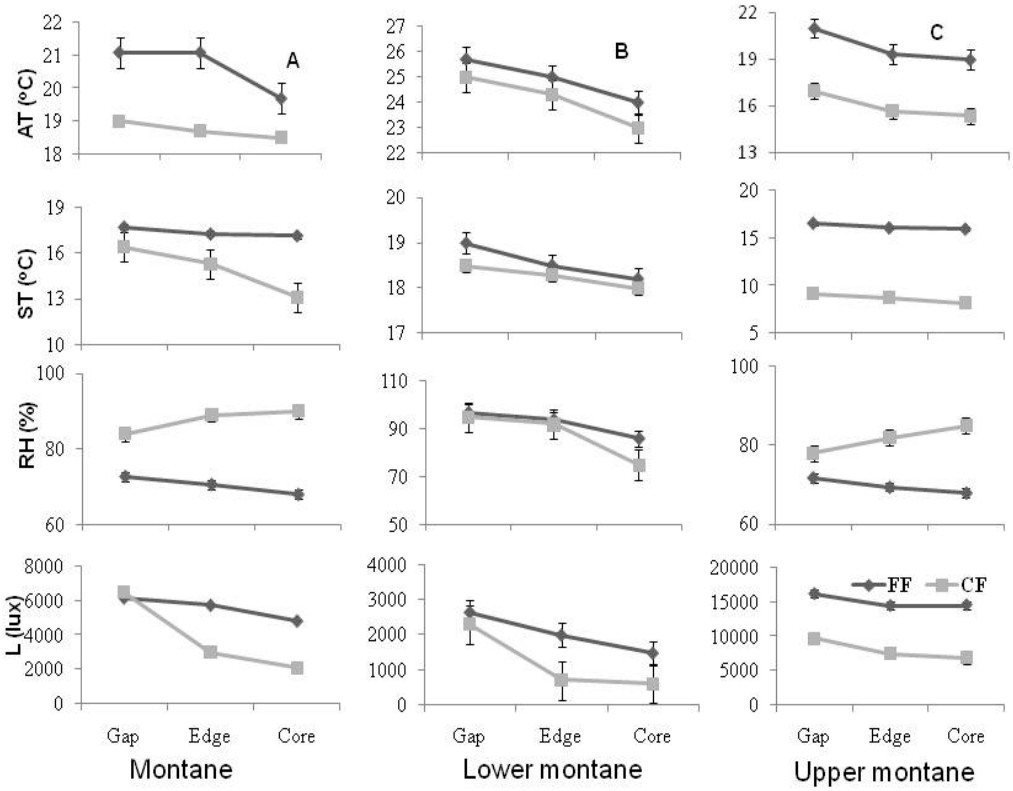


Figure 6.1. Mean of air temperature (AT), soil temperature (ST), relative humidity (RH) and light (L) and their standard errors in the gap, edge, and core area in forest fragments (FF) and continuous forests (CF) in Montane (A), Lower montane (B), and Upper montane (C) forests in KBR.

Microenvironmental factors related to tree abundance in 25 fragments

The overall pattern of tree species diversity across the forest fragments has been depicted through CCA. The first two canonical axes explained 10.3% and 6.1% of the total variance and were significant at 0.001 levels. AT, ST, RH are related to Axis 2 and were important factors determining the tree species distribution across the fragments (Figure 6.2), while light related to Axis 1 was important in the subsequent forest fragments. The ordination plot shows the relative position of the tree species along the line of environmental vectors depicting species response to the environmental variables in respective fragments (Figure 6.2).

Inferred ranking of the species along an environmental variable are presented in the rank biplot (Figure 6.3). The gradation of tree abundance with respect to environmental factors is also clearly presented. The position of the species viz., *Ficus nemoralis*, *Viburnum coriaceum*, *Engelhardtia spicata*, *Castanopsis indica* etc., at the

extreme high soil moisture content end indicated the relative level of responses of various species to soil moisture. Similarly the position of *Engelhardtia colebrookianum*, *Castanopsis indica*, *Rhus semialata* etc., at extreme high end with respect to AT, *Viburnum coriaceum* , *Castanopsis indica* etc., with ST and *Acer campbellii*, *Symplocos ramosissima*, *Quercus lamellosa* etc., at extreme high end with respect to light also explains the microenvironment preferences of the species.

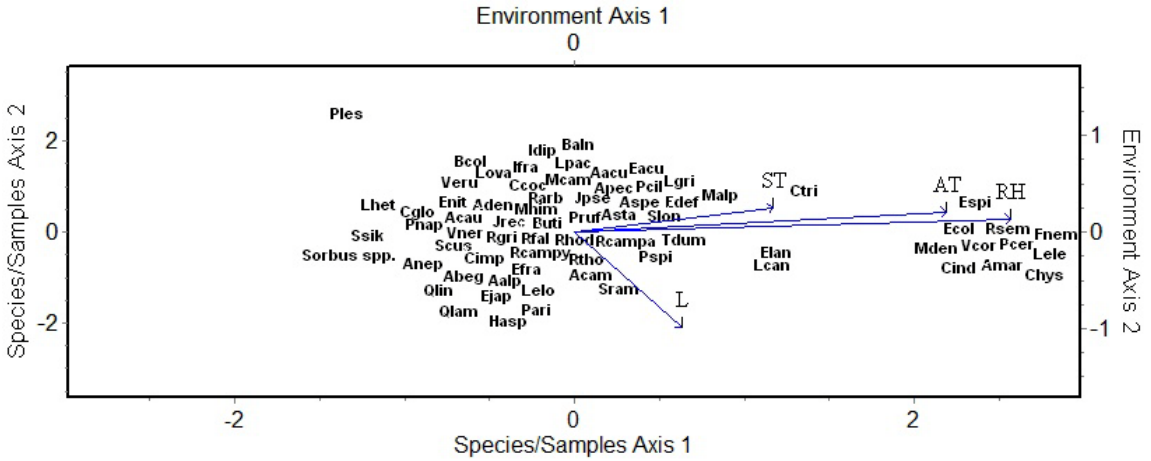


Figure 6.2. CCA ordination diagram using abundance data of 71 tree species from 25 forest fragments and microenvironmental variables across the forest types in KBR. In the ordination diagrams, species names are abbreviated using four letters from the scientific name of each species by combining one initial letter for the genus and three initial letters for the specific epithet) (full name of species is given in Annexure 3).

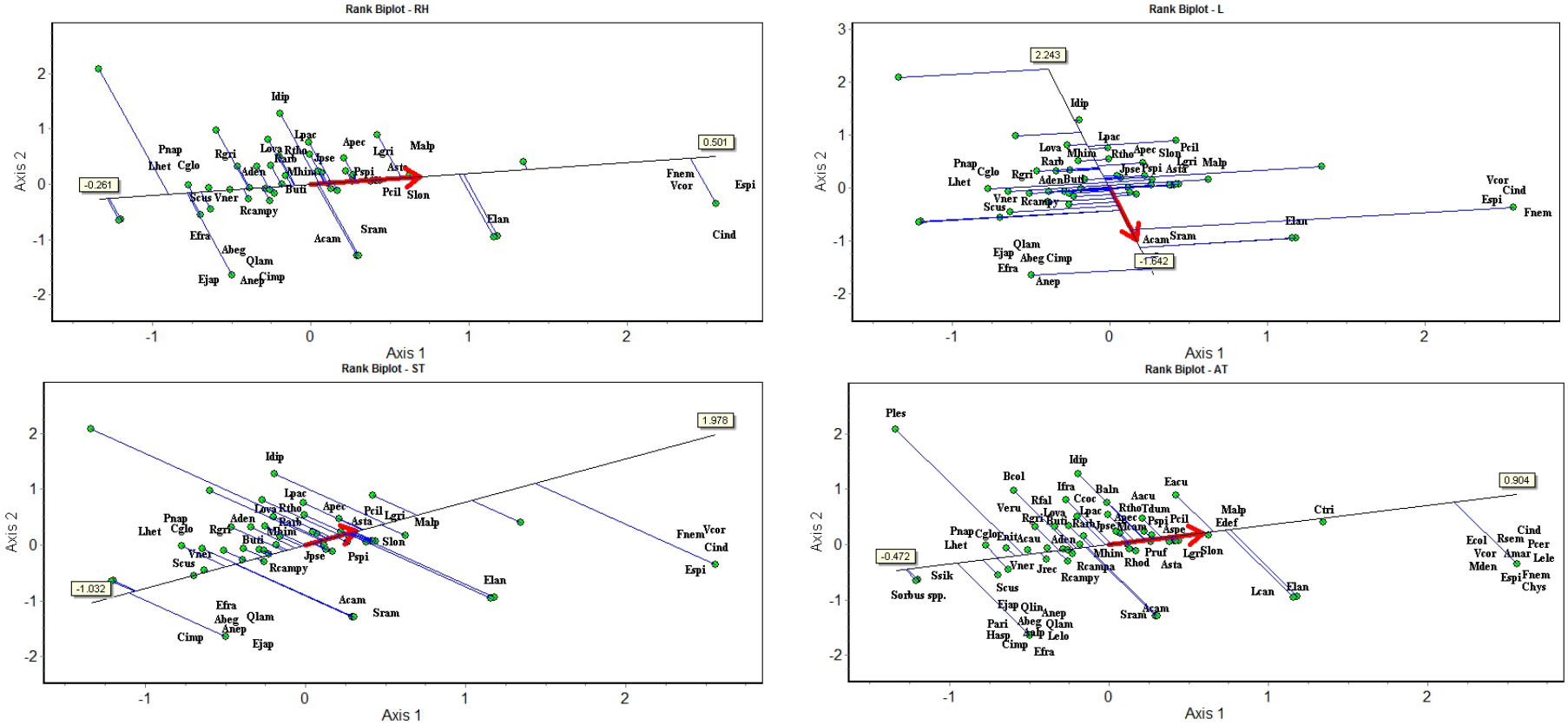


Figure 6.3. Rank biplot created through CCA. In this plot tree species are arranged along the vector representing the strength of microenvironmental variables. Direction of vector indicates the strength of similarity.

Classification of tree communities

Seventy one tree species from 25 fragments were classified into eight specific tree communities (Figure 6.4) shown in the dendrogram along with indicator species that characterise them. The first level of TWINSPAN classification separated the 25 fragments into two main groups, 1 and 2, containing 24 fragments (negative dichotomy) and one fragments (positive dichotomy), respectively. The indicator species at the first level of division was *Albizia margianta.* At the first level of division, group 2 was classified into one distinct tree community (C8) with tree species *Abies densa - Viburnum nervosum.*

The second level of TWINSPAN division of group 1 classified the 24 fragments into two groups, group 3 containing 4 fragments (C1, on the negative dichotomy) and group 4 containing 20 fragments (on the positive dichotomy). *Abies spectabilis* was the indicator species for group 3 and 4.

The third level of TWINSPAN division separated 20 fragments from group 4 into group 5, and 6. In group 4, *Abies densa, Lithocarpus pachyphylla* and *Magnolia campbelli* were the indicator species. Group 5 with *Betula utilis* as indicator species was again grouped into group 7 and 8. Group 7 form a separate community (C2) with FF18-FF20. Group 8 was further divided into 11 and 12 with C3, C4 and C5 communities with total 10 fragments (Figure 6.4). *Maddenia himalaica* and *Buddleia colvilei* were the indicator species in the subsequent step of classifications.

Similarly, the fourth level of TWINSPAN division separated group 6 into group 9 and group 10, with C6, and C7 communities, with seven fragments. *Acer pectinatum* was the indicator species for group 9 and group 10.

Tree species preferring medium size fragments (2-7 ha) were *Abies densa, Rhododendron arboreum, R. falconeri, Abies densa* etc., and were mostly in Upper montane forests. Tree species like *Castanopsis hystrix, Alnus nepalensis, Buddleia colvilei, Ficus semicordata, Eurya acuminata* were found in medium size fragments in Lower montane forests. *Rhododendron hodgsonii, R. thomsonii,* was mostly found in

smaller size fragments in Upper montane forests. Larger fragments in Upper montane forests contained all the species from smaller and medium size fragments.

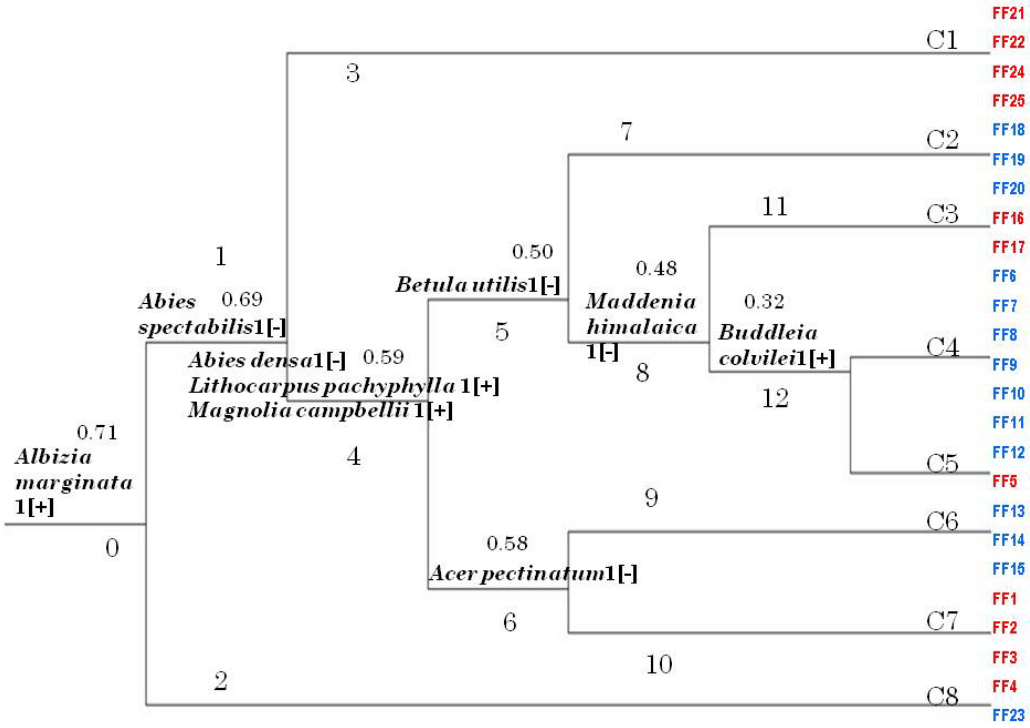


Figure 6.4. Classification of fragments based on tree species composition using TWINSPAN. C1-C8 represents classification of 25 fragments into eight tree community types. The numbers 0 to12 are code names of groups at each level of division and the decimal figures represent the eigen values. Indicator species are also shown: [-] characterises the upper and [+] lower group.

DECORANA shows the spatial distribution of tree species in 25 fragments (Figure 6.5). Species at the negative end of the axis 1 were *Picea spinulosa*, *Populus ciliata*, *Prunus rufa*, *Rhododendron campanulatum*, *R. campylocarpum*, *Salix longiflora*, *S. sikkimensis*, *Sorbus* spp., and *Viburnum nervosum*, all of them being more abundant in the Upper montane forests fragments. On the other hand most of the species at the positive end of the axis 1 such as *Lithocarpus pachyphylla*, *Quercus lamellosa*, *Magnolia campbellii*, *Polygala arillata*, *Alnus nepalensis*, *Rhododendron arboreum*, *Eurya japoniaca* were from Lower montane and Montane forests fragments. It may be concluded that distribution of the species in the ordination space was according to both the tree community types and their area of occurrence in forest fragments in three Montane forests. The eigenvalue of 0.87 and 0.63 (Table 6.2) show that there is a good

dispersion of tree species along the first and second ordination axes. Most of the DECORANA clusters matched with the TWINSPAN classifications, indicating that classification and ordination of tree species data were complementary to each other.

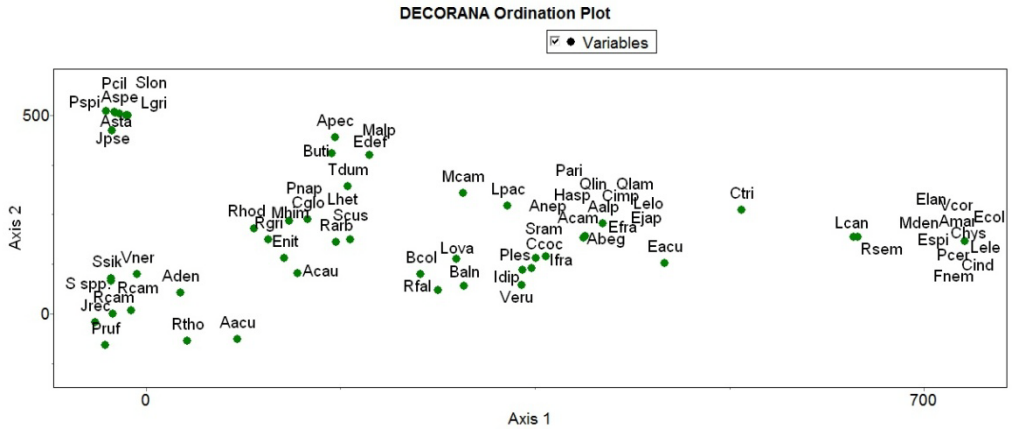


Figure 6.5. DECORANA ordination for depicting the spatial distribution of tree species in 25 fragments.

Table 6.2. Summary of the DECORANA ordination of tree species across the 25 fragments in KBR.

Axes	**1**	**2**	**3**	**4**	**Total inertia**
Eigenvalues	0.87	0.63	0.17	0.09	1.13
Length of the gradient	5.09	3.74	2.19	1.41	
Cumulative percentage variance of species data	20.4	38.5	49.1	58.8	
Sum of all unconstrained eigenvalues					1.13

6.4. Discussion

Fahrig (2003), based on the review of 17 empirical data from small-scale experimental studies to continental-scale analyses pointed out that the effects of fragmentation on diversity were ambiguous and as likely to be positive as negative. Similar results were found for tree communities in Atlantic tropical forest in Brazil (Metzger 1997), where tree diversity of the forest fragments appeared to be similar among patches of different sizes. Similarly, tree species diversity in the highlands of Chiapas, Mexico, is not related to patch size or to any other spatial attribute (Ochoa-Gaona et al. 2004). In the Atlantic tropical forests, forest connectivity and the complexity of the matrix may be more important than fragment area and isolation in explaining variation in tree species richness (Metzger 1997) and functional group richness (Metzger 2000). However, in a study conducted in the montane Atlantic forests of south-eastern Brazil, fragment size was

found to be the major determinant of changes in woody plant composition and guild structure (Tabarelli et al. 1999).

In KBR however a positive tree species-area relationship indicated that species abundance was a function of fragment area, highlighting the importance of area as one of the most important determinants of species richness in fragmented habitats. Positive species- area relationship is attributed to larger fragments containing larger samples and effectively more species. It was also observed that the edges had a positive effect on tree diversity at the plot level. The 'α' diversity for trees in all the forest fragments was lower than that of adjacent continuous forests clearly indicating the role of forest fragmentation in reducing the species diversity. In KBR, long edge in the fragments of larger sizes is due to the presence of the forest asymmetrical projections, a frequent situation along ravines, hills lopes and mountain terrain. As fragmentation increases, such asymmetrical projections disappear or become separated producing less complex shape (nearer to circularity or low shape index value). Fragments with equivalent size may contain contrasting amount of forest species in interior or edge resulting from their shape complexity. Therefore, there is reason to expect the species-area relationship in fragments to be simple.

Fragment isolation is more accurately viewed as a measure of the lack of habitat in the landscape surrounding the fragment. More isolated fragment will have fewer habitats in the landscape surrounding it. Isolation was not a significant variable in determining species richness in KBR forest fragments. Study areas in KBR is characterised by complex topography, which include variation of slope, angle and aspect. Relatively, close fragments may occupy different topographic positions, and they may affect the dispersal success of plant propagules between them. In addition to distance, slopes and aspects as well as hilltops in them may limit potential exchange of organisms. Plant species, especially those mainly dispersed by gravity will be most affected.

There are changes in microclimatic conditions in the fragments, and modified environmental conditions may not be appropriate for many species in continuous forests (Hobbs & Yates 2003; Laurance et al. 2002). Environmental factors are by and large influenced by the vegetation. Such influences vary among the factors; temperature for example steadily decreases towards fragment centre. Light intensity on the other hand exhibited a steep diminution as depicted in the curve. Such forest edge effect is having serious implications on the forest ecosystem, either directly or indirectly. Direct influence results in the alteration of habitat quality (in terms of micro-climatic factors). This may bring about a gradual change in species composition; in which case, the habitat generalists may overcome the habitat specific species thriving in the interior environment. For example, the occurrence of *Castanopsis tribuloides*, *Magnolia campbellii* and *Abies densa* are very rare in forest margin but mature trees growing near the forests edge can persist.

Relatively lower values of the first two constrained CCA axes apparently indicate that the environmental variables are not sufficient to predict the distribution of tree species extracted by CCA, but they do predict a substantial part of remaining variations. Therefore the clustering of the tree species in different fragments differs with respect to environmental preferences. Clustering of FF6, FF7, FF8, FF9, FF10, FF11 and FF12 in TWINSPAN and species in DECORANA was due to environmental preferences as depicted through CCA.

TWINSPAN analyse the distribution of tree species and it separated into eight communities in all the forest fragments in KBR. A somewhat similar approach was used by Hussian et al. (2008) in Kumaon Himalaya for studying species composition and community structure of forest stands.

By classifying fragment into community with different species composition, one can ensure that all the major communities are represented by fragments. The TWINSPAN

classification of tree species from 25 forest fragments in KBR indicated the presence of eight tree communities. The distinction in the composition of the communities was not marked except for C-8 from Lower montane forests. C-8 community is characterised by lower altitudinal tree species with high diversity and evenness indices. Other tree communities from Montane and Upper montane forests fragments have high tree species affinity and abundance of specific tree species as evident from the indicator species analysis. Tree species like *Castanopsis hystrix*, *Alnus nepalensis, Buddleia colvilei*, *Ficus semicordata*, *Eurya acuminata*, were found in medium size fragments in the Lower montane forests. *Rhododendron hodgsonii, R. thomsonii,* was mostly found in smaller size fragments in the Upper montane forests. Larger fragments in Upper montane forests contained all the species from smaller and medium size fragments.

In KBR, the influence of forest isolation on tree species diversity at the fragment level was not as strong as those of environmental variables. It is clear from the above discussion that each forest fragment was separated from the other with a distinct community. Tree species abundance within each community in different fragments was influenced by microenvironmental variables. Thus a significant change in light, soil temperature and moisture regime along the fragments size gradient played an important role in influencing the composition and abundance of tree species thereby separating into 8-communities across the 25 fragments.

Annexure 3. Botanical names of tree species with code, encountered in the 25 forest fragments in KBR, Sikkim.

Code	Scientific name
Aden	*Abies densa* Griffith.
Aspe	*Abies spectabilis* D. Don (Spach)Acer pectinatum
Apec	*Acer pectinatum* Wall.acuminatum
Aacu	*Acer acuminatum* Wall. Ex D. Don
Acam	*Acer campbellii* Hook .f. & Thoms. Ex Hierncaudatum
Acau	*Acer caudatum* Wall.
Asta	*Acer stachyophyllum* Hiern
Aalp	*Alangium alpinum* (C. B. Clarke) Sm. & Cave
Abeg	*Alangium begoniaefolium* Baill.
Amar	*Albizia marginata* (Lam.) Merr.
Anep	*Alnus nepalensis* D. Don
Bajn	*Betula alnoides* Buch.-Ham ex D. Don
Buti	*Betula utilis* D. Don
Bcol	*Buddleia colvilei* Hook. F. Thom.
Cglo	*Casearia glomerata* Roxb.
Chys	*Castanopsis hystrix* Miq.
Cind	*Castanopsis indica* A. de Candolle
Ctri	*Castanopsis tribuloides* S. DC.
Cimp	*Cinnamomun impressinervium* Meisn.
Ccoc	*Colquhounia coccinea* Wall.
Elan	*Elaeocarpus lanceaefolius* Roxb.
Ecol	*Engelhardtia colebrookianum* Lindl. ex Wall
Espi	*Engelhardtia spicata* Blume
Edef	*Enkianthus deflexus* Schneider
Efra	*Evodia fraxinifolia* Hk. f.
Eacu	*Eurya acuminata* DC.
Ejap	*Eurya japonica* Thunb.
Enit	*Eurya nitida* Korthals
Fnem	*Ficus nemoralis* King
Hasp	*Hydrangea aspera* Buch. – Ham. ex D. Don
Idip	*Ilex dipyrena* Wall.
Ifra	*Ilex fragilis* Hk. f
Jpse	*Juniperus pseudosabina* Fisch. Et C. A. Mey
Jrec	*Juniperus recurva* Buch.-Ham. ex D. Don
Lgri	*Larix griffithii* Hk. f.
Lcan	*Leucosceptrum canum* Sm.
Lhet	*Lindera heterophylla* Meisn.
Lele	*Lithocarpus elegans* Soepadmo
Lpac	*Lithocarpus pachyphylla* Rehder
Lelo	*Litsaea elongata* (Wall. ex Nees.) Benth. et Hk. f.
Lova	*Lyonia ovalifolia* (Wall.) Drude
Mden	*Macaranga denticulata* (Blume) Mueller
Mhim	*Maddenia himalaica* Hk. F. & Thom
Mcam	*Magnolia campbellii* Hk. f. & Thom

Malp	*Merrilliopanax alpinus* (Clarke) Shang
Ples	*Pentapanax leschenaultii* (DC.) Seem
Pspi	*Picea spinulosa* (Griff.) A. Henry
Pari	*Polygala arillata* Buch.-Ham. Ex D. Don
Pcil	*Populus ciliata* Wall. Ex Royle
Pcer	*Prunus cerasoides* D. Don
Pnap	*Prunus napaulensis* (Ser.) Steud.
Pruf	*Prunus rufa* Hk. f.
Qlam	*Quercus lamellosa* Smith
Qlin	*Quercus lineata* Blume
Rarb	*Rhododendron arboreum* Smith
Rcam	*Rhododendron campanulatum* D. Don
Rcam	*Rhododendron campylocarpum* Hk. f.
Rfal	*Rhododendron falconeri* Hk. f.
Rgri	*Rhododendron griffithianum* Wight
Rhod	*Rhododendron hodgsonii* Hk. f.
Rtho	*Rhododendron thomsonii* Hk. f.
Rsem	*Rhus semialata* Murray
Slon	*Salix longifolia* Muhl.
Ssik	*Salix sikkimensis* Anderson
Scus	*Sorbus cuspidata* (Spach.) Hedl.
Sspp.	*Sorbus* spp.
Ttil	*Toricellia tiliifolia* DC.
Tdum	*Tsuga dumosa* (D. Don) Eichler
Vcor	*Viburnum coriaceum* Blume
Veru	*Viburnum erubescens* Wall. Ex DC.
Vner	*Viburnum nervosum* D. Don

Chapter 7
Regeneration of selected trees and lianas

7.1 Introduction

Trees and lianas are two dominant life-forms in the tropical forests. Compared to trees, abundance and diversity of lianas are greater in disturbed habitats such as tree fall gaps than the surrounding undisturbed forests (Putz 1984; Schnitzer & Carson 2000). Lianas contribute to forest regeneration by regenerating through seeds and vegetative propagation. Lianas compete directly with trees and therefore can alter the forest composition. Lianas require greater light level than the tree species. However, our understanding of liana regeneration is poor.

In natural forests, regeneration of trees and lianas takes place either through genets (seeds) or ramets (vegetative means like sprouting/coppicing or root suckers). The status of woody species regeneration in a forest ecosystem can be inferred from their population structure (Marks 1974; Vablen et al.1979; Pritts & Hancock 1983; Saxena et al. 1984b; Khan et al. 1987; Bhuyan et al. 2003). Successful regeneration is predicted by the presence of adequate number of seedling, saplings and young individuals in a given population (Saxena & Singh 1984). Besides seedling regeneration, coppicing is generally the primary regeneration mechanism, where stem and roots remain in place (Ewel 1977; Murphy & Lugo 1986; Murphy et al. 1995). In natural tropical forests where large-scale disturbance occurs as a result of clearing, burning and extensive storm damage, regeneration from stem coppice is very important (Byer & Weaver 1977; Putz & Brokaw 1989; Bellingham et al. 1994). Gilbert et al. (2006) reported that lianas and trees were not different in their regeneration requirements during seedling and sapling stages.

The forests in Khangchendzonga Biosphere Reserve experience varying intensity of natural and anthropogenic disturbances. It is important to have knowledge on regeneration ecology of tree and liana species in the BR for effective management of forests as well as the species. The status of regeneration with special emphasis on taxonomically as well as ethnobotanically important species in these forests would help in better managing these species. An assessment of regenerative capacity of different species in these forests would help predicting the forest regeneration trends as well as the future structure of the forests. Two tree species viz. *Toricellia tiliifolia* DC., and *Evodia fraxinifolia* Hook. f. and two liana species viz. *Holboellia latifolia* Wall., and *Entada phaseoloides* (L.) Merr., were selected for their detailed regeneration study. *T. tiliifolia* is a monotypic genera, thus important from conservation point of view, while *E. fraxinifolia* is widely used for medicine by the local communities. The liana species *H. latifolia* is a taxonomically important plant being a primitive taxon and is also an edible fruit yielding plant. On the other hand, *E. phaseoloides* is an important plant used in ethnomedicine. The population sizes of all these four species are relatively small and an investigation into their regeneration issues may address their perpetuation in the forests.

7.2 Methods

Species selected for study

Evodia fraxinifolia, a medium sized tree belonging to family Rutaceae, is locally used for food and as a medicine for dysentery. *Toricellia tiliifolia*, a monotypic genera belonging to family Toricelliaceae is a small tree. It is mostly used as flag pole by the local people. The liana species, *Holboellia latifolia* belongs to family Lardizabalaceae and is found in the montane forests of BR. *Entada phaseoloides*, a large liana, belongs to family Fabaceae and is used for extracting poison by soaking in water. It is also taken after roasting by the local

people. It is also used for washing the hair. The distribution of these selected four species varied among the forest types in the BR (Table 7.1).

Table 7.1. Distribution of the four selected species in the KBR and their importance.

Life form	Species	Distribution		Importance
		Forest types	Elevation (m)	
Trees	*E. fraxinifolia*	Lower montane-montane	1600-2500	Ethnobotanical/medicinal
	T. tiliifolia	Lower montane	1450-2000	Taxonomic
Lianas	*H. latifolia*	Lower-upper montane	1800-3100	Ethnobotanical
	E. phaseoloides	Lower montane	1200-1700	Ethnobotanical/medicinal

Phenology

Ten individuals of each species were selected in the respective forest types during February, 2007 for studying the phenological events such as time of active shoot growth, flowering, fruiting and leaf fall/ leaf flush. The quantitative data such as flower and fruit production and the data on the vegetative and reproductive behaviour of the species were collected through periodic observations.

Regeneration status of tree and liana species

Regeneration status of the selected species was studied by quantification of seedling, sapling and coppice (Khan et al. 1986 & Uma Shankar 2001) populations.

Population structure of the selected species

Population structure of the selected tree species was studied by classifying the individuals into four size classes viz. seedling, (0-5 cm CBH), sapling (6-10 cm CBH), young (11-60 cm CBH), and adult (> 60 cm CBH). In case of liana, seedlings (< 0.2 DBH and < 1.3 m in height) and adult (> 0.2 cm DBH and > 1.3 m in height) were taken for population study.

Effect of stump size on sprouting intensity

Coppice regeneration was important in *Toricellia tiliifolia* and *Holboellia latifolia.* To study the effect of stump size on sprouting, 40 cut stumps for each of the two species were marked with paint. All the previously sprouted shoots were removed in February, 2007 for

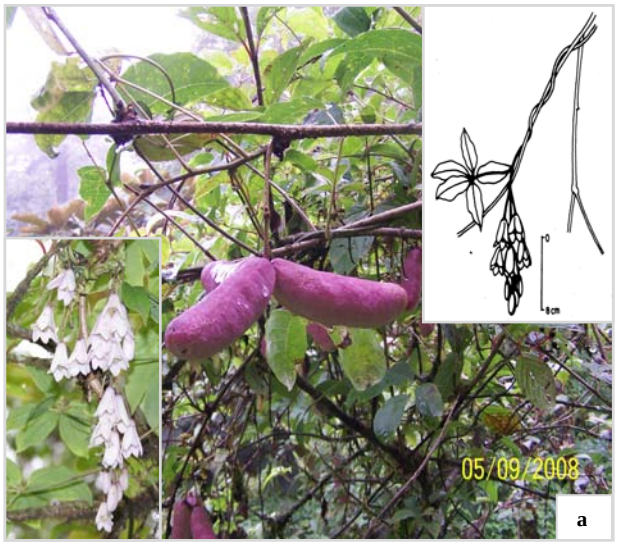
uniformity in sampling. The number of sprouts in each stump was noted after one year in February, 2008. The stumps were categorized into different girth classes to assess the relationship between stump girth size and number of sprouts. The corresponding total number of sprouts found within the girth classes was then analysed using one-way ANOVA.

Estimation of flower and fruit production

For estimation of flower and fruit production, 5 fruiting trees in each DBH class of *T. tiliifolia*, *E. fraxinifolia* and *H. latifolia* were selected and tagged in the respective forests of occurrence. However, flower and fruit production in *E. phaseoloides* could not be studied due to its small population size and difficulty in accessing the species at top canopy layer. The numbers of flowers and fruits produced by these marked individuals were estimated for three consecutive years (2006-2008). Flower bud count was taken as a criterion to estimate the flower production. The fruits were counted on the plant itself just before maturity. Since dispersal of fruits starts during maturation on the plant itself, fruit production estimates made on the initial stage of maturation represented the total fruit production including those dispersed during the maturation phase.

The flower and fruit production was estimated following Barik et al. (1996). Total flower/fruit production = total number of branches x mean number of sub-branches per main branch x mean number of inflorescences per sub-branch x mean number of flower buds/fruits per inflorescence.

For each tree/liana, mean number of inflorescence per branch was calculated from a sample of 10 branches and mean number of flower buds/fruits per inflorescence was calculated from a sample of 50 inflorescences. ANOVA was performed to test the variation in fruit and flower production due to DBH class, forest type and year. Flower abortion was estimated by subtracting the values of fruit production from those of flower production.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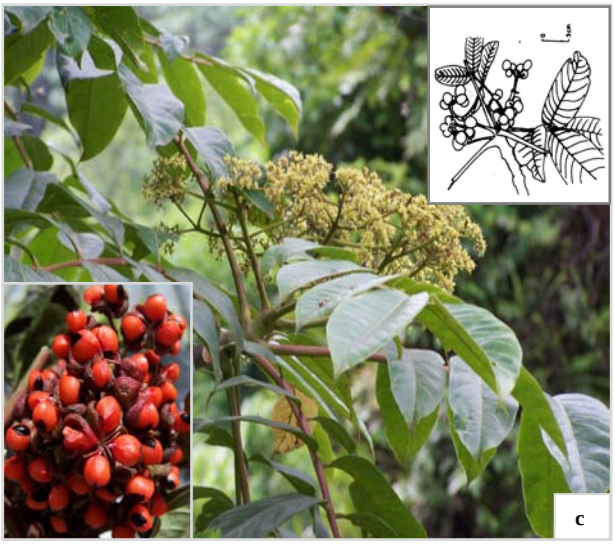
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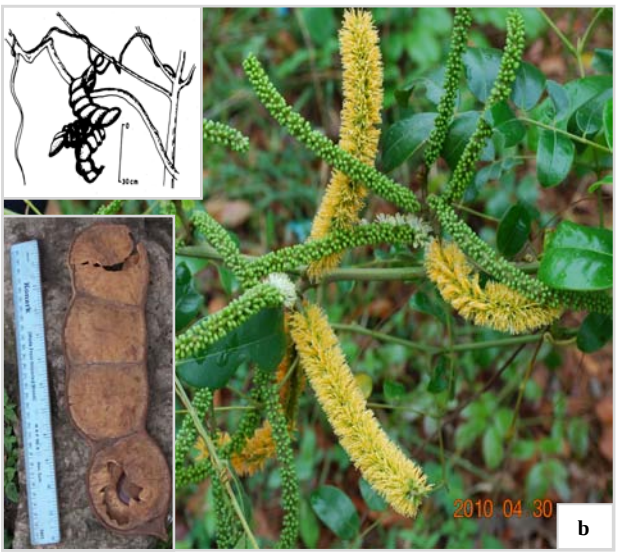
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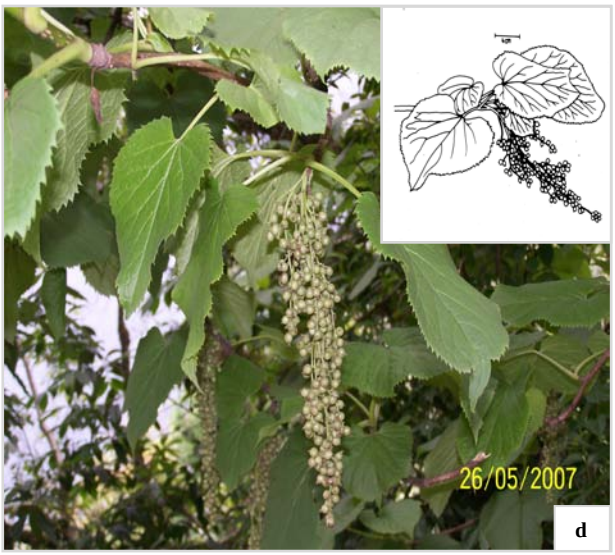
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Plate 7.1. Tree and liana species selected for regeneration studies: (a) *Holboellia latifolia* (b) *Entada phaseoloides*; (c) *Evodia fraxinifolia* and (d) *Toricellia tiliifolia*

Assessment of soil seed bank of selected species

To assess the fate of the remaining seeds after dispersal/disappearance of fruits during the post-fruit fall period, seed banks were estimated beneath the five marked trees in the respective forest stands just before the next fruit rain. As the seeds of the selected species usually remain either above or below the litter layer, they could be counted easily. To study the seed bank, five 1 m x 1 m quadrats were randomly laid in each concentric circle around the parent tree with radius increasing by a factor of 5 upto 45 m. The seeds were collected from each quadrat and total seed bank was computed for each marked tree during 2007 and 2008.

Seed viability

Seed viability was determined using the tetrazolium (TTZ) assay (International Seed Testing Association 1993) and/or germination trials. For the tetrazolium assay the

embryo-cum-endosperm fractions of 100 seeds per species were extracted from the associated fruit/seed structures, and placed in a 1% aqueous solution of 2,3,5-triphenyl tetrazolium chloride for 48 hours at 25 °C. Embryos were scored as viable if stained red or pink. Germination trial was, however, used for *T. tiliifolia* due to their hard seed coat; and was conducted using samples of 100 seeds placed in a germination tray with 3-4 cm thick layer of garden soil moistened regularly with tap water.

Seed germination

In situ seed germination test was undertaken during August-October, 2007 for *Toricellia tiliifolia* and *Evodia fraxinifolia*, and October-December, 2007 for *Holboellia latifolia* and *Entada phaseoloides.* The *in situ* experiment was conducted to study the effect of litter depth and forest type on seed germination. Seeds of selected species were collected from the parent plant and un-damaged seeds were separated from the damaged ones by floatation method. 100 seeds of heavy weight category were selected and sown in each treatment i.e. above

litter, under litter (2-4 cm depth) and in a cleared forest floor each with a bed size of 2 m x 2 m. The beds were covered with nylon mesh of size 2 mm to avoid predation of the seeds but to allow light and air to pass through it. Seeds were considered germinated with the emergence of 2 mm radicle and the observations were continued till complete cessations of seed germination.

Seedling recruitment and mortality

The study was conducted for *H. latifolia*, *Entada phaseoloides* and *E. fraxinifolia*. For studying mortality pattern, seedlings were marked in the two-leaved stage after germination. Seedling recruitment for the selected species was studied for two consecutive years and was studied by tagging the seedlings with aluminium labels. The first year's tagged seedlings were monitored over a period of two year. Plants that were

damaged and re-sprouted were not considered as dead. The mortality rate of the seedling populations was calculated at seasonal intervals following Condit et al. (1996):

$$\frac{ln\,(N_0) - ln\,(N_t)}{t}$$

where N_0 is the number of initial seedlings, N_t is the number of seedlings remaining alive at time t (month) and ln (n) is the natural logarithm of N. The mortality rates were calculated for the periods March-June 2007, June-September 2007, September-December 2007, December-March 2008, March-June 2008, June-September 2008 and September-December 2008.

Plant life history trait

The life history traits were evaluated by examining the following parameters: (1) area of occupation and habitat characteristics –the density of seedlings in open areas or periphery of forests, and in understory habitats were compared; (2) Seed weight–Mean seed weight was determined by weighing 50 randomly collected seeds of each species; species with mean seed weight of $\geq$ 0.5 mg were treated as large seeded species and others were small seeded species; (3) Soil seed bank – five random quadrats of 1 x 1 m^2 were laid around the parent plant after one month of the fruiting period, and the number of seeds in the top 0-10 cm layer soil in each quadrat was counted; species with $\geq$ 100 seeds per m^2 were treated as species having large seed bank and the rests were treated as species with small seed bank; (4) Leaf mass/ unit area – leaf mass was estimated from the oven dry weight of randomly collected 50 leaves of each adult plant and was divided by the respective leaf area of each leaf. The leaf area was determined with the help of a portable leaf area meter (LICOR); (5) Disturbance related colonization – the disturbance-colonization relationship was analyzed based on the frequency of occurrence of adult plants in periphery or forest edges, and under storey habitats; (6) Dispersal mechanism – the modes of dispersal of

species were evaluated by considering the fruit and seed characteristics and through field observations; and (7) Climbing mechanism – the climbing mechanisms were broadly classified into two categories viz., tendril climbing and others including twining, root climbing and scrambling. Species were classified into pioneer and non pioneer guilds based on the above life history traits adapted from Turner (2001). The tendril climbing was considered as a pioneer syndrome and all the other climbing mechanisms were treated as non-pioneer syndrome.

Effect of forest fragmentation on seedling density

Plants of 5–30 cm tall and lianas of 10–100 cm tall were considered seedlings. The smaller seedlings than the above mentioned dimension were not included as they were very difficult to identify and to distinguish them from other seedling populations.To distinguish a liana seedling from a vegetative offshoot or a sprout from a broken liana, the connectivity and diameter of root or shoot were examined by excavating soil in a circular ring around the

plant. In each fragment, $1m^2$ seedling sampling plots were placed along the periphery and interior at random within each fragment varying in sizes. The number of sampling plots for both periphery and interior were equal but number of sampling plots varied with sizes of the forest fragments.

7.3 Results

Ecological description of the selected species

The ecological characteristics of the four study species such as shade tolerance, successional status, fruiting period, fruit type, colour in maturity, dispersal agent, seed weight and germination types are presented in Table 7.2.

Table 7.2. Ecological description of the selected four species.

Name of species	*T. tiliifolia*	*E. fraxinifolia*	*H. latifolia*	*E. phaseoloides*
Shade tolerance	Low	Partial shade	Low	Low
Successional status	Early	Intermediate	Early	Intermediate/late

Fruiting period	March-April	Sept-Oct	Sept-Nov	July-Oct
Fruit type	Berry	Berry	Berry	Legumes
Colour in maturity	Dark-purple	Black	Black	Brown
Dispersal agent(s)	Water	Animals/humans	Animals/humans	Birds/Water
Mean seed wt.(g) ± SD	0.05±0.04	0.01±0.01	0.10±0.02	14.10±3.54
Germination type	Epigeal	Epigeal	Epigeal	Hypogeal
Germination (%)	3±0.88	43±8.80	75± 2.88	80±5.77

Phenology of the study species

Various phenological events such as vegetative growth (VG), flower bud formation, flowering, fruit set, leaf shedding, leaf flushing and seed dispersal have been depicted in Figure 7.1. *Evodia fraxinifolia* strictly followed a sequential order of one phenological event followed by the other with least overlapping between any two events. In *T. tiliifolia* active vegetative growth, bud formation and flowering initiated more or less at the same time, particularly during late winter. The leaf shedding and seed dispersal also overlapped with each other. Fruit set to fruit maturity takes longer period in *T. tiliifolia* and coincided with

monsoon season. Leaf flushing was very common in lianas like *H. latifolia* and *E. phaseoloides*. In *H. latifolia* leaf flushing occurred during winter season while in *E. phaseoloides*, it was prominent during the months of April, May and June till early season of monsoon. Flowering, fruit setting and seed dispersal followed more or less similar sequence in both the species of lianas.

Population structure and regeneration status of trees and lianas

Population structure of some of the selected species is depicted in Figure 7.2. *T. tiliifolia* showed a higher number of individuals in the adult stage and lesser number of individuals during seedling and sapling stages in the Lower montane forests. This species was absent in other two forest types. In case of *E. fraxinifolia* density of individuals was more in the seedling and sapling stages in Lower montane and Montane forests, whereas in Upper montane forests lesser individual of this species was encountered. Individuals of *H.*

latifolia were also more during seedling stage in all the forest types. *E. phaseoloides* was characterized as a growing population in Lower montane forests with maximum number of individuals in the seedling stage, and adult stage was less prominent.

Based on the population of selected species during different stages, *T. tiliifolia* had fair regeneration and *E. fraxinifolia* had good regeneration among the tree species. Among the lianas, both the species i.e. *H. latifolia* and *E. phaseoloides* had good regeneration; however the density of *E. phaseoloides* during adult stage was less. Seedling populations in all the forest stands showed marked differences between wet (June-September) and dry (October-March) seasons with more number of tree and liana species in the seedling stage in the wet season. No apparent difference was found in the sapling populations between dry and wet seasons.

Holboellia latifolia

Phenophases / Months	J	F	M	A	M	J	J	A	S	O	N	D
VG		■	■	■								
LF	■										■	■
FBF				■								
F				■	■	■						
FS							■	■	■	■		
SD										■	■	■

Entada phaseoloides

Phenophases / Months	J	F	M	A	M	J	J	A	S	O	N	D
VG			■	■	■							
LF				■	■	■						
FBF			■	■	■							
F					■	■	■					
FS								■	■	■	■	
SD	■	■									■	■

Toricellia tiliifolia

Phenophases / Months	J	F	M	A	M	J	J	A	S	O	N	D
VG											■	■
FBF											■	
F	■											■
FS	■	■	■	■	■	■						
LS							■	■	■	■		
SD									■	■		

Evodia fraxinifolia

Phenophases / Months	J	F	M	A	M	J	J	A	S	O	N	D
VG	■	■	■									
FBF			■	■								
F					■	■						
FS						■	■	■	■			
LS								■	■	■		
SD									■	■	■	■

Figure 7.1. Phenology of *Holobellia latifolia*, *Entada phaseoloides*, *Toricellia tiliifolia*, and *Evodia fraxinifolia*. VG- vegetative growth, FBF- flower bud formation, F- flowering, FS- fruit setting, LS- leaf shedding, LF- leaf flushing, SD- seed dispersal

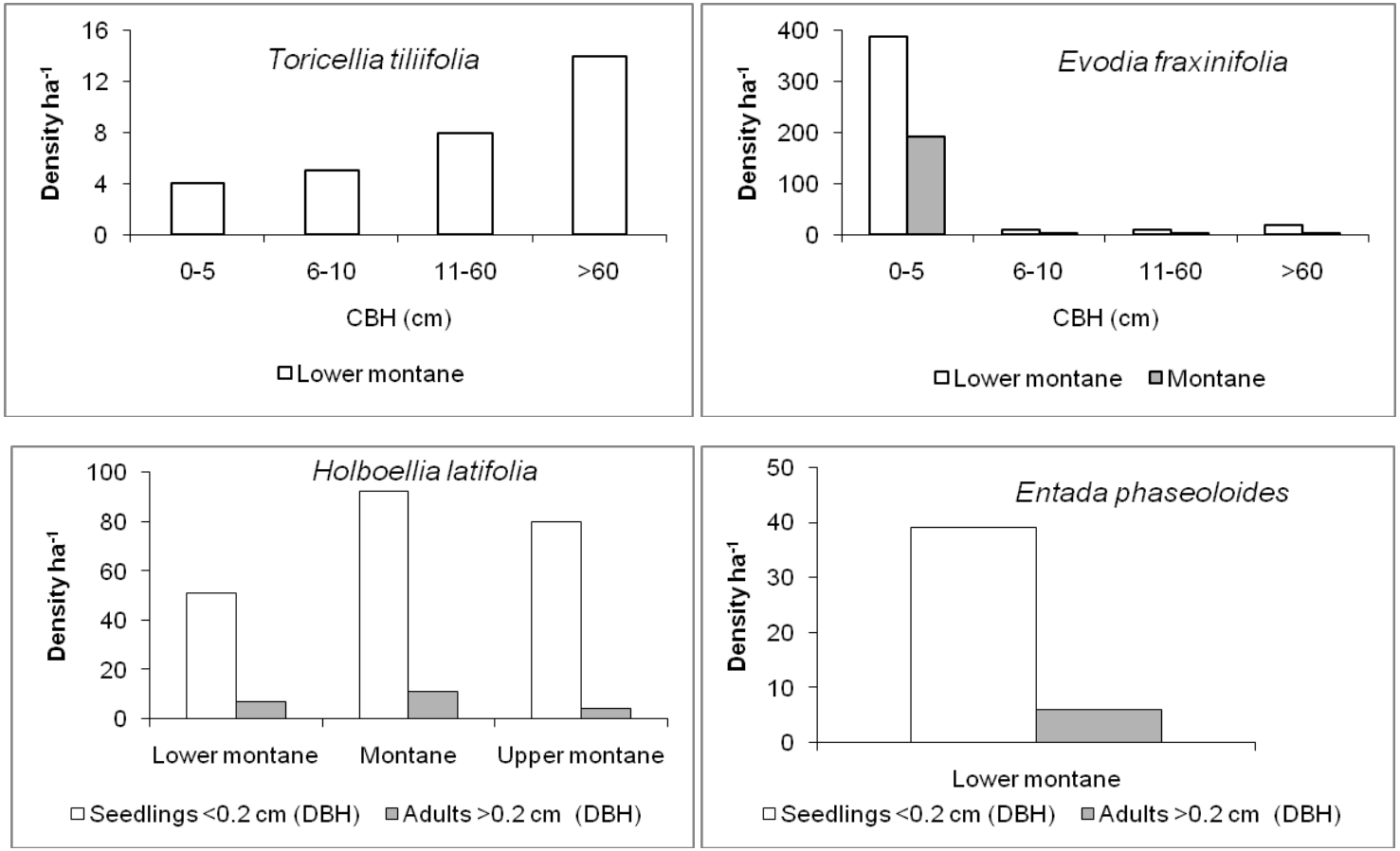

Figure 7.2. Population structure of selected trees (*T. tiliifolia* and *Evodia fraxinifolia*) and lianas (*H. latifolia* and *E. phaseoloides*) in KBR.

Effect of stump size on sprouting of selected species

Coppice regeneration was studied only in *T. tiliifolia* and *H. latifolia*. Both the species coppice profusely.

Table 7.3. ANOVA to test the variation in sprouting intensity due to stump diameter in *Holboellia latifolia* and *Toricellia tillifolia.*

	Stump DBH (cm)	n	Levels	One-way ANOVA		
				F	P	df
Toricellia tiliifolia	5-45	40	4	48.83	0.001	3
Holboellia latifolia	2-25	40	4	55.49	0.001	3

Stump size significantly affected sprouting intensity in *T. tiliifolia*, and *H. latifolia* ($P < 0.001$) (Table 7.3). Average number of sprouts was more in the higher DBH classes of 26-35 cm and 36-45 cm in *T. tiliifolia*, and 14-19 and 20-25 cm in *H. latifolia.* The tree and liana species did not show significant variation in sprouting intensity however liana species tend to initiate sprouting at lower DBH classes than tree species.

Regression models depicting the relationship between stump size and number of sprouts showed significant polynomial relations for two species ($P = 0.001$) as shown in Table 7.4.

Table 7.4. Regression models showing the relationship between stump size and number of sprouts.

Species	Regression model	n	R^2	P- value	Range of 'x'
Toricellia tiliifolia	y = 0.49x + 1.96	40	0.89	0.001	5-45
Holboellia latifolia	y = 0.61x + 1.61	40	0.90	0.001	2-25

'N' - no. of observations, 'y' - no. of sprouts and 'x' - stump size

Flower and fruit production

The number of flower and fruit produced in *Toricellia tiliifolia* was significantly higher ($P < 0.001$) in the year 2008 than in 2006 and 2007 (Table 7.5). The number of aborted flowers was highest in 2007. Flower and fruit production varied significantly across DBH classes ($P < 0.001$).

Mean flower and fruit production was higher in the Lower montane forests stands compared to the Montane forests stand for *Evodia fraxinifolia*. The percentage of flower abortion was higher (11%) in the Montane forests stands than in the Lower montane forests stands. Flower production varied significantly ($P < 0.001$) across DBH, forest types and year. Fruit production varied significantly between DBH and forest types, but flower abortion varied significantly across DBH only.

For liana (*Holboellia latifolia*), both flower and fruit production was higher in the Upper montane forests as compared to Montane and Lower montane forests stands. But flower abortion was highest (83%) in the Lower montane forests and varied significantly across the DBH class ($P < 0.001$). Flower and fruit production varied significantly across the DBH and forest types ($P < 0.001$). Abortion, flower and fruit production did not show significant variation across the years.

Soil seed bank

Size of the soil seed bank for *T. tiliifolia*, *E. fraxinifolia* and *H. latifolia* varied significantly ($P < 0.001$) with girth size (Table 7.6). Soil seed bank across the girth size and year were insignificant for all the three species.

The size of the soil seed bank of *T. tiliifolia* was significantly higher in both the years (2007-2008) in the Lower montane forests (Figure 7.3). The seed bank differed in size between the species. *T. tiliifolia* seed bank size was (190-200) m^{-2}, that for *E. fraxinifolia* (100-200) m^{-2} and for *H. latifolia* it was (90-140) m^{-2}.

Table 7.5. Mean (±SD) of flower production, fruit production and flower abortion in different forests and years. F-ratios show their variances across DBH, stands and years.

Name of species	Forests	2006			2007			2008			F-ratios
		Flower	Fruit	Abortion (%)	Flower	Fruit	Abortion (%)	Flower	Fruit	Abortion (%)	
Toricellia tiliifolia	LM	49945 ±27664	47200 ±26763	6 ±3	59933 ±32531	55213 ±30417	8 ±3	81293 ±10152	75240 ±312	8 ±2	**DBH:** Fl- 9949.1**, Fr- 10038.2**, Ab-7.6* **Year:** Fl- 5595.2**, Fr- 5013. 2**, Ab- 3.8*
Evodia fraxinifolia	LM	88800 ±44417	81193 ±45661	12 ±8	90753 ±46959	82209 ±46924	12 ±6	88020 ±44520	81760 ±45997	10 ±6	**DBH:** Fl- 15926.7**, Fr- 16863.8**, Ab- 58.5**
	M	50527 ±36912	45330 ±32862	11 ±4	50920 ±36908	45303 ±32779	11 ±3	50936 ±37046	45454 ±32910	11 ±3	**Forest:** Fl- 7462.1**, Fr- 7657.2**, Ab- 0.12^{ns} **Year:** Fl- 4.4*, Fr- 0.48^{ns}, Ab- 1.02^{ns}
Holboellia latifolia	LM	2437 ±2060	411 ±346	83 ±1	2488 ±1978	418 ±352	84 ±2	2372 ±1947	416 ±352	83 ±1	**DBH:** Fl- 23212.3**, Fr- 2226.3**, Ab- 24.8** **Forest:** Fl- 5847.5**, Fr- 494.13**, Ab- 22.03** **Year:** Fl- 2.69^{ns}, Fr- 1.94^{ns}, Ab- 0.045^{ns}
	M	2866 ±2045	565 ±431	81 ±3	2871 ±2031	578 ±451	81 ±2	2896 ±2056	585 ±460	81 ±3	
	UM	4161 ±1021	745 ±214	82 ±1	4169 ±995	784 ±207	81 ±4	4316 ±1068	773 ±234	82 ±2	

*P<0.05, **P<0.001, ns - not significant, Fl - flowers, Fr - fruits and Ab - flower abortion

Table 7.6. ANOVA for testing the ariation in seed bank due to tree girth size, forest type and year.

Source of variation	F-ratios		
	T. tiliifolia	*E. fraxinifolia*	*H. latifolia*
Girth size (DBH)	16.485***	9.950***	4.997***
Forest type	-	0.406^{ns}	0.100^{ns}
Year	0.003^{ns}	0.041^{ns}	2.472^{ns}
Girth size x forest type	-	1.075^{ns}	0.276^{ns}
Forest type x year	-	3.547^{ns}	0.299^{ns}
Girth size x year	0.015^{ns}	0.706^{ns}	2.869^{ns}
Girth size x forest type x year	-	0.294^{ns}	0.091^{ns}

****P*<0.001, ns – not significant

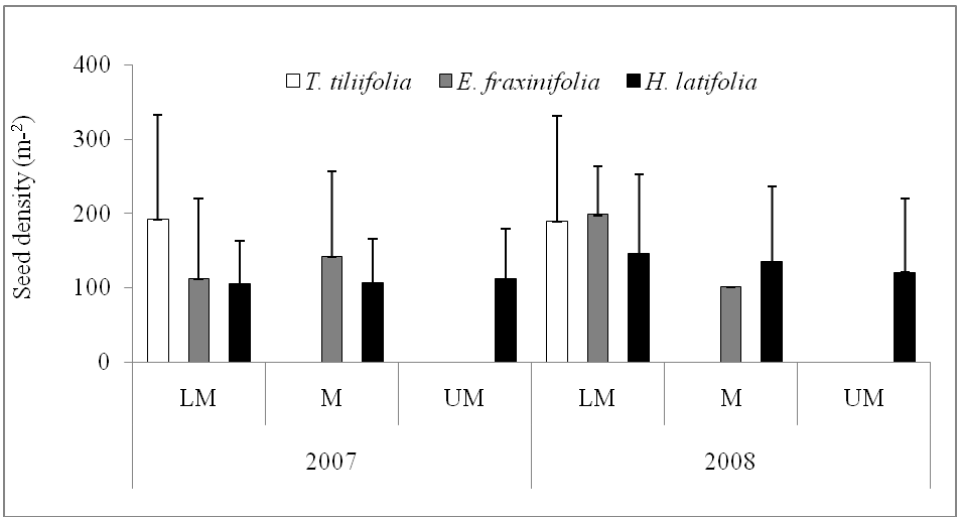

Figure 7.3. Density of the seeds of selected species in the soil seed bank in Lower montane (LM), Montane (M), and Upper montane forests. (Mean and SD; n = 10).

Seed viability

Seed viability of the selected species decreased consistently along a temporal scale (Table 7.7). *H. latifolia*, recorded viability period of almost two years but decreased considerably with time. *E. phaseoloides* and *E. fraxinifolia* recorded viability period of 15 months but that also decreased considerably with time. In case of *T. tiliifolia*, seed germination was high even after 9 and 12 months of storage period.

Table 7.7. Seed Viability (%) during storage at room temperature (values are based on a sample of 100 seeds)

Time interval (months)	*H. latifolia*	*E. phaseoloides*	*E. fraxinifolia*	*T.tiliifolia*
0	84±2.08	93±1.20	49±3.35	3±0.54
3	77±1.85	88±0.88	43±2.08	2±0.57
6	66±3.05	70±3.75	40±1.20	3±0.54
9	61±1.45	52±1.45	35±1.45	2±0.57
12	42±1.45	35±2.51	12±1.45	6± 0.37
15	35±2.51	11±0.57	8±0.57	0.00
18	33±2.02	0.00	0.00	0.00
21	32±1.76	0.00	0.00	0.00
24	17±1.45	0.00	0.00	0.00

Seed germination

Seed germination in all the species varied in different forest stands and in litter treatment. Seeds of *Holboellia latifolia* germinated well especially in the Montane forests (Figure 7.4). Stand quality characterised by forest type, and litter treatment significantly affected seed germination for *H. latifolia* and *E. fraxinifolia* species ($P < 0.001$), and in *E. phaseoloides* ($P < 0.05$) (Table 7.8).

Table 7.8. ANOVA for testing the effects of forest type and leaf litter treatment on seed germination of selected species.

Source of variation	F-ratios		
	H. latifolia	*E. phaseoloides*	*E. fraxinifolia*
Forest type	42.420***	3196.271***	241.814***
Treatment	221.480***	4.451*	18.941***

*P<0.05, ***P<0.001, ns - not significant

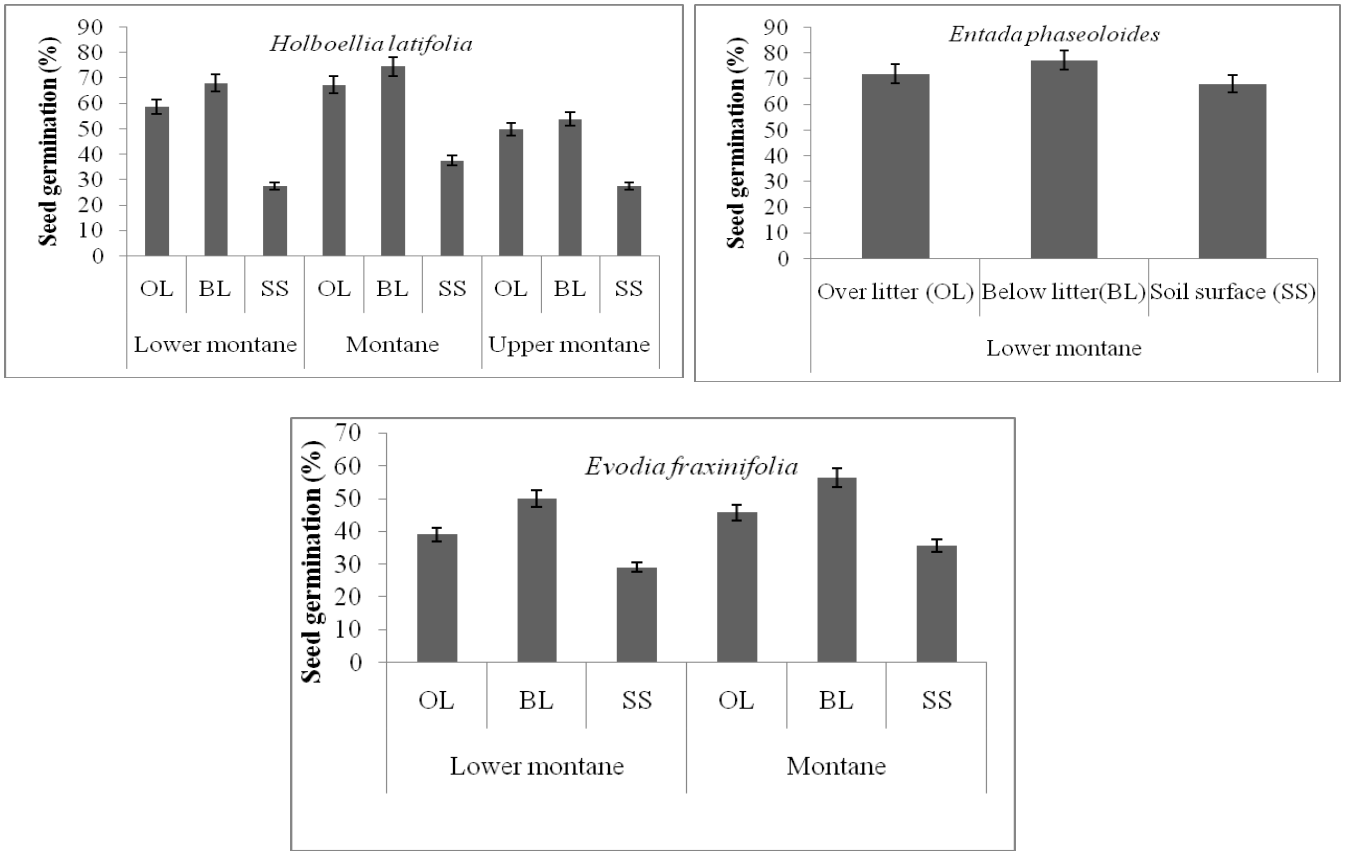

Figure 7.4. *In situ* seed germination under different treatments.

Seedling recruitment and population dynamics

Seedling recruitment for *H. latifolia* was higher in the Montane forests than in the Lower and Upper montane forests. For, *Evodia fraxinifolia*, seedling recruitment was higher in the Lower montane than in the Montane forests. Year wise, the seedling recruitment for all the species was higher in the year 2008 than in the year 2007 (Table 7.9).

Table 7.9. Seedling recruitment and mortality of the selected species in three forests.

Species	Forest type	Recruitment 2007	Recruitment 2008	Mortality (%)
Entada phaseoloides (liana)	LM	90	130	56.25
Holboellia latifolia (liana)	LM	120	134	61.90
	M	142	160	62.04
	UM	123	154	57.50
Evodia fraxinifolia (tree)	LM	116	132	70.59
	M	112	127	79.00

High seedling mortality of *H. latifolia* occurred during the three months (March-June) of germination. However, the seedling survivorship curves for *E. fraxinifolia* and *E. phaseoloides* showed a sharp reduction in the number of individuals after 3 and 6 months period respectively and continued till the seedlings were one year old, after which the seedling population stabilized (Figure 7.5).

The pioneer-climax dichotomy

The pioneer-climax axis or dichotomy among the Montane woody species has been associated with many features of their biology as evaluated in Table 7.10. Based on the seven life history traits four studied, species were classified into pioneer and non-pioneer category. *Holboellia latifolia*, *Evodia fraxinifolia* and *Toricellia tiliifolia* may be classified as pioneer species, while *Entada phaseoloides* as intermediate to non-pioneer species.

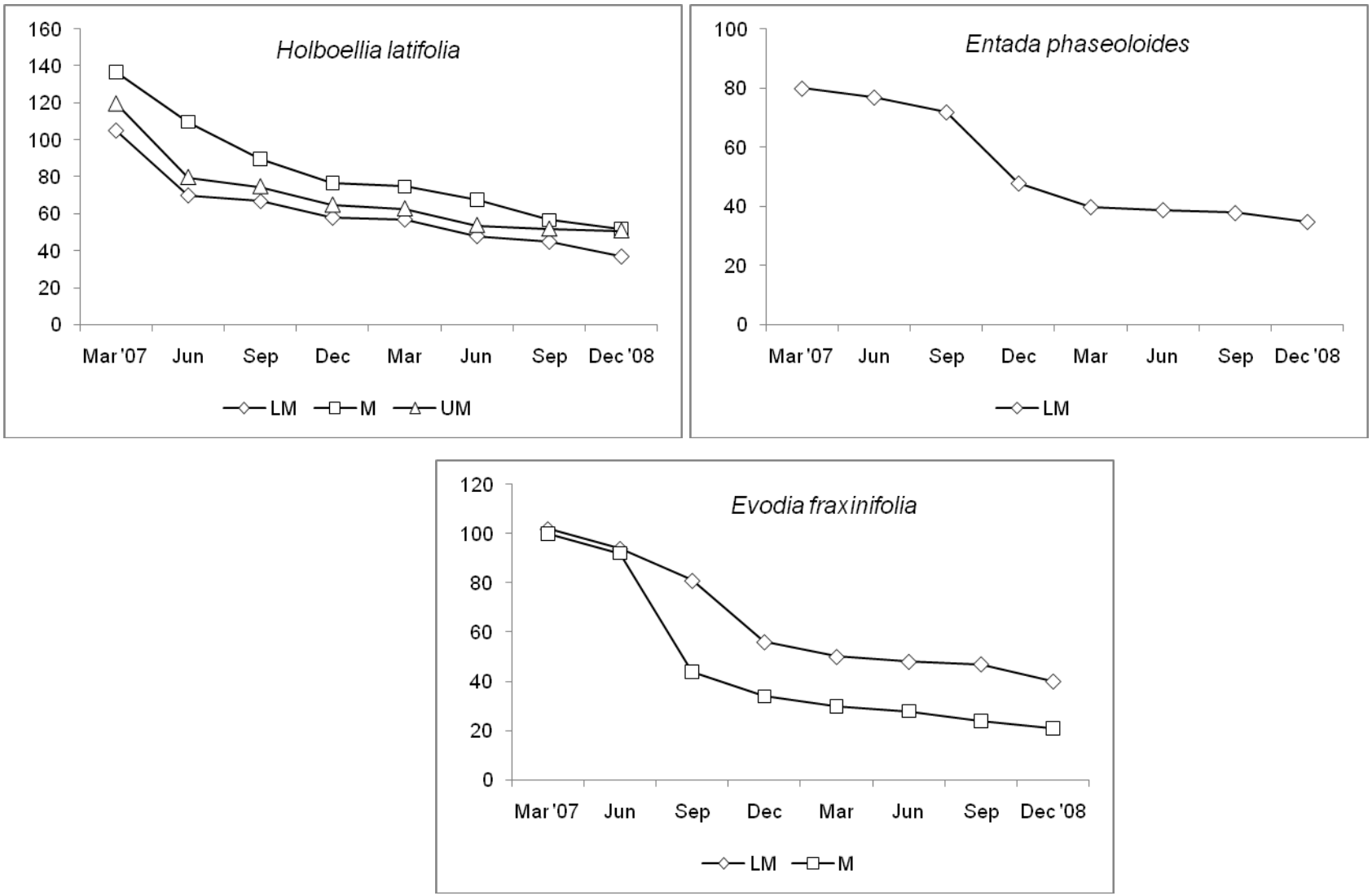

Figure 7.5. Seedling survivorship curves of the selected species in the three forests of KBR.

Table 7.10. Life history trait of four species studied and used for classifying them into two distinct guilds viz., pioneer and non-pioneer species.

Life history trait	*H. latifolia*	*E. phaseoloides*	*E.fraxinifolia*	*T. tiliifolia*
Habitat	Intermediate	Intermediate	Intermediate	Intermediate
Seed weight	Small	Large	Small	Small
Soil seed bank	Large	Small	Large	Large
Leaf mass per unit area	Larger	Larger	Smaller	Smaller
Disturbance effect	Intermediate	Intermediate	Intermediate	Edges/periphery
Dispersal mechanism	Zoophily	Hydrophily	Zoophily	Hydrophily
Climbing mechanism	Twining	Tendriller	Not applicable	Not applicable

Effect of forest fragmentation on seedling density

The mean seedling density of all the species varied significantly under different habitat, and fragment type for all the 25 fragments. However, seedling density of *E. phaseoloides*, did not vary along the periphery and interior of the forest fragments (Table 7.11).

Table 7. 11. ANOVA to test for the effects of habitat and fragment type on mean seedling density for the selected species in 25 forest fragments.

Effect	**F-ratio**			
	E. fraxinifolia	*E. phaseoloides*	*T. tiliifolia*	*H. latifolia*
Habitat	8.29***	1.00^{ns}	5.20**	39.26***
Fragments type	11.79***	3.00***	6.61***	27.25***

***P*<0.005, ****P*<001, ns- not significant

The relationship between microenvironmental variables and seedling density as shown by stepwise forward multiple regression analysis, indicated that phosphorus for *E. fraxinifolia* and *H. latifolia*, and both soil pH and light for *T. tiliifolia*, pH alone for *E. phaseoloides* were important determinants of seedling abundance across the 25 forest fragments (Table 7.12).

Table 7.12. Results of forward stepwise multiple regression analysis of environmental variables with mean seedling density across the fragments.

Environmental variables	Coefficient	Standard Coefficient	Standard Error	t	Probability>t	Constant
E. fraxinifolia						
Phosphorus	-0.024	-0.643	0.006	-4.02	0.001	1.121
T. tiliifolia						
pH	0.556	0.701	0.099	5.613	0.000	-2.573
Light	0.007	0.302	0.003	2.418	0.024	
E. phaseoloides						
pH	0.204	0.548	0.065	3.142	0.005	-0.909
H. latifolia						
Phosphorus	-0.022	-0.638	0.006	-3.975	0.001	1.435

7.4 Discussion

Regeneration of selected species i.e. two lianas *E. phaseoloides*, *H.latifolia* and two tree species i.e. *T. tiliifolia* and *E. fraxinifolia* selected for detailed study showed good regeneration in their respective forest stands. The probable reason could be favourable niches that the present sites provide them for active regeneration. The selected species *E. phaseoloides* was found only in Lower montane forests while *E. fraxinifolia* was found both in Lower montane and Montane forests, whereas *H. latifolia* was found in all the forest types studied. Thus, the spatial distribution of the parent plant could also have some role in the overall regeneration of the species in question.

The reverse J-shaped population structure of *Entada phaseoloides*, *Holboellia latifolia* and *Evodia fraxinifolia* indicated prevalence of favourable condition for regeneration of this species. Similar population structure was reported by Debanski et al. (2000) from the Australian subtropical rain forest and Rao et al. (1990) from subtropical montane forests of north-east India. Lower rate of natural regeneration of monotypic genera, *T. tiliifolia* as evident from J-shaped regeneration in the Lower montane forests stands could be that during the month of May the seeds are exposed to a long moisture excess condition and high temperature (July-August). This is followed by heavy rainfall

(May onwards). During this period (May-August) a substantial amount of seed is lost and whatever seedlings germinate have to undergo intense competition with the thick ground vegetation. This could be the possible reason for poor seedling recruitment.

The regeneration process in the respective forests has been affected by the different intensities of disturbance. While seedling are the chief mode of regeneration in all the forest types, vegetative mode of regeneration through coppices/sprouts has also a role to play in the regeneration mechanism along with seedlings for respective species and in forest types. Many workers have found that sprouting contributes significantly to natural regeneration in tree cut forest stands (McLaren & McDonald 2003b). Studies on the effect of stump size on sprouting of selected trees and liana species showed significant positive results. Average number of sprouts was also more in the higher girth classes of 26-35 cm and 36-45 cm in *Evodia fraxinifolia* and 14-19 cm and 20-25 cm in liana species, *Holboellia latifolia*. It is evident that liana species tend to initiate sprouting at lower girth classes than tree species. This raises the question of the specific contribution of the ramets (broken and fallen branches that resprout and form roots) versus the genets (single individual plants from sexually formed seeds) in the composition of *Holboellia* thickets in its natural habitat. In Panama, Putz (1984) noted the propensity for lianas to sprout vigorously from fallen stems. Based on seedling excavations, Putz found that 90% liana species in the understorey were ramets.

Bellingham (1993) reported that larger sized stems produced on an average more shoots and larger leading shoots which might be as a result of larger sized stems having greater carbohydrates reserves that can be mobilized to facilitate higher levels of

sprouting. The larger sized stems are bound to have a larger root biomass which will also be able to trap more of the resources needed to support growth.

Conversely, Khan and Tripathi (1987) reported that in a disturbed subtropical wet hill forest of north-east India, sprouting percentage of the stumps and number of sprouts per stump of *Alnus nepalensis*, *Quercus dealbata*, *Q. griffithii* and *Schima khasiana* decreased with stump diameter. This has been attributed to the increasing bark thickness which provides mechanical hindrance while sprouting and physiological changes in tree species with advancement of age due to which the capacity for rejuvenation by vegetative means decreases.

Coppice shoot density had a significant effect over diameter ($P < 0.05$) on both liana and tree species. High shoot number may not be an indication of successful vegetative regeneration. They may act as an indemnity against the death of one or a few leading shoots, result in significantly lower biomass recovery. McLaren and McDonald (2003) reported that coppice regrowth offered a considerable resilience to disturbance in a disturbed tropical dry limestone forest in Jamaica where successful regeneration by seed is highly susceptible to rainfall seasonality. Thus in this case regenerating through coppicing could help in recuperation of fragmented forests.

Phenological behaviour of the study species showed differences as well as similarities among each other in patterns of vegetative growth and reproductive development. Comparisons among the studied species shows that the period of overlapping between primary shoot growth, flower bud formation and flowering increased with an increase in seed/fruit size. For example, in *Endata phaseoloides* (seed size-14.10±3.54) and Holboellia *latifolia* (0.10±0.02), the above mentioned phenophases

overlap with each other, but maintained a strict chronology of events; whereas in *Toricellia tiliifolia*, and *Evodia fraxinifolia* there were less overlapping in the phenophases.

The phenological events of *E. phaseoloides* recorded greater overlapping; vegetative phases coincided with reproductive development. Castro-Diez et al. (2003) found in some woody species of the Mediterranean region that species with big fruits/seeds exhibited a high degree of overlap between primary shoot growth, flower bud formation and flowering, which has been related to being negatively selected for by the risk of frosts, and by the internal competition with fruit maturation, respectively. The three phenophases would have been forced to occur simultaneously within a shorter period and to share the available resources. On the contrary, the shorter length of the fruit setting period in small-fruited species leaves a longer period to complete primary shoot growth, flower bud formation and flowering so that they can be protracted to reduce competition between them (Castro-Diez et al. 2003).

In the present study, species with bigger reproductive organs (*Entada phaseoloides*) required longer periods to ripen compared to other species. On the contrary, *Toricellia tiliifolia*, *Evodia fraxinifolia* and *H. latifolia* had a shorter vegetative growth period compared to other species. Similar observations were made by Primack (1985) in trees of Florida, Eriksson and Ehrlen (1991) in north-European plants and by Castro-Diez et al. (2003) among Mediterranean woody species. This could be tentatively explained following Castro-Diez et al. (2003), that the carbon allocation shift from vegetative to reproductive meristem would occur earlier in species of bigger fruits or seeds, thus affecting the period of vegetative growth. The average duration of vegetative

growth in lianas was higher than in tree species, where as the differences between selected trees species was not that apparent. Fruiting in lianas was concentrated during the dry season but in tree species, it almost coincided with rainy season. It could be argued that liana species due to its nature of occurring in the disturbed habitat needs higher intensity of light (personal observation) for germination that is accomplished only in the dry season when forest canopy relatively open and increases the chances of survival.

Flower and fruit production varied significantly across DBH classes for all the species. Overall production increased with girth sizes. This could be attributed to the larger crown size of the individuals of higher girth classes (Bhuyan 2002).

Forest stands significantly affected production in *Evodia fraxinifolia* and *Holboellia latifolia* that had individuals producing more number of fruits/seeds in the Lower montane compared to the Montane forests whereas, in *H. latifolia* flower and fruit production was higher in Upper montane forests as compared to Montane and Lower montane forests. This is in conformity with Barik et al. (1996) who attributed greater fruit production to increased availability of sunlight in the disturbed stands. For instance, high light intensity may elevate bud temperature, which may lead to increase in the concentrations of growth regulators particularly gibberellins (Pharis & Kuo 1977; Ross et al. 1983), stimulating flowering and fruiting. Moreover, light regime associated with temporary water stress in disturbed stands is known to stimulate bud initiation in some forest trees (Kozlowski 1981).

Soil seed bank plays an important role in maintaining the ecological and genetic diversity of forest communities (Thomson & Grime 1979) and in assuring community

regeneration following disturbance (Houle & Phillips 1988). The size of the seed bank differs between the species. Seed bank of *Holboellia latifolia* displayed a significant spatial variation within as well as across forest stands with less seed density in the Upper montane forests. As the fruits of *H. latifolia* are a good source of food for the birds, possibility of the fruits being consumed by them is always inevitable. However, some of the fruits get hidden under the litter layer thus preventing the frugivores from noticing it.

In situ seed germination for the selected species was significantly affected by forest type (characterised by open or closed canopy). The general trend shows that seed germination percentage was more in case of seeds placed below litter. This could be attributed to the moisture content inside the litter which has helped the seed in germination. Effect of temperature alterations or changes in the light environment, associated with gap formation on seed germination has been explained by many workers (Bazzaz & Pickett 1980; Denslow 1987). Ulft (2004) argued that it might be due to the reason that the seedlings are dependent on photosynthesis soon after germination.

Maximum number of recruitments occurred for *Holboellia latifolia* and was highest in the year 2008. *Entada phaseoloides* had the least recruitment amongst all the selected species. Seedling recruitment in *Evodia fraxinifolia* was also more in the year 2008 compared to 2007. This could be attributed to greater production of fruits and seeds, associated with copious seed germination.

Considering the species response to different forest environment and their life history trait as well as the species trait, *Holboellia latifolia*, *Evodia fraxinifolia* and *Toricellia tiliifolia* may be classified as pioneer or early successional species, while *Entada phaseoloides* as intermediate to non-pioneer species. Similar studies were carried

out to delineate different group of plant species based on their specific response to different life history traits in the tropical forests of the world (Swaine & Whitmore 1988; Whitmore 1998; Bazzaz 1991).

Fragment type can influence seedling abundance (Benitez-Malvido 1998). The density of naturally occurring selected species seedlings was substantially higher in larger fragments than in smaller ones. The seedling densities of four species were higher in the periphery than in the interior of forest fragments. It appears that the factors reducing seedling germination and subsequent establishment are operating in the interior of fragments or otherwise, microenvironmental factors may be conducive for the growth of the four selected species along the periphery. The changes in canopy structure at the edge may also influence species composition of seedlings (Ward & Parker 1989). During seedling stage, mean density was strongly related to soil phosphorus for *E. fraxinifolia* and *H. latifolia*, indicating the importance of these factors in seedling establishment. Light intensity and soil pH were positively correlated with seedling density of *T. tiliifolia* species, indicating the role of light only during the juvenile phase of the species. The role of light in the establishment of seedlings of some woody species has been argued by Cai et al. (2007). On the other hand, it must be considered that soil pH interacts with the many biotic and abiotic soil factors which might affect the effectiveness of an ectomycorrhizal isolate to improve plant growth. The correlation of seedling density with various microenvironmental factors provides important cue for managing the regeneration of these important plant species. However, liana species can withstand this drastic environmental changes and able to flourish along the edges of the fragments. Smaller fragment had more or less similar liana density because of similar

microenvironment in the interior and along the edges of the fragments. Laurance et al. (2000, 2001) had similar finding from BDFFP study sites, where the liana are known to favour forest disturbance and tend to increase in density and diversity along forest edges.

It could be concluded that forest types significantly affects the seed production, seed dispersal, seed bank as well as seed germination by creating a heterogeneous environment of abiotic factors (by changing the microclimate) along with the biotic factors (chiefly the animal components). Finally it can be inferred that seedling survival and mortality of the selected species in the different forest stands in Khangchendzonga Biosphere Reserve is governed by both endogenous as well as exogenous factors. Nevertheless, seedling growth rate of the selected species is strongly controlled by environmental factors and forest types. The shifts in abundance of tree/liana seedling species in fragments of different classes suggest that liana species can withstand the environmental alterations produced by forest fragmentation, whereas tree species are susceptible and tend to disappear from fragments.

Chapter 8
General Discussion

Fragmentation of habitats has serious impacts on landscapes by affecting ecosystems, populations and species (Liernet 2004). Various anthropogenic activities cause loss of habitats of several plant and animal species. Such activities may reduce fragment size, increase distance between fragments and enlarge the edges at the expense of interior habitat. The combined effect of all these may lead to local extinction (Fischer & Stöcklin 1997). The process of fragmentation of natural habitats is increasing exponentially worldwide and represents one of the foremost threats to biological diversity. Forest fragmentation affects demographic and genetic structure of forest plant populations by enhancing the edge effect, changing the interactions between pollinators, hampering the migration between fragments and inducing a genetic drift and potential inbreeding depression (Liernet 2004; Honnay et al. 2005). Although a substantial number of studies investigating the effects of fragmentation on animal populations are available (Fischer & Lindenmayer 2007), such studies on plant species are rare (Zacharias & Brandes 1990; Lawesson et al. 1998; Honnay et al. 1999; Feoli et al. 2003; Hobbs & Yates 2003; Kolb & Diekmann 2005).

The present research work was undertaken in the high elevational ranges of Khangchendzonga Biosphere Reserve in Sikkim. Various anthropogenic and natural disturbance causing factors such as wild fire, landslide, trekking , tourism, cattle grazing, windthrow, snow avalanche etc, responsible for creation of fragments were identified, characterized and studied in detail to quantify their frequency and intensity. Detailed analysis of vegetation was undertaken in different forest types and the effect of forest fragmentation on tree diversity was assessed. Regeneration of two important tree and

liana species was studied and the impact of fragmentation on regeneration was analyzed. The ecological questions which have been answered through this work are:

1. How the diversity of vascular plants varies in different forest types of KBR?
2. Which microenvironmental factors are related to the plant diversity in different forest types?
3. How the spatial and temporal patterns of forest fragmentation varied in KBR during the study period?
4. What are the causes of forest fragmentation?
5. What was the intensity of disturbance causing forest fragmentation?
6. Do larger forest fragments have a greater diversity of woody plants than the smaller ones?
7. How the microclimatic variables differ spatially i.e. fragments vis-a-vis continuous forest, along a fragment size gradient, and from forest edge to the interior within a fragment?
8. Can such spatial variations in microclimatic conditions be related to the observed pattern of tree diversity?
9. How tree and liana species respond to fragmentation during their regeneration phase?
10. How seedling populations vary in the periphery and interior of the forest fragments?
11. What are the various microenvironmental factors that contribute most to the seedling density in the forest fragments?

The three forest types viz., Lower montane, Montane and Upper montane differed in species composition and community characteristics. In total, 390 plant species belonging to 262 genera and 112 families were recorded. This included 78 tree species belonging to

47 genera and 30 families across the three forest types. Thirty eight shrub species belonging to 35 genera and 17 families were recorded. On the forest floor, 133 herb species belonging to 97 genera and 49 families were reported. Ninety two epiphyte species belonging to 57 genera and 31 families were documented. In total 43 liana species belonging to 37 genera and 28 families were recorded. The total numbers of species, genera and families in the Lower montane forest were greater than the Montane and Upper montane forests. The number of tree (36), shrub (26), herb (54), and liana (30) species reported from the Lower montane forests in KBR were less than that reported from Central Himalayan region (Kharkwal et al. 2005). The number of life forms was maximum in mid altitude zone of subtropical belt. The distribution was unimodal, with maximum values in the Lower montane and minimum in the higher elevational gradient. Empirical studies of Rahbek (1995), Wang et al. (2002) and Grytnes and Vetaas (2002) found that this pattern, in which species richness achieves maximum values in lower to intermediate elevations, is the most common one in a variety of ecosystems. According to Vetaas and Grytnes (2002), about half of the published studies showed a mid-elevation peak in plant species richness. The mid-elevation peak may also be a result of the intermediate location between the montane Himalayan flora and the flora in the lowland, which increases the chances for immigration from both directions, i.e. a mass effect (Shmida & Wilson 1985; Grytnes & Vetaas 2002). Hua (2002) also found similar distribution pattern of plant life form in Hubei province in China. It is similar to those of pteridophytes in Panama with maximum species at 500-1500 m elevational zone (Lellinger 1985). Similar observations were made for vascular plant species of semi-natural subalpine grasslands in Vang, Southern Norway (Austrheim 2002), and for tropical rain forests species (Lieberman et al. 1996; Vazquez & Givnish 1998).

The species richness in different forest types along an elevation gradient is governed by a series of interacting biological, climatic and historical factors (Colwell & Lees 2000). Further, elevation represents a complex gradient along which many environmental variables change simultaneously (Austin et al. 1996). Therefore, it is often referred to as a 'proxy' variable, including in the present study, while considering various environmental factors.

Several hypotheses have been put forward to explain elevation patterns of species richness. For example, optimum humidity conditions at mid-elevations (Rahbek 1995, 1997) and the high productivity in the mid-elevation region were related to high species richness and optimum resource combination concept was proposed (Rosenzwieg 1995). The observed species richness patterns of vascular plants in Khangchendzonga BR are in accordance with the assumption of productivity and optimum resource combination in the intermediate portion of the elevation gradient. The Lower montane forests with an optimal combination of environmental resources were preferred by many species to coexist (Brown 2001; Lomolino 2001). Therefore, large numbers of species of different life forms were found in this forest type in KBR.

The major decline in species richness above 1900 m elevation in KBR could be due to ecophysiological constrains, reduced growing season, low temperature and low ecosystem productivity in higher elevation (Körner 1998) and local edaphic factors. Moreover, a limited species pool of spermatophytes also affects the species richness in higher elevation, as environmental constraints are expected to exclude species from high elevational forests (Körner 1995). Hamilton and Perrot (1981) believed that the structure and distribution of plant communities in the higher mountain slopes are often related to temperature and other climatic factors, while those at lower elevations may be determined by more benign biotic or abiotic factors.

In mountain regions, the pattern of different forest types and other communities often corresponds to elevation and topography. Variation in microclimate with topography and elevation is a major factor of species distribution within a forest landscape. Mark et al. (2000) found topographic features (elevation, exposure and slope) to be responsible for the macroscale patterns of alpine vegetation distribution on Mount Armstrong in New Zealand. The empirical evidences relating plant diversity and microenvironmental factors in KBR have been provided in chapter 4.

Higher α- diversity of trees, herbs and lianas in the Lower montane forests was also attributed to the existence of a wide range of vegetation formation such as lowland forest, transitional forest, riverine forests etc. Among others, the diversity of different lifeforms in Lower montane forests can be attributed to the prevailing monsoon effects in the region, which remains one of the major factors for high vegetation diversity in the main Himalayan region (Singh & Singh 1987). Being at the meeting point of Indo-Malayan and Indo-Chinese biogeographical realms as well as Himalayan and peninsular India, it contains the floristic elements from all the biogeographical zones.

The variation in α and β-diversity values for epiphytes in the three forest types may be due to spatial microhabitats, following a gradient from moist part of the studied forest (Lower montane) to the drier part (Upper montane) and suggests that the distance to moisture source plays a crucial role in determining richness and composition of epiphyte communities. It has also been argued that epiphyte richness is associated with moisture of the slopes where they grow (Sanford 1968; Sudgen & Robins 1979). β-diversity measures the extent of species replacement or biotic change along environmental gradients (Whittaker 1972; Brokaw & Scheiner 1989). It also reflects the extent of similarity and habitat diversity among the forest types. In the present study, β-diversity for trees and lianas is lower than the shrub and herb components. This is similar

to that in BCI forest studied by Brokaw & Scheiner (1989). Low β-diversity might be due to variation in microenvironmental gradient, dispersal mechanism and abundance of abundant climax juveniles.

The lower species replacement rate between samples in the Lower montane forest portion of the elevational gradient suggests that environmental conditions prevailing there favour the coexistence of a larger number of species (Lomolino 2001; Wang et al. 2002). In contrast, the high species replacement rate in the higher elevation is indicative of wide environmental differences, and of adaptation among the functional types (Wang et al. 2002). Few species can tolerate the full spectrum of environmental conditions at gradient extremes (Sánchez-González & López-Mata 2003).

Similarity test for species composition among the three forest types showed that the forests were significantly dissimilar. These variations in species composition are primarily the result of subtle elevational variations (Grell et al. 2005).

The tree basal area of the Lower montane forests was higher than that of Montane and Upper montane forests which could be attributed to more number of individuals in different girth classes. The density-diameter distribution of tree population has been used to understand regeneration, disturbances and future stability of tree populations in forests communities (Rao et al. 1990).

The low dominance index value for different lifeforms in the Lower montane forests indicates more equitable resource distribution pattern among the constituent species than those in the Montane and the Upper montane forests (Crawley 1997). Such equitable resource distribution pattern might have made the Lower montane forests more species rich in comparison with Montane and Upper montane forests.

The total dominance of the Polypodiaceae among the epiphytes is in conformity with the general trend in humid montane epiphyte communities. Ordination of epiphyte

species on the basis of abundance data with respect to forests types resulted into slight overlapping of the Montane forests species with the Lower montane and Upper montane forests species. The similarities between orchids and fern species among different forests are mainly responsible for this overlapping.

The abundance of lifeform was influenced by the two most abundant families, Polypodiaceae and Orchidaceae. The abundance of this lifeform was also reported by Freiberg (1996) from French Guiana.

The host trees of epiphytes selected for the study ranged from ≥ 35 to < 90 cm DBH. The occurrence of large number of species on bigger girth classes can be explained by the larger area offered with a great variety of host architecture with different microhabitats for epiphytes (Annaselvam & Parthasarathy 2001). The significant relationship found between epiphyte species and trunk girth class in the three forests conforms to the report of Catling and Lefkovitch (1989) in Gautemalan forest. The epiphytic species richness increased with increase in height class of the host trees. Tree base is poor in epiphytes, only some Araceae inhabits in this strata. The richest trunk vegetation is found on the mossy substratum. This is common in the upper montane and montane forests on *Abies densa* and *Quercus lamellosa* host respectively. Harbouring more epiphytes in upper canopy in the three forest types may be because of bryophyte mats and fork of tree trunk which accumulate litter and humus and provide mechanical support.

Seasonal variation in air, soil temperature, moisture content, C and N concentrations as observed in the present study corroborates the findings of Barik et al. (1992) in a subtropical broad-leaved forest of north-east India. Differences in soil properties, elevation, topography and other environmental conditions in different forest types could explain substantially the observed differences in plant species diversityand

abundance in the three forests. An observed gradient in many environmental variables investigated was also related to the differences in structural and functional characteristics of the forest types studied along an elevation gradient in Tierra del Fuego by Frangi et al. (2005).

Relatively lower eigen values of the first two constrained CCA axes and greater eigen values of the first residual (non canonical) axis as obtained in the present study apparently indicate that the environmental variables are not sufficient to predict the main variations on species abundance extracted by CCA, but they do predict a substantial part of remaining variations in three forest types.

The strong clustering of lower montane trees, shrubs, herbs and lianas along the soil pH and light gradients in the CCA ordination plot supported the earlier observations on plant preference for less acidic soil and light (Lowe & Walker 1977; Putz 1984; Whitmore 1989; Phillips & Gentry 1994). The important role of light in determining the density and distribution of many liana species such as *Cissus repens*, *Clematis acuminata* and *Parthenocissus himalayana* is in conformity with the findings of Castellanos (1991), who concluded that liana species thrives well in areas of abundant light in the forest. Soil pH is related to the availability of soil nutrients, which helps in the overall growth of plants. Increase in species richness from acidic to neutral soil is common in temperate forests (Palmer 1990; Pausas 1994) and a pattern of richness increasing with higher pH has been reported in the Arctic tundra (Gough et al. 2000). Predictable variation in the relationship between richness and pH suggests strategies for biodiversity conservation. However, in the montane and upper montane forests, the composition of trees, shrubs, herbs and lianas is mainly driven by the edaphic variables (N, P and K). As shown by stepwise forward multiple regression analysis, light and soil P either alone or both influenced liana density in different forests. Soil pH in lower montane, and C in montane and upper montane forests influenced tree density. Liana and tree density in the montane

and upper montane forests were strongly related to soil nutrients such as N, P and pH, C respectively. Light and C either alone or both influenced shrub density in montane and upper montane forests. Soil pH, elevation in the montane, N and P either alone or both influenced herb density in montane and upper montane forests respectively. The role of soil nutrients in plant species distribution was emphasised by Dewalt et al. (2000, 2006) and Godefroid et al. (2007) corroborating to the present finding.

Epiphytic species had strong clustering around RH, elevation and air temperature. Result of stepwise forward multiple regression analysis also showed the influence of light, elevations and relative humidity on epiphyte density. The elevational gradient and role of RH in epiphyte richness was also emphasised by Kufer et al. (2004), and Kharkwal et al. (2005) corroborating to the present finding. But many epiphytes require high exposure and others like certain filmy ferns, cannot endure either as much light or the associated aridity (Hietz & Briones 1998). Consequently, epiphytes segregated along environmental gradients in different forest types.

Various mechanisms of forest fragmentation have been summarized following Jaeger (2000) (Figure 8.1). Of the 25 fragments studied in detail, 10 fragments followed incision process, 8 followed dissection process, 4 followed perforation and rest followed other remaining processes. Thus, incision was the main process of fragmentation in KBR.

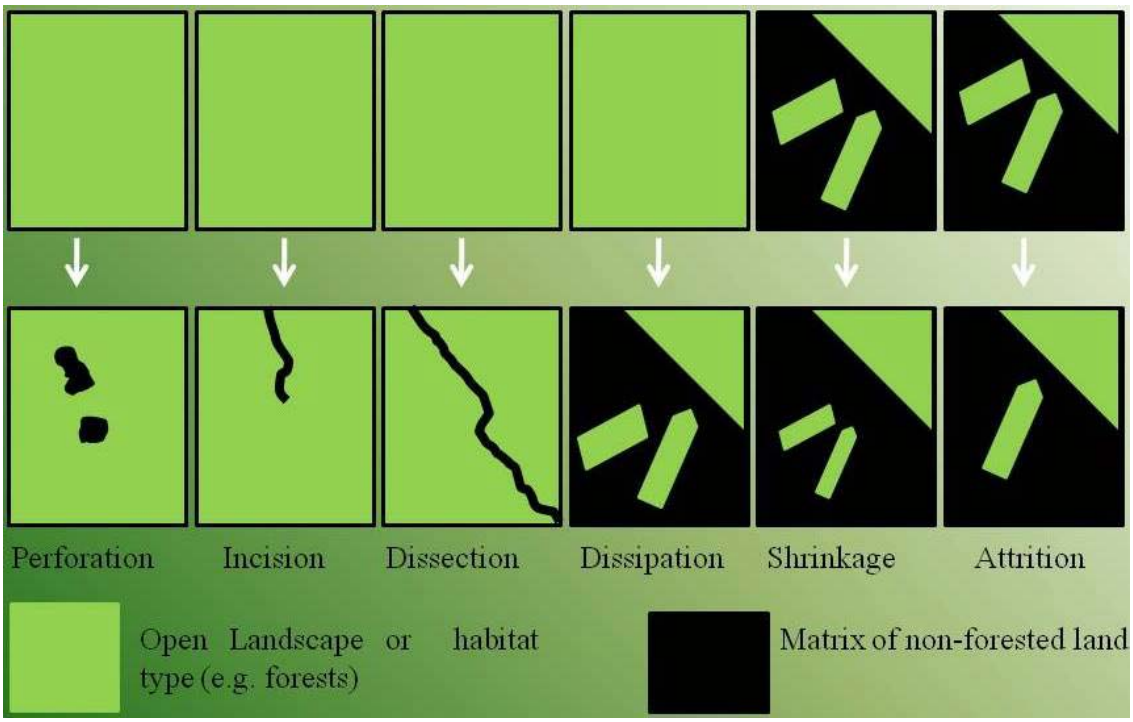

Figure 8.1. Processes of forest fragmentations (after Jaeger, 2000).

The results of the study on temporal change in forest fragmentation pattern in KBR helped in detecting the type of change, location of change, and quantifying the changes taking place in KBR. Land cover changes over the study period showed conversion of forest category into other classes thereby fragmenting the natural forest cover. Results from the imageries confirmed the decrease in meadow and open forest areas during the study period. This has been attributed to the extensive grazing by cattle and human disturbances in the high elevation forests in the past. The decrease in glacier beds on other hand might be related to global climate change phenomena.

Forest fragmentation did not occur as continuous process. In the present study, the forest fragmentation was more during 1999 than in the subsequent time series. As evident from the imageries, there has been discontinuity in the formation of forest fragments. On an average, 674 (± 103.1) fragments were present in the KBR during the study period and major portion of fragments were in the smallest size classes (< 1 ha). The number of forest fragments during the last decade (1999) was at its peak and declined towards 2008.

Average annual fragmentation in the KBR was 0.007%, which is comparatively less than others studies around the world. Keles et al. (2008) in their study at Trabzon province reported higher annual rate of forest fragmentation (0.41%). Li et al. (2009) in their study in Alabama forest have shown that the rate of forest fragmentation may go as high as 2% annually.

Lesser number of fragments in higher size classes will have significant effect on the response of some species in the study area. Whereas, abundance of large number of forests fragments in lower size class might be due to landslide, a universal phenomenon in the mountain ecosystems, and hilly areas of Sikkim. There had been heavy mud triggered avalanche in 1995, which also affected KBR, which might have created the forest fragments.

The increase in fragmentation is related either to natural or anthropogenic sources (Wade et al. 2003; Geist & Lambin 2001). The forest fragmented by anthropogenic factors is at higher risk of further fragmentation or removal than forest fragmented by natural causes (Wade et al. 2003). Identifying only anthropogenic causes of forest fragmentation may be a useful tool for policy and decision makers, allowing for improved risk assessments and better targeting of areas for protection or remediation.

Ranking of disturbance parameters according to its observed intensity although seems arbitrary, it does characterize various disturbances that took place during the study period. At the forest scale, pattern of disturbance may be strongly influenced by topography, pre-existence of matured forests, and time since past disturbance. But fragmentation due to chronic age old and low intensity disturbances like illegal transboundary grazing and trekking routes have substantially contributed to the creation of forest fragments in KBR.

The study revealed a decelerated rate of forest fragmentation in KBR over the time period 1999-2008. In KBR, reduction in the number of forests fragments during the study period might be due to (i) complete disappearance of a certain number of forest fragments converting them into continuous forest blanks, (ii) merger of forest fragments with main continuous forests through regeneration, especially in the montane and upper montane forest matrix, and (iii) stricter rules and regulations enforced by state forest department after the year 2000, imparting of forest awareness knowledge to local peoples living in fringe areas of KBR and protection measures taken by some local bodies and NGOs.

It was observed that there was difference in tree diversity according to fragment size. Given the slow response of tree population to isolation of remnant fragments, it is likely that the full impact of these changes will not become apparent for some time (Hanski & Ovaskainen 2002; Helm et al. 2006).

The 'α' diversity for trees in all the forest fragments was lower than that of adjacent continuous forests: alpha diversity ranged between 5-22 per fragment but variation was even greater in continuous forests, ranging between 7-25 species per site.
A positive tree species-area relationship indicates that species abundance was a function of fragment area, highlighting the importance of area as one of the most important determinants of species richness in fragmented habitats. This is an incontrovertible fact which was also reported by Page et al. (2010). Similar results were found for tree communities in Atlantic tropical forest in Brazil (Metzger 1997), where tree diversity of the forest fragments appeared to be similar among fragments of different sizes. In the same way, tree species diversity in the highlands of Chiapas, Mexico, is not related to fragment size or to any other spatial attribute (Ochoa-Gaona et al. 2004). However, in a study conducted in the montane Atlantic forests of south-eastern Brazil, fragment size

was found to be the major determinant of changes in woody plant composition and guild structure (Tabarelli et al. 1999).

Isolation plays an important role in colonization processes and is one of the important predictors of species diversity in fragmented biotas (MacArthur & Wilson 1967). In the present study, however, there is little evidence suggesting the influence of isolation on plant species richness.

Microclimatic condition differs in the forest fragments, and modified environmental conditions may not be appropriate for many species in continuous forests (Hobbs & Yates 2003; Laurance et al. 2002). This microclimatic variable in turn is influence by the vegetation and such influence have a different response by these factors, temperature for example decreases towards forest fragment centre, and it shows a steady decline. Light intensity on the other hand exhibited a steep diminution as depicted in the study.

Relatively lower values of the first two constrained CCA axes apparently indicate that the environmental variables are not sufficient to predict the distribution of tree species within the spatial gradient in a fragment, but they do predict a substantial part of remaining variations. Therefore the clustering of the tree species in different fragments differs with respect to environmental preferences. Clustering of FF6, FF7, FF8, FF9, FF10, FF11 and FF12 in TWINSPAN and species in DECORANA was due to environmental preferences as depicted through CCA.

TWINSPAN analysed the distribution of tree species and it separated them into eight communities. Tree species abundance within each community in different fragments is influenced by microenvironmental variables. Thus a significant change in light, soil temperature and moisture regime in fragments size gradient played an important role in influencing the composition and abundance of tree species.

Regeneration of selected species i.e. two lianas *E. phaseoloides, H.latifolia* and two tree species i.e. *T. tiliifolia* and *E. fraxinifolia* had good regeneration in their respective forest stands. The probable reason could be favourable niches that the respective sites make available for active regeneration. *E. phaseoloides* was found only in lower montane forest while *E. fraxinifolia* was found both in lower montane and montane forests, where as *H. latifolia* was found in all the three forest types. The spatial distribution of the parent plant played an important role in regeneration of the species in question.

Reverse J-shaped density-diameter curves as obtained for *Entada phaseoloides, Holboellia latifolia* and *Evodia fraxinifolia* indicated prevalence of favourable condition for regeneration of these species. Similar population structure was also reported by Debanski et al. (2000) for the Australian subtropical rain forest. Lower regeneration rate of monotypic genera, *T. tiliifolia* in the lower montane forest stands could be explained as follows. During the month of May the seeds are exposed to a long moisture stress condition and high temperature during July-August. This is followed by heavy rainfall (July onwards). During this period (May-August), a substantial amount of seed is lost and whatever seeds germinate have to undergo intense competition with dense ground vegetation.

Regeneration through seeds was the main mode of regeneration for all the four species. Vegetative mode of regeneration through coppices/sprouts did have a role to play in the regeneration mechanism particularly in the species having coppicing ability. Many workers have found that sprouting contributes significantly to natural regeneration in tree cut forest stands (McLaren & McDonald 2003b). Studies on the effect of stump size on sprouting of selected tree and liana species showed significant positive results. Average number of sprouts was more in the higher girth classes of 26-35 cm and 36-45 cm in tree

species (*Evodia fraxinifolia*) and 14-19 cm and 20-25 cm in liana species (*Holboellia latifolia*). It was evident that liana species tend to initiate sprouting at lower girth classes than tree species. In Panama, Putz (1984) noted the propensity for lianas to sprout vigorously from fallen stems. Bellingham (1993) reported that larger sized stems produced on an average more shoots, which might be result of larger sized stems having greater carbohydrates reserves that can be mobilized to facilitate higher levels of sprouting.

Coppice shoot density had a significant effect over diameter ($P < 0.05$) on both liana and tree species. High shoot number may not be an indication of successful vegetative regeneration. They may act as an indemnity against the death of one or a few leading shoots, result in significantly lower biomass recovery. McLaren and McDonald (2003) reported that coppice regrowth offered a considerable resilience to disturbance in a disturbed tropical dry limestone forest in Jamaica where successful regeneration by seed is highly susceptible to rainfall seasonality. In KBR, regeneration through coppicing could help in recuperation of fragmented forests.

Phenological behaviour of the study species showed differences as well as similarities among each other in patterns of vegetative growth and reproductive development. Comparisons among the studied species show that the period of overlapping between primary shoot growth, flower bud formation and flowering increased with an increase in seed/fruit size. For example, in *Endata phaseoloides* (seed size14.10±3.54) and *Holboellia latifolia* (0.10±0.02), above mentioned phenophases overlap with each other, but maintained a strict chronology of events; whereas in *Toricellia tiliifolia,* and *Evodia fraxinifolia,* there is less overlapping in the phenological events. Castro-Diez et al. (2003) found in some woody species of the Mediterranean region that species with big fruits/seeds exhibited a high degree of overlap between

primary shoot growth, flower bud formation and flowering, which has been related to being negatively selected for by the risk of frosts, and by the internal competition with fruit maturation, respectively. The multiple phenophases would have been forced to occur simultaneously within a shorter period and to share the available resources. On the contrary, the shorter length of the fruit setting period in small-fruited species leaves a longer period to complete primary shoot growth, flower bud formation and flowering so that they can be protracted to reduce competition between them (Castro-Diez et al. 2003).

In the present study, species with bigger reproductive organs (*Entada phaseoloides*) required longer periods to ripen compared to other species. On the contrary, *Toricellia tiliifolia, Evodia fraxinifolia and H. latifolia* had a shorter vegetative growth period compared to other species. Similar observations were made by Primack (1985) in trees of Florida, Eriksson and Ehrlen (1991) in north-European plants and by Castro-Diez et al. (2003) among Mediterranean woody species. This could be explained following Castro-Diez et al. (2003) that the carbon allocation shift from vegetative to reproductive meristem would occur earlier in species of bigger fruits or seeds, thus affecting the period of vegetative growth. The average duration of vegetative growth in lianas was higher than in tree species, where as the differences between selected trees species was not that apparent. Fruiting in lianas was concentrated during the dry season but in tree species, it almost coincided with rainy season. It could be argued that liana species occurring in the disturbed habitat needs higher intensity of light for germination that is accomplished only in the dry season when forest canopy is relatively open and, increases the chances of survival.

Flower and fruit production varied significantly across DBH classes for all the species. Overall production increased with girth sizes. This could be attributed to the larger crown size of the individuals of higher girth classes (Bhuyan 2002).

Forest types significantly affected production in *Evodia fraxinifolia* and *Holboellia latifolia*, which had individuals producing more number of fruits/seeds in the lower montane compare to the montane forest, whereas, in *H. latifolia*, flower and fruit production was higher in upper montane forest as compared to montane and lower montane stands. This is in conformity with Barik et al. (1996) who attributed greater fruit production to increased availability of sunlight in the open canopy forests.

Soil seed bank plays an important role in maintaining the ecological and genetic diversity of forest communities (Thomson & Grime 1979) and in assuring community regeneration following disturbance (Houle & Phillips 1988). Seed bank of *Holboellia latifolia* displayed a significant spatial variation within as well as across forest stands with less seed density in the upper montane forest. As the fruits of *H. latifolia* are a good source of food for the frugivorous species, possibility of the fruits being consumed by them is always expected. However, some of the fruits get hidden under the litter layer thus preventing the frugivores from noticing it.

In situ seed germination for the selected species was significantly affected by forest types (characterised by open or closed canopy). The general trend was that seed germination percentage was more in case of seeds placed below litter. This could be attributed to the moisture content retained in the litter layer which helped the seeds to germinate.

Seedling recruitment varied among the forest types. Maximum number of recruitment was for *Holboellia latifolia* and was highest in the year 2008. *Entada phaseoloides* had the least recruitment amongst all the selected species. Seedling recruitment in *Evodia fraxinifolia* was also more in the year 2008 compared to 2007. This could be attributed to greater production of fruits and seeds.

Fragment type can influence seedling abundance (Benitez-Malvido 1998). The density of naturally occurring selected species seedlings was substantially higher in larger fragments than in smaller ones. The seedling densities of four species were higher in the periphery than in the interior of forest fragments. It appears that the factors reducing seedling germination and subsequent establishment are operating in the interior of fragments or otherwise, microenvironmental factors may be conducive for the growth of the four selected species along the periphery. The changes in canopy structure at the edge may also influence species composition of seedlings (Ward & Parker 1989). During seedling stage, mean density was strongly related to soil phosphorus for *E. fraxinifolia* and *H. latifolia*, indicating the importance of these factors in seedling establishment. Light intensity and soil pH were positively correlated with seedling density of *T. tiliifolia* species, indicating the role of light only during the juvenile phase of the species. The role of light in the establishment of seedlings of some woody species has been argued by Cai et al. (2007). On the other hand, it must be considered that soil pH interacts with the many biotic and abiotic soil factors which might affect the success of an ectomycorrhizal isolate to improve plant growth.

The correlation of seedling density with various microenvironmental factors provides important cue for managing the regeneration of these important plant species. However, liana species can withstand the change in drastic environmental conditions and able to flourish along the edges of the fragments. Laurance et al. (2000, 2001) had similar finding from Biological dynamic of forest fragments project study sites in Brazil Amazon, where the lianas are known to favour forest disturbance and tend to increase in density and diversity along forest edges. Smaller fragment on the contrary had more or less similar liana seedling density because of similar microenvironment in the interior and along the edges of the fragments.

Based on the findings of the study, following important conclusions were drawn:

1. Diversity of trees, lianas, shrubs, epiphytes and herbs decreased with increase in elevation. Therefore, the diversity was highest in the lower montane forests.
2. Considerable differences in floristic composition among the plant communities in three forest types indicate the important role of elevation and prevailing environmental conditions in determining species composition.
3. There was a decrease in meadow and open forest areas during the study period. This was attributed to reduction in extensive grazing by cattle and human disturbances especially in the high elevation forests.
4. Total number of forest fragments was more in 1999 which decreased considerably by 2008. Incision process was responsible for most fragmentation in KBR.
5. In KBR, reduction in the number of forests fragments during the study period was due to (i) complete disappearance of a certain number of forest fragments converting them into continuous forest blanks (ii) merger of forest fragments with main continuous forests through regeneration, and (iii) stringent protection measures enforced by state forest department, imparting of forest conservation knowledge to local people, and awareness activities undertaken by some local bodies and NGOs.
6. Species abundance was lower in forest fragments than that of continuous forests. The response of various tree species to fragmentation varied significantly.
7. The plant species had differential response to various microenvironmental factors and it varied during adult and regenerating phases.
8. Based on the response of species as analyzed through eight life history traits, the species were divided into two distinct guilds. *Holboellia latifolia*, *Evodia fraxinifolia* and *Toricellia tiliifolia* were classified as pioneer species, while *Entada phaseoloides* as intermediate to non-pioneer species.

Forest is the most important natural resource in the world fulfilling the diverse requirements of human population. Biodiversity of the forest ecosystems of the world is under threat due to forest fragmentation. Forest fragmentation refers to any process that leads to conversion of continuous forests into patches of forest fragments separated by non-forested land. Indiscriminate use of forest resources and over-exploitation of forest produces have caused serious damage to natural forest ecosystems and rich biodiversity of the Himalayan region. Loss of diversity and structural damage of forest communities limits plant recruitment and decreases the ecosystem productivity, affecting the overall ecosystem functioning (Symstad & Tilman 2001).

The forest ecosystems of Khangchendzonga Biosphere Reserve (KBR) (27°06 - 28°05′ N, 88°02′-88°47′ E) in the Eastern Himalayan state of Sikkim in north-eastern India is under pressure from the growing needs of human population and other anthropogenic/biotic pressures such as cattle grazing, landslide, forest fire etc., posing serious threats to the several taxonomically and ethnomedicinally important plant species. The present study was conducted in three forest types viz. Lower montane, Montane and Upper montane forests of KBR to understand the effect of forest fragmentation on plant diversity and regeneration. The main objectives of the present work were to: (1) To prepare an inventory of plant species including lianas and epiphytes, (2) To study the causes and pattern of forest fragmentation in KBR and (3) To study regeneration ecology of a few taxonomically and ethnomedicinally important tree and liana species.

The findings of the study are summarized below:

- The number of plant species in different lifeforms was highest in the Lower montane, than in the Montane and Upper montane forests. Seventy eight tree species with 47 genera and 30 families were recorded from the three forest types. Thirty eight shrub species belonging to 35 genera and 17 families were recorded. On the forest floor,

133 herb species belonging to 97 genera and 49 families were documented. Ninety two epiphyte species belonging to 57 genera and 31 families were recorded. Forty three liana species belonging to 37 genera and 28 families were recorded from the three forest types.

- α- diversity values and dominance indices for all the lifeforms decreased with increasing elevation except the dominance index for epiphytes, which remained same in the Montane and the Lower montane forests. However, it was the least in the Upper montane forest.
- β-diversity values between the Lower montane and Upper montane forests for all the lifeforms were highest and ranged between 0.71to 0.98. As expected, adjacent forests had lower β-diversity values than the fragments. Amongst the lianas, β-diversity value was lower compare to other lifeforms studied.
- Forest types differed significantly in plant species composition (tree, shrubs, herbs, epiphytes and lianas; Clark'*s R* statistic = 0.95, 0.63, 0.95, 0.47, 0.64 respectively and $P < 0.001$ for all).
- The density of tree decreased with increasing elevation ($F = 22.50$, $P < 0.001$). Tree density was highest in the Lower montane (463 stems ha^{-1}) forests followed by the Montane (239 stems ha^{-1}), and the Upper montane forests (256 stems ha^{-1}). Overall density of the shrubs differed significantly among the forest types ($F = 11.82$, $P < 0.001$). It was highest in Lower montane forest (319 stems ha^{-1}) and lowest in the Montane forest (101 stems ha^{-1}) and again it increased to 234 stems ha^{-1} in the Upper montane forest. On the other hand, density of herbaceous species did not differ significantly across the forest types ($F = 0.90$, $P = 0.44$). Highest density was in the Montane forests (711500 individual ha^{-1}), followed by the Upper montane and Lower montane forests (625000 individuals ha^{-1} & 609500 individuals ha^{-1} respectively). The density of epiphytes decreased significantly along the elevation ($F = 8.53$, $P =$

0.001). It was highest in the Lower montane forests (5200 individuals 20 tree^{-1}), followed by the Montane and the Upper montane forests (4830 & 2390 individuals 20 tree^{-1} respectively). Liana density also decreased with increasing elevation (F = 70.18, P < 0.001). It had 83 stems ha^{-1} in the Lower montane, 73 stems ha^{-1} in the Montane and 38 stems ha^{-1} in the Upper montane forest.

- With an increase in elevation, the trees, shrubs, herbs, epiphytes, and liana species-abundance curves exhibited higher dominance. The common tree species were *Alnus nepalensis, Castanopsis hystrix, Elaeocarpus lanceaefolius, Eurya acuminata* and *Rhus javanica* in the Lower montane forests. *Lithocarpus pachyphylla*, *Quercus lamellosa, Q. lineata* and *Rhododendron arboreum* in the Montane forests. *Abies densa* and *Rhododendron* spp., were common in the Upper montane forests. Amongst the shrubs, dominant species in the Lower montane forests were *Elsholtzia flava, Melastoma normale, Oxyspora paniculata, Rubus ellipticus, Rubus mollucanus,* and *Thysaenolaena maxima*. *Arundinaria maling, Deutzia compacta* and *Rosa sericea* were dominant in the Montane forests while, *Berberis* spp., *Rhododendron anthopogon, R. lepidotum* were dominant in the Upper montane forests. In the herbaceous flora, *Bidens pilosa, B. biternata, Carex filicina,* and *Elsholtzia blanda* were dominant in the Lower montane forests. *Fragaria nubicola*, *Persicaria runcinata, Phlomis bracteosa* were dominant in the Montane forests, while *Anaphalis triplinervis, Juncus* spp., and *Poa alpina* were dominant in the Upper montane forests. Amongst the epiphytes, Pteridophytic species were dominant in all the forest stands. The three dominant and codominant epiphyte species in the Lower montane forests were *Hoya linearis, Lepisorus nudus* and *Vittaria elongata*. In the Montane forests the three dominants species were, *V. elongata, L. nudus* and *Pleione humilis.* In the Upper montane forests dominant species were *Cystopteris sudetica*, *Onychium* spp., and *P. humilis*. Amongst the lianas, *Cissus repens*, *Clematis acuminata, Hydrangea*

anomala and *Parthenocissus himalayana* together were dominant in the Lower montane forests, while *Actinidia callosa*, *Holboellia latifolia*, *Rubus paniculatus*, and *Schisandra grandiflora* were dominant in the Montane forests. *A. callosa*, *Clematis montana*, *H. latifolia*, *Schisandra neglecta*, and *S. grandiflora* were dominant in the Upper montane forests.

- Epiphytes with Caespitose life form were highest. According to taxonomic classification, pteridophytic community was the highest, followed by herbaceous, shrubby and climbing epiphytes. Shrubby and climbing epiphytes contributed substantially to total epiphytes composition.
- Tall trees with larger girth supported greater number of epiphytes in all the forests.
- Tree basal area was highest in the Lower montane forest (92.57 m^2 ha^{-1}) compared to the Montane and the Upper montane forests (49.96 and 58.04 m^2 ha^{-1}, respectively). The basal area of lianas also followed a similar trend, i.e. 3.54, 2.25 and 0.13 m^2 ha^{-1} in the Lower montane, Montane and Upper montane forests, respectively.
- Microenvironmental variables viz. air temperature, soil temperature, soil moisture content, Phosphorus (P) and Nitrogen (N) varied significantly (ANOVA $P < 0.01$) among the three forest types. Air temperature, soil temperature, soil moisture content, soil Carbon (C) and N varied significantly (ANOVA $P < 0.05$) among the seasons.
- Although CCA explained overall poor species-environment relationship for all the vegetation components i.e. tree, shrub, herbs, epiphytes and liana across the forest types (explaining only about 20% of variabilities), the relationship with microenvironmental factors was significant (Monte Carlo test; $P < 0.009$).
- Soil N, pH, P and proxy variable ‘elevation’ were important determinants of tree species distribution across the forest types. On the other hand, soil N, C, pH, P and K were important determinants of shrub and herb species distribution across the forest types. While air temperature and the proxy variable ‘elevation’ were important

determinants of epiphyte species distribution across the forests. Liana species distribution across the forest types was determined by light, soil pH, N, P and the proxy variable 'elevation'.

- Forward stepwise multiple regression analysis revealed that, soil N was significant in influencing the overall distribution of tree and shrub species along the forest types (*P* = 0.000). While, soil P alone was significant in influencing herb species distribution along the three forests (*P* = 0.031). Elevation (*P* = 0.010) alone, soil P and light (*P* = 0.000) were significant in influencing the overall distribution of epiphyte and liana species respectively across the forest types. Soil pH and air temperature in the Lower montane, and C in the Montane and Upper montane forests were important in influencing tree density. In case of shrubs light and soil C in the Lower montane, and soil C alone in the Montane and Upper montane forests were important. For the herbs, soil pH and elevations in the Lower montane, and soil N, P concentrations in the Montane and soil P alone in the Upper montane forests were important. Light in the Lower montane, elevation in the Montane and RH in the Upper montane forests were important in influencing epiphytes diversity. Light in the Lower montane, soil P concentration in the Montane, and both light and soil P in the Upper montane forests were important determinants of liana abundance.
- Extent of forest fragmentation in KBR was studied by analyzing the imageries pertaining to the three time intervals, i.e. 1999, 2002 and 2008. The total number as well as area under fragments represented a declining trend during the period, 1999 to 2008. Considerable changes were found in the distribution, number and size of forest fragments during the decade.
- The number of fragments in different size classes decreased sharply from 875 in 1999 to 533 during 2002 and again it increased to 615 during 2008. During the entire study

period, the average annual fragmentation rate was 0.7 ha $year^{-1}$, equivalent to 0.007%.

- Mean fragment size decreased from 4.4 ha in 1999 to 3.9 ha in 2008. This decline in mean fragment size was associated with decrease in fragment density and a substantial reduction in the size of the largest forest fragments during the study period.
- Important causes of forests fragmentation in KBR were agriculture, grazing, NTFPs cultivation/extraction, timber/poles, trekking routes, settlement, tourism, road, wind-throw, landslide, snow avalanche, and wild fire. The anthropogenic disturbances from agriculture, NTFPs cultivation/extraction, and agriculture decreased sharply in 2008.
- Incision was the dominant process of fragmentation as revealed from the analysis of fragmentation process in 25 identified fragments.
- The species diversity indices of 25 forest fragments were lower than those of adjacent continuous forests.
- The α-diversity for trees was significantly higher in the Montane forest fragments than in the other two forests.
- The β-diversity was high between Upper and Lower montane forest fragments (0.91), and Lower montane and Montane (0.81) forests. The Montane and Upper montane forest fragments had the lowest β-diversity value of 0.62. Within Montane forest, β-diversity between the largest fragment (72.2 ha) and smallest fragment (0.1 ha) was 0.50 and in the Upper montane forest β- diversity between the largest fragment (72.2 ha) and smallest one (1.02 ha) was 0.80.
- Air and soil temperature, light intensity decreased with increasing distance towards the forest interior, while relative humidity showed an increasing tendency.
- TWINSPAN analysis classified 71 tree species from 25 fragments into eight specific tree communities (C1-C8).

- TWINSPAN ordination of 25 forest fragments clearly showed the separation of fragments according to forest types.
- Tree species such as *Castanopsis hystrix, Alnus nepalensis, Buddleia colvilei, Ficus semicordata, Eurya acuminata* were found in the medium size fragments in the Lower montane forests. *Rhododendron hodgsonii, R. thomsonii,* was mostly found in smaller size fragments in the Upper montane forests. Larger fragments in Upper montane forests contained all the species from smaller and medium size fragments.
- The distribution of the tree species in the ordination space was performed using DECORANA according to tree community types and their area of occurrence in forest fragments in the three forests. The eigenvalue of 0.87 and 0.63 also confirmed that there was a good dispersion of tree species along the first and second ordination axes.
- Most of the DECORANA clusters matched with the TWINSPAN classification, indicating that classification and ordination of tree species data were complementary to each other.
- *T. tiliifolia* showed a higher number of individuals in the adult stage and lesser number of individuals during seedling stage, in the Lower montane forests. In case of *E. fraxinifolia,* density of individuals was more in the seedling stage in Lower montane and Montane forests, whereas in Upper montane forests no individual of this species was encountered. Individuals of *H. latifolia* were also more during seedling stage in all the three forest types. *E. phaseoloides* indicated a growing population in the Lower montane forests with maximum number of individuals in the seedling stage, and all other stages were less prominent.
- *T. tiliifolia* had fair regeneration and *E. fraxinifolia* had good regeneration among the tree species. Among the lianas, both the species (*H. latifolia* and *E. phaseoloides*) had

good regeneration, however the density of *E. phaseoloides* during adult stage was less.

- Seedling populations in all the forests showed marked difference between wet (May-July) and dry (November-March) seasons with more number of tree and liana species in the seedling stage in the wet season. No apparent difference was found in the sapling populations between dry and wet seasons.
- Stump size significantly affected coppice regeneration in *T. tiliifolia,* and *H. latifolia* ($P < 0.001$). Both the tree and liana species did not show variation in sprouting intensity. However, the liana species tend to initiate sprouting at lower girth classes than the tree species. Regression models showing the relationship of stump size with number of sprouts yielded a significant polynomial relationship for above two species ($P = 0.001$).
- *Evodia fraxinifolia* strictly followed a sequential order of one phenological event followed by the other with least overlapping between any two events. In *T. tiliifolia* active vegetative growth, bud formation and flowering initiated more or less at the same phase, particularly during late winter, leaf shedding and seed dispersal also overlapped with each other. In *H. latifolia* leaf flushing occurred during winter season while in *E. phaseoloides* it was prominent during the months of April, May and June till early season of monsoon. Flowering, fruit setting and seed dispersal followed more or less similar sequence in both the species of lianas.
- The number of flower and fruit produced in *Toricellia tiliifolia* was significantly higher ($P < 0.001$) in the year 2008 than in 2006 and 2007. Flower and fruit production varied significantly across DBH classes ($P < 0.001$) and it was higher in higher DBH classes. Flower and fruit production varied significantly across the DBH and forest types ($P < 0.001$). Abortion, flower and fruit production did not show significant variation across the years.

- Size of the soil seed bank for *T. tiliifolia*, *E. fraxinifolia* and *H. latifolia* varied significantly ($P < 0.001$) with girth size. The size of the soil seed bank of *T. tiliifolia* was significantly higher in both the years (2007 and 2008) in the Lower montane forests.
- Seed viability of the selected species decreased consistently across a temporal scale. *H. latifolia*, recorded viability period of almost two years but decreased considerably with time. *E. phaseoloides* and *E. fraxinifolia* recorded viability period of 15 months but that also decreased considerably with time.
- *In situ* seed germination of the liana and tree species varied among different forest types and treatments significantly in relation to litter (*H. latifolia* and *E. fraxinifolia* $P < 0.001$, and in *E. phaseoloides* $P < 0.05$).
- Seedling recruitment for *H. latifolia* was highest in the Montane forests, while, that of *E. fraxinifolia*, it was in the Lower montane forests.
- High seedling mortality of *H. latifolia* occurred during the three months of germination (March-June). Seedling survivorship curves for *E. fraxinifolia* and *E. phaseoloides* showed a sharp reduction in the number of individuals after 3 and 6 months period respectively and continued till the seedlings were one year old, after which the seedling population stabilized.
- The seedling densities of four species were higher in the periphery than in the interior of forest fragments.
- During seedling stage, density was strongly related to soil P for *E. fraxinifolia* and *H. latifolia*, indicating the importance of this factor in seedling establishment. Light intensity and soil pH were positively correlated with seedling density of *T. tiliifolia* species, indicating the role of light only during the juvenile phase of the species.
- Smaller fragment had more or less similar liana density because of similar microenvironment in the interior and along the edges of the fragments.

- Based on the response of species as analyzed through eight life history traits, the species were divided into two distinct guilds. *Holboellia latifolia*, *Evodia fraxinifolia* and *Toricellia tiliifolia* were classified as pioneer species, while *Entada phaseoloides* as intermediate to non-pioneer species.

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