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FIRE REGIMES AND THEIR ECOLOGICAL EFFECTS IN
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**FIRE REGIMES AND THEIR ECOLOGICAL EFFECTS IN SEASONALLY DRY
TROPICAL ECOSYSTEMS IN THE WESTERN GHATS, INDIA**

By

Narendran Kodandapani

A DISSERTATION

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Abstract

FIRE REGIMES AND THEIR ECOLOGICAL EFFECTS IN SEASONALLY DRY TROPICAL ECOSYSTEMS IN THE WESTERN GHATS, INDIA

By

Narendran Kodandapani

The tropics are currently witnessing significant deforestation, recent studies from the Global forest resources assessment of FAO suggests, that of the mean annual global deforestation of 16.1 million hectares for the time period 1990-2000, more than 90% of deforestation was from the tropics. In India greater than 75% of the original forest area has been deforested and in the Western Ghats only 6.8% of the original vegetation remains. A number of disturbances in these ecosystems such as logging, grazing, and forest fires are common to the Western Ghats. This dissertation investigates forest fires which are annual disturbance events in these forests, compares and contrasts fire regimes across the different vegetation types, and also examines their ecological effects.

I combined information from remote sensing imagery and a meticulous ground mapping effort of all fires in the Mudumalai Wildlife Sanctuary (MWLS) in the Western Ghats of India over the past 14 years (1989 – 2002). These spatial data on fire occurrence were integrated with maps of the specific vegetation types found in the MWLS to examine fire conditions in each class of vegetation. For the MWLS, all forest types were found to have an average fire-return interval (FRI) of <7 years and the sanctuary as a whole had a FRI of 2.7 years. Compared to a similar 13-year MWLS fire dataset from the 1910s, this represents an almost three-fold increase in fire frequency over the last 80 years. I estimate

a FRI of roughly 5 years for both the larger Nilgiri Biosphere Reserve and the entire Western Ghats. The estimated fire frequencies for the Western Ghats forests outside of protected reserves are considered very conservative given other recent reports.

I determined the spatial relationship between park boundary and mean fire-return interval from satellite data obtained between (1996 and 2005). Different models explain significant variations in the FRI. The linear model is the most realistic and explains 30% of the variation in FRI as a function of distance from park boundary. I implemented a logistic regression model to determine other significant determinants of forest fires in the landscape. Elevation, mean annual rainfall, and forest fractional coverage emerge as significant predictors of fire in the landscape. Results from variogram models indicated the increased importance of the spatial structure in predicting fire occurrence in the landscape.

The study of fires between 1996 and 2005 shows that the mean fire sizes in tropical dry deciduous forests were four fold larger than mean fire sizes in tropical moist deciduous forests and two fold larger than tropical dry thorn forests. Maximum fire size in the tropical dry deciduous forest was 20 fold larger than moist deciduous forest while it was only 6 fold larger than the tropical dry thorn forest. Fires have resulted in significant mortality on small (0 – 5 cm dbh) size class in the tropical dry deciduous and tropical moist deciduous forests. The differences in fire regimes have resulted in significant effects on the regeneration, structure, composition and diversity in these vegetation types.

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My dissertation research in the Western Ghats has been part of a long journey that I undertook many years ago as an ecologist/biogeographer. The unique landscape aspects of the Western Ghats in terms of its biogeography, its environmental history, and biodiversity were inspiring and motivated me to embark on this research.

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Graduate school at Michigan State University was new and challenging during my first few days, Prof. Joseph Messina gave me all the necessary encouragement to adapt to the new environment. His continued support during the completion of my dissertation was invaluable.

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Chapter 1

The Western Ghats: Climate, Biogeography, Human Presence, and Primer in Fire Ecology

Introduction:

The tropics are currently witnessing significant deforestation, recent studies from the Global Forest Resources assessment of FAO suggests that, of the mean annual global deforestation of 16.1 million hectares for the time period 1990-2000, more than 90% of deforestation was from the tropics (Lambin et al. 2003). Apart from this asymmetry in deforestation across the globe, there is considerable variation in the relative rates of logging and deforestation between the continents. Asia has the highest relative rates of both deforestation and logging among all the continents (Laurance 1999). In India, greater than 75% of the original forest area has been deforested and in the Western Ghats only 6.8% of the original vegetation remains (Myers 2000). Simultaneously, the tropical forests are extremely ancient, diverse, and ecologically complex occupying only 7% of the Earth's landmass, they possess greater than 50% of the World's biological wealth (Wilson 1999). Part of the reasons behind this deforestation crisis in the tropics has been the explosion in human numbers, the Western Ghats in India has been described as the hottest of biological hotspots, however it is also the hotspot with the highest population densities (Cincotta et al. 2000).

About 40% of the Earth's tropical and subtropical landmass is dominated by forest. A quarter of this area is under tropical rain forests, while the remaining three-quarters are under dry and moist forests (Murphy and Lugo 1986). Of the 328 million hectares

of land area in India, approximately 20% is under forests of different percentages of canopy cover (Forest Survey of India 2003). The original extent of primary vegetation in the combined areas of the Western Ghats and Sri Lanka was about 182 500 km² of this only about 12500 km² remains (Myers et al. 2000).

Statement of Purpose

Many parts of Africa and Asia have been transformed by human activities (D'Antonio and Vitousek 1992). These ecosystems have been transformed chiefly by the use of slash and burn agriculture. In India, there has been a longstanding relationship between the use of the forest and human communities living on the fringes of the forest (Gadgil 1993; Chandran 1997). Forests provide a diversity of products and services to people living near and away from forests (Kodandapani et al. 2001).

Disturbances such as logging, forest fires, and grazing are common to most forests in the Western Ghats (Silori and Mishra 2001; Kodandapani et al. 2004; Menon and Bawa 1998). In the tropics and in the Western Ghats in India the study of forest fires and their ecological effects are complex due to the use of forests in the past and the transformation of forests through disturbances (Gadgil 1993; Andersen 2003). Forest fires are landscape scale disturbance events in the Western Ghats in India. These fires are an almost annual occurrence in the deciduous ecosystems (Kodandapani 2004).

Many of these forest fires are a direct consequence of humans, who reside on the fringes of these ecosystems and venture into these ecosystems to collect a number of forest products, such as fruits, tubers, fuelwood, and leaf manure (Kodandapani 2001; Rai 2003). Forest fires in the high elevation evergreen forests and the evergreen forests distributed at lower elevations in the Western Ghats are rare, although dry season fires have been reported in the low elevation evergreen ecosystems in some

parts of the Western Ghats (Daniels et al. 1995). The spatial and temporal patterns of these forest fires have not been well established. Fire regimes such as the size of forest fires, the interval between fires, intensity and severity of fires play an important role in shaping landscape pattern, processes, and functional linkages in ecosystems (Lyon et al. 2000).

A concise and clear conceptual framework to monitor and assess forest fires has been a limitation in the study of forest fires and their ecological effects in the Western Ghats. The forest fire regime provides one such conceptual framework to study fires, the fire regime describes the spatial extent of fires, the temporal variability of fires, the intensity and magnitude of fires, the season of fires, and the ecological effects of the fires (Smith 2000; Agee 2000; Kilgore and Taylor 1981; Heinselman 1978). However, hardly any studies have been conducted to quantify and assess various components of these disturbances regimes and their effects on the Western Ghats.

Questions:

1. What is the fire regime of the Nilgiri Landscape, specifically how do they compare and contrast among the tropical dry deciduous forest, tropical moist deciduous forest, and tropical dry thorn forests?
2. What are the landscape variables that influence spatial and temporal patterns of fire occurrence and spread in the Nilgiri landscape?
3. What are the effects of fire regimes on the structure, composition, diversity, and regeneration of the different forest types?

Review of literature

A number of studies have been conducted in the tropics to examine various aspects of forest fires. Uhl and Kauffman (1990) examined the effects of forest fires in the Amazon, they examined flame characteristics and bark thickness in relation to survival after fire, and hence differences in mortality among tree species. Cochrane and Schultz (1999) examined mortality due to fire among woody plants and its effect on the structure, composition, and biomass of forests in Para, Brazil. Studies in the Amazon have shown the importance of positive feedbacks between fire susceptibility and deforestation, these studies have also demonstrated the increased fire severity due to subsequent fires in evergreen forests of the Amazon (Cochrane et al. 1999; Cochrane 2001). Barlow et al. (2003) have examined tree traits such as bark thickness, presence of buttresses, bark roughness and the presence of latex, and resin, and their relationship to mortality due to fire in the Amazon. Barlow and Peres (2003) have also examined the effects of fires on wildlife in the Amazon. Studies in a seasonally dry forest in Bolivia have shown the relative importance of tree bark thickness and thermal properties of bark in protecting trees from cambial kill (Pinard and Huffman 1997). Ross et al. (2002) conducted studies in the Australian tropics applying remote sensing data to assess the effects of fire size and time since fire on different aspects of forest composition. Slik et al. (2002) compared mortality in logged and burnt forests of SE Asia. Recent studies in a 50 ha plot in the Western Ghats has revealed that fire is one of the primary causes of mortality especially among lower size classes of woody plants between 0 and 10 cm dbh (Sukumar et al. 2004; John 2000). Studies in the African savanna woodland ecosystem have shown the synergistic interactions between fire, elephants, and humans in maintaining a dynamic equilibrium in the ecosystem (Dublin 1995; Norton-Griffiths 1979).

A number of methods have been adopted by fire ecologists to assess and monitor the effects of fire in ecosystems (Cochrane and Souza 1998). A number of sensors with different spatial, temporal, radiometric, and spectral resolutions mounted on satellites have been incorporated in the delineation of burnt areas in landscapes across the globe (Cochrane et al. 1999; Cochrane 2003). Landsat TM and ETM+ data have been extensively applied to the monitoring of burnt areas in landscapes across the globe. In the tropics, Cochrane and Souza (1998) have applied Landsat data to generate algorithms to separate burnt and unburnt forests, further they have been successful in separating recently burnt areas and areas burnt about 2 years after the fire event with reasonable accuracy. These methods basically use spectral endmembers of vegetation, shade, and non-photosynthetic vegetation to separate the burnt and unburnt areas. In another study, data from Landsat TM and AVHRR were applied to calculate the NDVI and from the NDVI data, the burned and unburned areas were delineated. The method assumes that there is a significant decline in the NDVI value of a pixel before and after a fire event (Domenikiotis et al. 2002).

Fire occurrence and spread within a forest depends on the fire environment; climatic conditions, the fuel conditions, and topographic conditions (Pyne et al. 1996; Agee 2000). In certain regions, such as the Sierras, the aspect and elevation had an important effect on the occurrence and size of fires (Caprio 1999). The studies found a significantly greater number of fires on South aspects compared to North aspects, he also found that the annual area burned was larger in South facing aspects, however in terms of size the largest fires occurred in the North aspects. Similar results were also obtained from a study in France, wherein South, southwest, and southeast facing slopes had a higher proportion of burnt areas when compared with slopes facing

North (Mouillot et al. 2003). Periods of below average rainfall in many studies have been implicated as one of the important reasons for susceptibility of forest to fires (Siebert et al. 2001; Roberts 2000; Woods 1989).

The Western Ghats: Physiography

The Western Ghats are a long chain of mountains all along the west coast of southern peninsular India stretching for 1600 km (WWF 1998). The coordinates of the hill range extends from 8° N to 21° N latitudes and between 73° E and 77° E longitudes (Fig 1-1). The mountain range has at least two conspicuous gaps along its chain, one at Palghat and the other at Shencottah (Radhakrishna 2001). The mountains extend over the states of Gujarat, Maharashtra, Goa, Karnataka, Tamil Nadu, and Kerala. Important mountain peaks include the Ballairayan durga, Brahmagiri, Banasuram hills, Nilgiri hills, Dodabetta, Anaimalai, Palni hills, Varushanad hills, and the Mahendragiri hills. The Nilgiri hills are a very important land mass and represents the meeting point of the Western and the Eastern Ghats. The Nilgiri landscape is the focus of this study and is situated in the Nilgiri Biosphere Reserve (NBR) (Prabhakar 1994; Daniels 1996). A number of rivers originate in the Western Ghats, while some like the Cauvery, the Godavari, Krishna, and Tambirparani are East flowing others such as the Kali, Netravati, Sharavathy, Periyar, and Pamba are West flowing (Raman 2000).

Geology and Palaeoecology

The geological history has left an indelible mark on the flora and fauna of the Indian subcontinent. The separation and movement of the Indian subcontinent from the Gondwana landmass towards the Asian plate took place sometime during the Jurassic,

following this in the Cretaceous the next episode of separation from Africa and Madagascar occurred (Subrahmanya 2001). Subsequent to this, at the boundary to the Cretaceous and the Tertiary, the next mega event in the geologic history of the Indian subcontinent occurred. At this point of time approximately 65 million years ago huge volcanic eruptions occurred, referred to as the Deccan traps, these continental flows extended over a million km² and averaged 1000 m in thickness (Radhakrishna 2001). The next stages in the evolution of the Western Ghats included the sinking of the sea floor, domal uplift caused by the thermal expansion of the mass over hotspots of volcanic activity, rifting and collapse of the western surface leading to the formation of an escarpment like structure in the Western Ghats (Radhakrishna 2001). Still others suggest an alternate hypothesis, for example Pascal (1988) suggests that the Western Ghats were a result of the collision of the Indian subcontinent with the Asian plate and subsequent tectonic activity resulted in the formation of the Western Ghats.

The rocks and soils of the Western Ghats are related to the dynamic geologic past of this region. North of latitude 16 °N, the Deccan traps and hence volcanic rocks and soils overlie the parent material of Archaean rocks. South of this latitude, the Dharwar system of ancient metamorphic rocks prevail until 13 °N latitude and further south the pre-Cambrian crystalline rocks prevail (Raman 2000). The major soils include red-colored Chromic and Calcic Luvisols (Radhakrishna 2001).

The time period since the migration of the Indian landmass and its subsequent collision with the Asian plate witnessed the appearance and disappearance of various organisms through geologic time in the Asian subcontinent. Although much of the evidence regarding the past distribution of organisms is scanty and present

occasionally in the form of fossils in certain parts of southern India, more recent palaeobotanical studies by Sukumar et al. (1995) have provided more conclusive evidence from radio carbon, oxygen isotope studies, and the identification of pollen of species present in peat bogs in certain parts of the Nilgiris in India. These studies have reconstructed the past climate in the Nilgiris over the past 30 ,000 years and also related them to events such as the glaciation and retreading of glaciers during this time period. Further in sync with these events they have also reconstructed the distribution of forests and grasslands and evidence on the species comes from the identification of pollen of species and photosynthetic pathways of species during these time periods (Sukumar et al. 1995; Rajagopalan et al. 1997).

Biogeography

A number of theories have been postulated regarding the biogeography of the Western Ghats, one hypothesis suggests that organisms evolved in the Indo Malayan region and then dispersed into peninsular India and the Western Ghats via the Assam gateway and hence the Western Ghats forms the limits of species distributions within this larger habitat (Mani 1974). Still others suggests that the species distributions of the Western Ghats reveal its geologic history, its attachment to the Gondwana landmass, subsequent migration perhaps eliminated certain temperate species, many species of fauna could have evolved in the humid tropics and many of the fishes, amphibians and reptiles could also have relationship with the Gondwana landmass (Mani 1974). Still others suggest a continuous distribution of organisms from Indo Malay since a number of organisms are similar in North-East India as well as the southern Western Ghats. However, in between these two nodes, species are extremely different, which some suggest could be due to the disruptive forces of

anthropogenic activities in the regions between these two nodes, while others suggest, events such as climate change have resulted in the isolation of the Western Ghats from the rest of the Indo Malayan region (Mani 1974).

The biological diversity of the country is represented by a wide spectrum of biogeographic zones which includes the Trans Himalayan, the Himalayan, the desert, the semi-arid, the Western Ghats, the Deccan peninsula, the North East India, Gangetic plains, and the islands and coastal ecosystems (Rodgers and Panwar 1988). The Western Ghats has been designated as a unique biogeographic region within India (Mani 1974; Rodgers and Panwar 1988). Along with Sri Lanka it has been designated as one of the 25 hotspots of biodiversity (Myers et al. 2000).

Relationship between Climate and Vegetation

Three important climatic gradients have resulted in the current distribution pattern of vegetation and species in the Western Ghats. The North-South gradient in the number of dry months, the southern end has a short dry season of two to five months, whereas the northern parts of the Western Ghats have a longer dry season of five to eight months. Although the number of dry months increases South to North, the total annual rainfall at certain localities are similar irrespective of location. The next important gradient is the West-East gradient, caused by the orographic effects induced by the Western Ghats. Rainfall along the coast could range from 3500–4000 mm, at higher elevations in the Western Ghats annual rainfall could be as high as 7000 mm and on the leeward side of the Ghats the annual rainfall could be as low as 500 mm (Raman 2000). The third most important gradient is the altitudinal gradient in temperature. The highest point in the Western Ghats is at Anaimudi peak at 2695 m

asl. The mean temperature for the coldest month ranges from 25° C at sea level to 11° C at an elevation of 2400 m asl (Pascal 2001). These gradients have resulted in a rich diversity of life forms in a number of taxa.

The Vegetation

The Western Ghats can be divided into four phytogeographical regions, the Western Ghats from the River Tapti to Goa (between 15° and 21° N latitudes). The relief in this part of the Western Ghats exhibits tremendous variability, rising to almost 1000 m from sea level in about 2 to 3 km. Along the foothills on the western side of this region, the scrub and dry semi-deciduous forests are distributed at elevations of 200 to 500 m and rainfall of 37 to 61 cm. The dry deciduous forest occurs on the eastern side of the Western Ghats, at elevations between 500 to 1166 m, with rainfall of 50 to 152 cm; the moist deciduous forest occurs on the windward side at elevations between 500 and 833 m, with annual rainfall of 100 to 200 cm. The evergreen forests that occur here are not typical of the rest of the Western Ghats, although the rainfall is about 625 to 750 cm, it has been classified as the montane subtropical evergreen forest. Here trees are structurally and floristically different from the evergreen forests found further South of the Western Ghats (Pascal 2001).

The second part of the Western Ghats along this latitudinal continuum lies between 15° and 12° N. The main forest types observed in this part include the dry thorn forest, the moist deciduous forest, and the wet evergreen forests. The wet evergreen forests found in this part of the Ghats are three storied and some species could be as tall as 35-40 m. The deciduous forest lies between 666 m and 1000 m, trees here are much smaller, about 12-24 cm tall. The eastern slopes reveal a simpler floristic

composition and have a combination of dry deciduous and dry thorn forests (Subramanyam and Nayar 2001).

The third part of the Western Ghats is the Nilgiris and extends between latitudes 10° and 12° N. It forms a compact plateau and serves as the landmass connecting the Western and Eastern Ghats. The high relief has resulted in the formation of two very interesting vegetation forms, the high elevation evergreen forests and the high elevation grassland ecosystem (Sukumar et al. 1995).

The last part includes the rest of the hill chain from 8° to 12° N. This region harbors wet evergreen forests on the windward side between 500 and 2500 m, at higher elevations the high elevation evergreen forests are distributed. Annual rainfall ranges from 250 to 500 cm. The moist deciduous forest is found between 500 and 900 m and annual rainfall ranging from 240 cm to 350 cm. On the leeward side on the mountain, the dry thorn forest and the dry deciduous forest are distributed (Subramanyam and Nayar 2001).

Unique vegetation and endemism

The species diversity of the Western Ghats are poor in comparison to the other tropical forests. On a two hectare plot in the Western Ghats, the diversity of species > 10 cm dbh was about 60, whereas in certain other tropical forests in Malaysia, and the Amazon, the species diversity was much higher, for example in Malaysia it was 227 species/2 ha. However, the Western Ghats differs from these other tropical forests in the number endemic species found in the forests. More than 4000 plant species are found in the Western Ghats. Of this, approximately 1500 are endemic. A

number of genera are endemic to the Western Ghats. Genera that are highly speciose, containing a number of endemics include the *Dipterocarpus* (92% of 13 species), the cane palms *Calamus* (92% of 25 species), and balsams *Impatiens* (86% of nearly 90 species), *Ebenaceae* (60% of 20 species), and leguminous trees (100% of 12 species) (Raman 2000). There is a general North South gradient in endemism, the percentage of endemic species increases from North to South of the Western Ghats. Thus of 352 endemic evergreen arborescent species found in the Western Ghats, almost 70% are found between latitudes 8° and 10° N, while only 10% or less are found at latitudes > 15 °N (Ramesh 2001).

Fauna

The animal diversity of the Western Ghats is rich especially among the lower vertebrate species. Nearly 42% of the 245 fish species found in the Western Ghats are endemic to the region (Kumar et al. 1999). 75% of the 120 amphibian species found in the Western Ghats are endemic to the region (Johnsing 2001). Two groups of amphibians especially important for their endemism are the limbless caecilians and the rhacophorid tree frogs (Raman 2000). Approximately 40% of reptiles found in India occur in the Western Ghats. Reptilian groups with high endemism include the uroelptid snakes distributed mainly in the Western Ghats and Sri Lanka. Mammalian diversity of the Western Ghats is less important in comparison with other taxa, about 125 species of the 400 found in India occur in the Western Ghats. Twelve of these species including the Nilgiri Tahr, lion tailed macaque and two genera the Latidens (bats) and Platacanthomys (rodents) are endemic to the Western Ghats.

Human Presence and Deforestation

A number of estimates have been made of the rates of deforestation in the Western Ghats and fragmentation of these forests. Gadgil and Meher-Homji (1986) estimated that the potential area under evergreen forest had been reduced from 62,000 km² to anywhere between 5,288 (8.5%) km² and 21,515 (34.7%) km² in the mid 1980s. Deforestation has been particularly intensive in the southern Western Ghats which lost a quarter of its forest cover between 1973 and 1995 (Jha et al. 2000). National statistics on deforestation compiled by the Ministry of Environment and Forests of India showed a decline in forest cover in the 5 Western Ghat states (Maharashtra, Goa, Karnataka, Tamil and Tamil Nadu) by 20% between 1972 and 1982 and a continuing decline by another 4.32% between 1982 and 1990.

A more recent estimate by Myers (2000) provides a combined estimate of deforestation of primary vegetation in the Western Ghats and Sri Lanka. The study puts the current estimate of these primary forests at 12,450 km², which is 6.8% of the original extent of 182,500 km². This has resulted in the fragmentation of forests, with consequent effects on biodiversity. Menon and Bawa (1997) conducted a study on the rates of forest fragmentation in the Western Ghats between 1920 and 1970 and they found a 40% decline in the natural forest cover in the intervening period. The mean patch size decreased by 83% and the number of patches increased four fold.

Human Occupation of the Western Ghats

Human presence in the Western Ghats goes back to the Palaeolithic, over 12,000 yr BP. During the Mesolithic, about 5000 yr BP, transition to cultivation could have originated in the region. During the Neolithic, c. 3500 yr BP, expansion of cultivation

and pastoralism and conversion of primary forests to more secondary formations could have started, this is evident from palynological studies (Chandran 1997; Hockings 1996). Although human presence in the Western Ghats dates back many millennia, largescale destruction of forests began in the 19th century (Gadgil 1990; Raman 2000). Before the 19th century, much of the forest exploitation resulted from the extraction of non timber forest products and slash and burn agriculture. This could have resulted in small gaps within the forests, which closed after the cessation of activities by communities living within these forests. However, largescale changes in the Western Ghats began with the arrival of the British in the early 19th century. A number of forest patches were systematically converted into plantations of teak (*Tectona grandis*), tea (*Camellia sinensis*), coffee (*Coffea arabica*), rubber(*Hevea braziliensis*), *Eucalyptus sp*, wattle (*Acacia meansii*). Beside the conversion of forests into plantations, timber extractions were also carried out within much of the Western Ghats (Raman 2000). The moist and humid forests of the Western Ghats were inaccessible until the British built many roads to explore and exploit the resources within these forests. The incidence of malaria in these forests ensured limited use by indigenous communities living in these forests for many thousands of years (Gadgil and Guha 1993; Chandran 1997). Post independence deforestation has accelerated within the Western Ghats in the form of hydroelectric projects, further expansion of area under tea and coffee plantations, conversion to agriculture land and other forest plantations and a number of forest based industries (Gadgil 1990).

The Nilgiri Biosphere Reserve

My research has been mainly conducted in the Nilgiri Biosphere Reserve (NBR). The NBR is representative of the Western Ghats in terms of climate, biogeography, and

land use land cover changes. The Nilgiri Biosphere Reserve was created in the Western Ghats to conserve biodiversity while simultaneously reconciling human pressures on forests by people living within and around the region. The NBR is located between 10° 45' and 12° 15' N latitudes and 76° and 77° E longitudes. It stretches from Coorg-Wynaad plateau in the North and West to Attapadi Bolampatti Hills in the South, and East into the Talamalai-Hasanur plateau of the Eastern Ghats (Daniels 1996). The study was located in three wildlife sanctuaries in the states of Karnataka, Tamil Nadu, and Kerala. I based my study on results obtained from the three contiguous forests located in these three states. The combined areas of the Wynaad wildlife sanctuary, the Bandipur wildlife sanctuary, and the Mudumalai wildlife sanctuaries formed the major part of the study area and I refer to it as the Nilgiri landscape in the rest of this dissertation.

Ecological history of Mudumalai wildlife sanctuary

The Mudumalai Wildlife Sanctuary, where much of this research has been carried out, lies in the Sigur Wynaad plateau and adjoining this unit is the Nilgiri plateau. Both these landforms were sparsely populated until the early 19th century, however, there have been reports of control over these areas by neighboring kingdoms that ruled before the 19th century (John 2000). The dense forest cover of these areas, although majestic in beauty were impenetrable due to the occurrence of malaria (Raman 2000). The British arrived in the Nilgiris during 1820-1830. They set up elaborate plans to harvest a number of timber species, which included species such as *Tectona grandis*, *Pterocarpus marsupium*, *Terminalia crenulata*, *Lagestroemia microcarpa*, and *Dalbergia latifolia* (John 2000).

The wildlife sanctuary was declared a reserved forest way back in 1889, with detailed plans for the extraction of timber from these forests. In 1940, the forest was formally declared the Mudumalai wildlife sanctuary and all hunting was prohibited within the forest. Events in the world at large had an impact on the wildlife sanctuary between 1940 and 1945, when parts of the sanctuary were used as battle preparation grounds for troops to be deployed in Burma to fight against the Axis forces there. Finally, in 1958, the sanctuary was expanded to encompass 318.7 km² and set aside with wildlife management as the sole objective (John 2000).

A Primer in fire ecology

Ecologists and biologists have studied disturbances and their effects in ecosystems by applying novel techniques and analysis schemes. Fire ecologists have focused their research efforts on fire in ecosystems and have used a number of words and terms specific to the studies of fire and its effects in the ecosystem. The following is a brief primer to any reader who may be unfamiliar with these terms and hence serves as a reference to the terms used in the remaining chapters.

The study of fire and its effects requires an understanding of the propagation of heat energy in the fire environment, the efficiency of this transfer depends on the properties of materials that are burned, the weather at the time of the fire, and features of the topography in the fire environment. The effects of the fire will depend on the interval between fires, the areal extent of the fires, and the intensity of the fires. The fire regime describe various components of the fire in the landscape, it is the general pattern of a fire in terms of its frequency, when forests are repeatedly subjected to fires a few well adapted species proliferate in the forest; the season of

fire occurrence, this is important as it translates into differences in the magnitude of the fire effect on organisms, growing season fires in a forest are more detrimental than dormant season fires; the size of the fire when related to the regeneration of tree species, provides information regarding the composition of a forest post disturbance; besides this the fire regime also describes prominent, immediate effects of fire in a vegetation type or ecosystem. Apart from these major components of the fire regime, the fire in a forest also depends on the fuel loads and changes associated with it such as the steady state fuel load and the productivity and decomposition within the forest; the composition of the fuel loads, which includes the types of fuels such as the grasses, leaf litter; the arrangement of the fuels in the fuel bed; the fuel moisture content; the synergistic interactions between fires and the contribution to the fuel loads of subsequent fires. All these components of the fire environment and its effects can be summed up as the fire regime.

The fire-return interval can be described as the time required to burn an area equal to the total forest area, with the caveat that certain areas may burn more than once, while certain areas may never burn. A related term is the fire frequency, it is the number of times in a given time period that an area has experienced a fire event. There are a number of ways that the frequency is estimated in a forest, it can be estimated for a unit area in the landscape, called the area frequency; it can also be estimated as a point frequency from a number of fire scarred trees in the landscape.

Fuels can be classified based on the size of the fuel. It is common to refer to these fuel size classes as time lags. The time lag refers to the amount of time required for a given fuel size to lose 63% of the difference in moisture and come into equilibrium

with the ambient environment. It can also be related to the surface area volume ratio of the fuel particle. Hence small size fuel particles have a higher ratio compared to larger size particles, this translates into differences in flammability of the fuels.

Apart from the surface area volume ratio, another important measure related to the spread of fire in a fuel bed is the packing ratio, this is a dimensionless ratio of the mass of the fuel particles and the volume of the fuel bed. It can be calculated if the density of the fuel particle is known. Along a fire spread gradient of high to low, grasses have a ratio of 0.001, litter 0.01, and fuel sticks 0.1. The ratio provides an indication of the aeration available between the fuel particles, which is critical for the combustion process. Hence grasses have ideal ratios as they are neither too compact nor too aerated, this ideal packing ratio of grasses leads to the rapid spread of fires in this fuel type.

The weather at the time of fire is important for the ignition process as well as the sustenance of the fire in the forest. High fuel temperatures combined with low fuel moisture content and low relative humidity in the forest leads to higher probability of fire in the forest. On the other hand low fuel temperatures combined with high fuel moisture content and high relative humidity results in reduced flammability of forests to fire.

Topography interacts with weather leading to either rapid or slow moving fires. Fires moving on the upslope behave as head fires, whereas fires moving on down slopes behave like back fires. Consequently the effects of these fires are significantly different, the residence time of fires on down slopes is much higher than for fires on

up slopes. In the northern hemisphere south and southwest aspects are exposed to longer periods of solar radiation and this results in increased drying and curing of fuels making them vulnerable to fire occurrence. At higher elevations and under open canopies, fuels are exposed to high wind speeds this leads to the drying of fuels.

The effect of fire on woody plant species in a forest depends on the properties of the bark as well as the reproductive mechanisms associated with a species in the forest. Bark thickness is an important determinant of the protection of a tree from cambial kill during a fire event. The conduction of heat energy through the bark depends on the conductivity of the bark, which in turn is related to the heat capacity of the bark and the thickness of the bark. Apart from the bark thickness the effect of fire on a tree also depends on the duration of the fire and heat exposure. The texture of the bark could also be an important factor in determining the effects of fire on a tree in a forest. Smooth and pale colored barks considerably reduce the effects of fire with reduced surface area as well as reflecting the heat energy, thereby leading to inefficient transfer of heat and reducing cambial kill.

Plants also reveal a number of 'adaptations' to fire in the environment. Certain plant species reproduce vegetatively from subterranean buds. Plants also exhibit behavior such as growth spurts just before a fire event, thereby protecting plants from mortality due to fire. Many trees also exhibit serotiny wherein seeds of the species are viable only when they are exposed to heat energy from a fire event. In the absence of fire these seeds remain unviable. It is in this back drop of the past climate, geology, ecology, and land use change that I conduct my research on forest fires in the Western Ghats.

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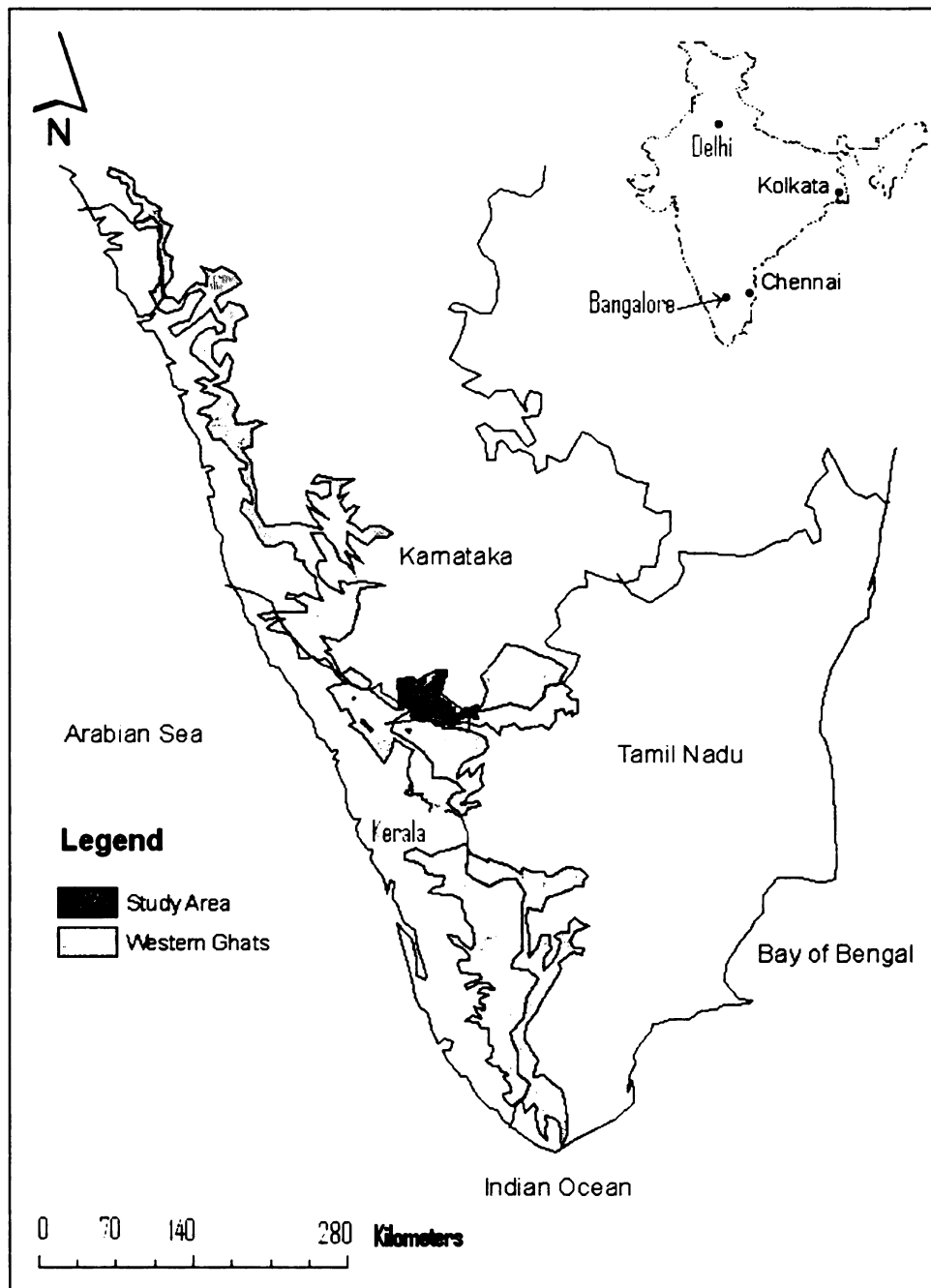
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Figure 1-1: Study area showing the Western Ghats and Nilgiri landscape



Chapter 2

Estimation of Forest Fractional Coverage and Mapping Forest Fires

Introduction:

A number of approaches exist to assess, monitor, and map landscape characteristics in the environment. The application of remote sensing data to these applications is well established and a number of satellites provide extensive data for mapping and monitoring of natural resources. In the past classification of land cover and land use from remote sensing data was based on a system of hard classification, wherein whole pixels were classified into a predefined class based on their spectral signatures. In recent times, a number of studies have adopted a novel approach, under this classification method, it is assumed that spectral information within pixels vary and can be separated into soil, vegetation, and water components (Atkinson et al. 1997). The method is referred to as spectral unmixing analysis and has been commonly applied to delineate satellite data to quantify the spatial extent of disturbance from logging and fire in the tropics (Wang et al. 2005; Cochrane and Souza 1998; Matricardi et al. 2005). I applied remote sensing data from Landsat ETM+ as well as IRS (Indian Remote Sensing) satellites to map and measure various biophysical attributes in the landscape as well as to map the spatial extent of forest fires in the Nilgiri landscape. Table 2-1 provides the data characteristics for the IRS (Indian Remote Sensing) data.

Remotely Sensed Data

A subset of ETM+ data was used to estimate forest fractional cover in the Mudumalai landscape. The image was acquired on the 9th of November 1999, this corresponds to the wet season in the region. The ETM+ image was corrected for radiometric, atmospheric, and topographic distortions. The next few paragraphs provides information on the techniques applied.

Atmospheric corrections:

The ETM+ image was radiometrically corrected by applying the information (solar zenith) in the header file as well from radiance values obtained from (Chander and Markham 2003). The image was converted from digital number (DN) values to top-of-atmosphere (TOA) radiance and then TOA reflectance in the range of (0,1).

Atmospheric effects were corrected by applying the 5S model (Susan 2005).

Estimating coefficients for atmospheric correction

The ETM+ image was acquired during late fall which corresponds to the wet season. The atmosphere model was set as a typical tropical atmosphere, and the aerosol model was continental with visibility = 23 km, a typical visibility in clear air. The 5S model was run to correct the atmosphere effect in the imagery.

The correction equations for bands 1-5 are:

$$\rho_{\text{surf } 1} = 1.0852\rho_{\text{TOA } 1} - 0.0457 \text{ ----- (1.1)}$$

$$\rho_{\text{surf } 2} = 1.0416\rho_{\text{TOA } 2} - 0.0226 \text{ -----(1.2)}$$

$$\rho_{\text{surf } 3} = 1.0218\rho_{\text{TOA } 3} - 0.012 \text{ -----(1.3)}$$

$$\rho_{\text{surf } 4} = 1.0028\rho_{\text{TOA } 4} - 0.0022 \text{ -----(1.4)}$$

$$\rho_{\text{surf } 5} = 1.0006\rho_{\text{TOA } 5} - 0.0003 \text{ -----(1.5)}$$

ρ_{surf} is the surface reflectance, and ρ_{TOA} is the reflectance on the top-of-atmosphere.

After atmospheric correction, the DN values were converted to surface reflectance in the range of (0,1).

For the data obtained from the IRS satellites, I incorporated the dark-object subtraction (DOS) method to correct for the atmospheric effects. I identified areas of dark shadow on the image. Normally these areas would have zero reflectance, however due to atmospheric effects, marginal reflectance was observed. I detected three sites on the image and noted their DN values in each band. I subtracted these values from the entire image to incorporate the atmospheric corrections. Since these satellite products are pre radiometrically corrected, no further radiometric corrections were incorporated.

Geometric correction

The images were geometrically corrected by identifying ground control points, permanent features such as road intersections and bridges over rivers on the satellite image and a topographic sheet. The topographic sheet 58A developed by the Survey of India was used in the purpose. Twelve points were identified and a second order polynomial model was applied to perform the geometric corrections (ERDAS 1997). The total root mean square error from the model was 0.3

Topographic corrections

The Mudumalai landscape exhibits extensive variation in the topography. The elevation ranges from 470 to 1251 m asl. The slope and aspect vary considerably at certain sites in the study areas. Sun-target-sensor geometry affects the reflectance

values at the surface. Therefore, I corrected for topographic effects in the surface reflectance image before further classification and analysis.

In this study, a bi-directional reflectance distribution function (BRDF) model (Rahman et al. 1993) was applied to correct the topographic effect in the ETM+ surface reflectance image. For each pixel, the local slope, azimuth aspect, and local incidence angle was calculated from DEM data. A nadir view was chosen as the standard view direction, and the reflectance from its local angle was adjusted to the one with standard incidence angle (0°).

Let θ_s , θ_v , φ_s , φ_v be sun zenith, satellite view zenith, sun azimuth, and view azimuth angles, these values are present in the image header file. ($\varphi_s = 0$ and φ_v corresponds to value from the image), α and β are the slope and aspect which was calculated from DEM data, local sun zenith θ'_s , local view zenith θ'_v , and local relative azimuth angle φ' was calculated with the following equations:

$$\cos\theta'_s = \cos\alpha \cos\theta_s + \sin\alpha \sin\theta_s \cos(\beta - \varphi_s) \text{-----(2.1)}$$

$$\cos\theta'_v = \cos\alpha \cos\theta_v + \sin\alpha \sin\theta_v \cos(\beta - \varphi_v) \text{-----(2.1)}$$

$$\varphi' = |\beta - \varphi_s| \text{----- (2.3)}$$

By considering the sun-target-satellite geometry θ_s , θ_v , φ (relative azimuth angle) as standard geometry, the surface reflectance in this geometry is modeled by applying the BRDF proposed by Rahman et al. (1993).

$$\rho(\theta_s, \theta_v, \varphi_s) = \rho_0 \frac{(\cos\theta_s \cos\theta_v)^{k-1}}{(\cos\theta_s + \cos\theta_v)^{1-k}} \frac{1 - \omega^2}{[1 + \omega^2 - 2\omega \cos(\pi - \xi)]^{3/2}} \left(1 + \frac{1 - \rho_0}{1 + G}\right) \text{----(2.4)}$$

where ρ_0 and k are two empirical surface parameters. I determined these surface parameters from the reflectance values chosen (described below) from the ETM+ image and the corresponding θ_s , θ_v , ϕ_s obtained from the DEM. I obtained 150 data values from the satellite image and the DEM and generated the coefficients from a BRDF inverse model. Ω is the function parameter controlling the relative amount of forward ($0 \leq \omega \leq 1$) and backward scattering ($-1 \leq \omega \leq 0$). The phase angle ξ and geometric factor G can be calculated by applying the following equations:

$$\cos \xi = \cos \theta_s \cos \theta_v + \sin \theta_s \sin \theta_v \cos \phi \text{ -----(2.5)}$$

$$G = \sqrt{\tan^2 \theta_s + \tan^2 \theta_v - 2 \tan \theta_s \tan \theta_v \cos \phi} \text{ -----(2.6)}$$

Similarly, the surface reflectance $\rho'(\theta'_s, \theta'_v, \phi')$ in local geometry can be calculated. The correction coefficient $\rho(\theta_s, \theta_v, \phi_s) / \rho'(\theta'_s, \theta'_v, \phi')$ was applied to the ETM+ surface reflectance image.

The topographically corrected image reveals greater differences in the vegetation especially at high slopes or regions of large topographic variations. Figures 2-1a and 2-1b show a subset of the ETM+ image (Band 4+3+2) near the Moyar valley before and after the BRDF correction. The BRDF correction was instrumental in removing some of the shadowed areas in the image and thus providing information on the vegetation features at these high slopes.

Thus the satellite images were subjected to radiometric, atmospheric, and topographic corrections before eventually carrying out the fractional estimates for the Mudumalai landscape.

Estimating forest fractional coverage of the Mudumalai landscape

Forest fractional cover (f_c) is the area covered by forest canopies per unit area on the ground. Tropical forests have a range of fractional cover. Forests in the Western Ghats are broadly classified into 5 main types, they include the tropical evergreen forests, the tropical moist deciduous forests, the tropical dry deciduous forests, the tropical dry thorn forests, and the highly endangered tropical high elevation evergreen ecosystems. However as outlined earlier most of these natural vegetation forms have been transformed through a variety of biotic actions. The result has been a spectrum of vegetation formations that reveal strong intra vegetation variations in canopy cover and density. Soil reflectance varies among the different wavelengths, the inherent differences between the two are applied to separate them in remotely sensed data. In many forested areas and in the Western Ghats, delineation of land cover may not be as straightforward. Forest fractional coverage can be estimated with sub-pixel information both in the reflectance domain (Hanan and Prince 1991; Price 1992; Radeloff et al. 1999) as well as the vegetation index domain (Wang et al. 2005; Matricardi et al. 2005). The estimation of forest fractional coverage in the reflectance domain proves a little problematic, due to choice of spectral bands. Most studies have applied the red and NIR to estimate fractional coverage, although the spectral signal from vegetation dominates both these bands, it is extremely sensitive to moisture, as well as the structure of soil and vegetation and external factors such as the atmospheric effects and the sun-target-sensor effects (Wang 2004). In order to overcome this problem of the reflectance domain, a number of studies have applied the vegetation index method which suppresses these effects and make vegetation and non-vegetation comparisons more stable (Jasinski 1990). Images in this dissertation are presented in color.

A linear unmixing model

Fractional coverage (f_c): Estimation of fractional green vegetation cover, f_c , from remotely sensed data is often associated with computation of spectral vegetation indices and their empirical relationships to fractional green vegetation cover. Because of their functional linkage with vegetation canopy properties, many spectral vegetation indices have been developed to enhance sensitivity to green vegetation signals and reducing external effects from soil and atmospheric variations (Qi et al. 2000). I applied a simple linear mixing model that assumes that pixel signals consist primarily of two components: soil and vegetation. Under this assumption, the fractional green vegetation cover is f_c and, the fraction $(1 - f_c)$ is the soil cover. The resulting signal, S , as observed by a remote sensor can be expressed as

$$S = f_c S_v + (1-f_c)S_s + \varepsilon \text{ -----(3.1)}$$

where, S_v is the signal contribution from the green vegetation component and S_s from the soil component and ε is an error term that accounts for non-soil or vegetation contributions (near zero and is hereafter not considered). Equation (3.1) can be applied to remotely sensed data in the reflectance domain (Maas 1998) and in the spectral vegetation index domain (Zeng et al. 2000).

Determining the optimum vegetation index:

The reflectance values of various surfaces are variable in different wavelengths of the EMR spectrum. Green vegetation exhibits higher values in the NIR and low values in the red wavelength. A vegetation index is a mathematical combination of multiple spectral bands, which can suppress the external effects mentioned above (Qi et al. 2000). Hence we can rewrite the equation as

$$VI = VI_{\text{canopy}}f_c + VI_{\text{soil}} (1-f_c) \text{-----}(3.2)$$

VI_{canopy} is the vegetation index associated with the canopy of a given pixel and the VI_{soil} , the index associated with the soil component.

A number of vegetation indices have been explored in the literature. The most commonly applied indices include the Normalized Difference Vegetation Index (NDVI); Soil Adjusted Vegetation Index (SAVI); Modified Soil Adjusted Vegetation Index (MSAVI);

I use the modified soil adjusted vegetation index (MSAVI) because it reduces the effect of the soil background by using a soil adjustment function (Wang 2004). The adjustment factor in MSAVI is also sensitive to changes of canopy density (Qi et al. 1994). The equation for MSAVI is given below in equation 3.3

$$MSAVI = \frac{\rho_{\text{NIR}} - \rho_{\text{red}}}{\rho_{\text{NIR}} + \rho_{\text{red}} + L_0} (1 + L_0) \text{-----}(3.3)$$

where L_0 is a soil adjustment function. It is expressed as a function of the soil line slope and the reflectance properties on the ground. I applied a value of 1.2 to the slope of the soil line, which was obtained by plotting NIR against red for vegetation and soil surfaces.

$$L_0 = [(\rho_{\text{NIR}} - \rho_{\text{red}})\text{slope} + 1 + \rho_{\text{NIR}} + \rho_{\text{red}}]^2 - 8.0 \times \text{slope} \times (\rho_{\text{NIR}} - \rho_{\text{red}}) \text{-----}(3.4)$$

In the MSAVI image of the Mudumalai landscape, the moist deciduous forests in the West have the highest fractional cover values and Moyar settlement in the East with fallow lands has the lowest values.

Thus equation 3.2 can be rewritten as (3.5)

$$MSAVI = f_c \times MSAVI_{veg} + (1 - f_c)MSAVI_{soil} \text{-----}(3.5)$$

Hence fractional coverage can be rewritten as

$$f_c = \frac{MSAVI - MSAVI_{soil}}{MSAVI_{veg} - MSAVI_{soil}} \text{-----}(3.6)$$

Where $MSAVI_{soil}$ is the VI value of an area of bare soil and $MSAVI_{veg}$ is the VI value of a pure vegetation pixel.

The two endmembers $MSAVI_{veg}$ and $MSAVI_{soil}$ in equation 5 were identified in the ETM+ image. A fallow field in the East of the landscape was chosen to represent the soil component in the study area. The moist deciduous forests in the West were chosen to represent the full cover green vegetation. The values obtained from these two regions were applied as endmembers in equation 5, $MSAVI_{veg}=0.8$ and $MSAVI_{soil}=0.1$.

Canopy fractional cover analysis

Figure 2-2 is the f_c map of the Mudumalai landscape, I have binned the values into 5 classes. All pixels with f_c values less than 0 or higher than 100% (reaching saturation) have been truncated into 0 and 100% respectively. The settlement at Moyar has low f_c , the tropical dry thorn forests in the East have values between 25 and 40%. The tropical dry deciduous forests have values between 35 and 55% and the tropical moist deciduous forests have values > 60%. There are a few areas in the

dry thorn forests with high f_c values. The presence of two shrubs, *Lantana camara* and *Chromoleana odorata* and reflectance from their leaves has resulted in these high values (Wang et al. 2005). Thus there is a gradient in the f_c with low values in East, increasing to high values in the West.

Validation

I validated the f_c with fisheye pictures taken along 100 m transects in the different vegetation types. The f_c values calculated with these pictures were used to calculate the canopy cover using the GAP analyzing software (ter Steege 1996). GPS locations of the transects were taken along with the canopy pictures, these were later superimposed over the f_c image. The mean canopy cover along the transect was calculated and the corresponding average pixel values of f_c were obtained from the ETM+ image. Figure 2-3 compares the ground measured canopy cover values and the ETM+ estimated f_c values. The ETM+ f_c estimated values and the canopy cover are correlated, the R-square value is 0.6, $F=19.8$, $p < 0.01$. The canopy cover values and the f_c values obtained from the ETM+ image are consistent with reference to the tropical moist deciduous forest and the tropical dry thorn forests, however the values of ETM+ values of tropical dry deciduous forests are lower than the corresponding canopy cover values. The difference between the ETM+ estimated values and the ground estimates for the tropical dry deciduous forests could be due to the canopy openings in these forests.

Forest Fire Mapping in the Nilgiri landscape

I developed a methodology specific to my study area. I performed supervised classification by generating training sites from burned areas that I identified on the

images. I applied an interactive process (explained below) by which, I collected spectral signatures of burnt areas in each of the three broad vegetation types present in the study area, namely the tropical dry deciduous, the tropical moist deciduous, and the tropical dry thorn forest ecosystems. The advantage with this method over using only a single vegetation type has been the ability to capture the variability in burned areas due to the differences in the structure, phenology, and exposure to soil fractions in these ecosystems. The fire maps were delineated by combining burned areas from each of the three vegetation types. The fire maps were assessed for their accuracy from actual ground truth data.

Interactive accuracy development of fire maps:

A few pixels in the fire maps were misclassified and a few pixels with shadows also were misclassified as burnt areas, I identified these areas and removed them from the fire map through the following steps. I first carried out despeckling to remove noise in the dataset, I passed a 3 x 3 low pass filter to remove isolated pixels to get rid of the 'salt and pepper' noise. Next I did a geographic link of the fire map and the false color composite and examined the fire map for misclassified areas, I detected a number of shadow pixels, that were misclassified as burnt areas, I corrected the misclassification and finally performed the accuracy assessment of the fire maps. The accuracy of the final fire map generated was 85%. Table 2-2 gives the error matrix for the fire maps obtained from remote sensing data for the Nilgiri landscape. GPS points were obtained for burnt areas during the field trip. These points were used to validate the fire maps. The accuracy assessment shows that the user's accuracy is 100% while the producer's accuracy is 67% whereas the overall accuracy of the fire map is 85%. A similar approach was applied in delineating fire maps of the remaining years in the

Nilgiri landscape and hence similar accuracies are expected for these previous years fire.

Vegetation map of the Mudumalai landscape

I obtained a vegetation map for the Mudumalai landscape by applying supervised classification methods. Spectral signatures of the different vegetation types were collected from the satellite data. The different vegetation types were delineated into tropical moist deciduous, tropic moist deciduous(degraded), tropical dry deciduous forests, tropical dry thorn forests, and settlements (fig 2-4). A high resolution map of vegetation for the Nilgiri Biosphere Reserve is currently available (Pascal and Prabhakar 1996). I used a similar classification system. Some misclassification of the different vegetation types existed, I reclassified these areas with reference to this map. The error matrix of the classification of the vegetation types are provided in table 2-3.

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Figure 2-1a: False color composites of Mudumalai landscape before BRDF correction

Figure 2-1b: False color composites of Mudumalai landscape after BRDF correction

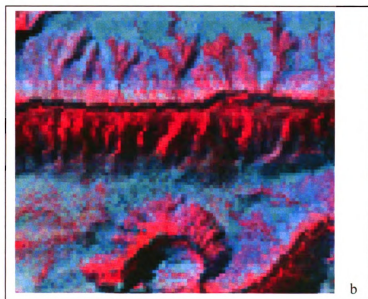
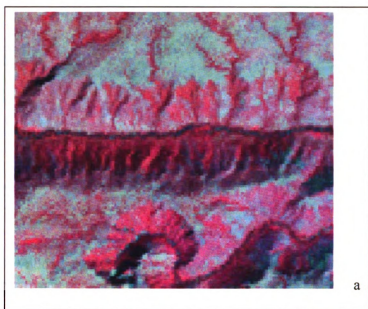


Figure 2-2: Forest fractional coverage of the Mudumalai landscape

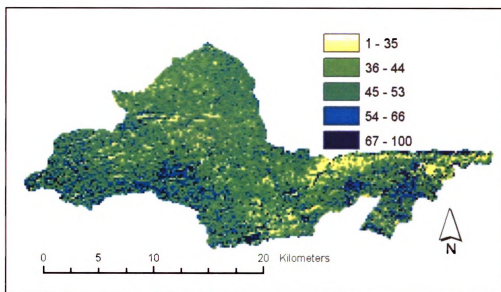


Figure 2-3: Validation of forest fractional coverage

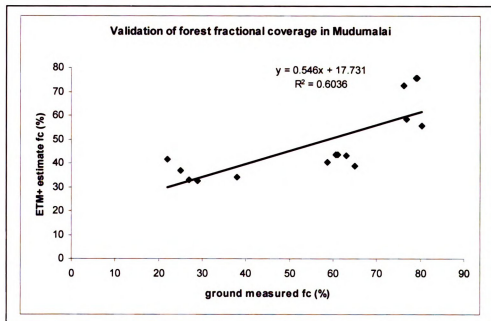


Figure 2-4: Vegetation type map of the Mudumalai landscape

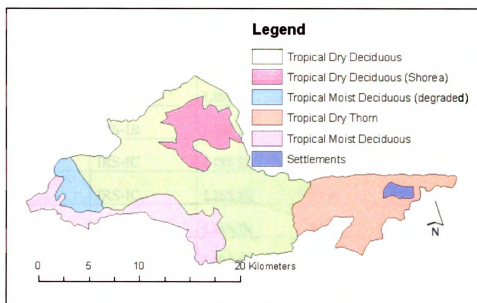


Table 2-1: Data features of satellite data

Year	Satellite Data	Sensor	Spatial Resolution	Date
1989	IRS-IA	LISS I	72 m	02/06/1989
1996	IRS-IB	LISS I	72 m	03/6/1996
1997	IRS-IC	LISS III	24 m	03/22/1997
1999	IRS-IC	LISS III	24 m	03/12/1999
2001	IRS-ID	LISS III	24 m	03/1/2001
2002	IRS-IC	LISS III	24 m	02/24/2002
2004	IRS-P6	LISS III	24 m	03/09/2004
2005	IRS-P6	LISS III	24 m	03/04/2005

Table 2-2: Accuracy assessment for fire map obtained in 2005

User's accuracy (%)	Producer's accuracy (%)	Overall accuracy (%)	Kappa coefficient
100	67	85	0.68

Table 2-3: Accuracy assessment for vegetation map of Mudumalai landscape

Vegetation Type	Producer's accuracy (%)	User's accuracy (%)	Kappa coefficient
Tropical Dry Deciduous Forest	75	75	0.63
Tropical Moist Deciduous Forest	85.7	100	1
Tropical Dry Deciduous (<i>Shorea</i>) Forest	75	75	0.7
Tropical Moist Deciduous degraded Forest	87.5	70	0.64
Tropical Dry Thorn Forest	77.7	100	1
Settlement	100	66.7	0.65
Overall	83	80	0.74

Chapter 3

Fire-return intervals: Spatial and temporal characteristics in different vegetation types

Kodandapani, N. Mark A. Cochrane and Sukumar, R. 2004. Conservation threat of Increasing fire frequencies in the Western Ghats, India. *Conservation Biology*. 18(6):1553-1561.

Introduction

Research in the tropics has focused on deforestation or static evaluations of forested areas and seldom has considered the contributing effects of landscape processes and biotic pressures in the loss of biodiversity (Sanchez-Azofeifa et al. 1999). Although studied separately to varying degrees, the combined ecological impacts of forest fragmentation and biotic pressures, such as grazing, logging, and forest fires, are poorly understood. The interactions and synergies between multiple disturbances have recently become a focus of study in the New World tropics (e.g., Cochrane 2001; Peres 2001; Laurance & Williamson 2001; Peres et al. 2003). However, similar research and understanding is lacking in the Old World tropics, which have been affected by millennia of human occupation and associated disturbances. Regions such as the Western Ghats in India are logical places in which to evaluate the ongoing effects of multiple, human-related disturbances.

In the Old World tropics, especially India, human population growth has led to extensive deforestation, greatly fragmenting remaining forests (Menon & Bawa 1998; Jha et al. 2000). Rainforests in various countries, including India, have largely been deforested in recent decades (Myers 1994; Whitmore 1997). Reports from the Ministry of Environment

and Forests in India, suggest that forest fires affect 37 million hectares of forests annually, and about 55% of the country's forest areas are being subjected to forest fires each year (Gubbi 2003). The Western Ghats is one of the world's biodiversity hotspots (Myers 2000), but reserves are limited in size and surrounded by an intervening matrix of land that is under intense human pressures. Furthermore, landscape processes, such as fragmentation and forest fires have been overlooked in conservation studies in the Western Ghats (Daniels et al. 1995; Jha et al. 2000). The spatial and temporal scales of disturbance, forest-fragment configuration and shape, land-use activity in the surrounding matrix, and additional ongoing disturbances can synergistically manifest as subtle deforestation events in tropical landscapes, threatening forest remnants with constant elimination (Aubreville 1947; Cochrane et al. 1999; Gascon et al. 2000; Cochrane & Laurance 2002). Thus, understanding the effects of these disturbances especially, forest fires and forest fragmentation can provide insights into current deforestation processes and also help direct conservation programs in these ecosystems.

As in other parts of the world, fire has been the tool of choice for clearing forests in the Ghats. Researchers have examined the rates of deforestation and degree of fragmentation in the Ghats, but they assumed forest remnants remain unchanged (Menon & Bawa 1997; Nagendra & Utkarsh 2003). Furthermore, the potential synergistic effects of fragmentation and fire have not been considered. Disturbance in ecosystems occurs at various scales. Certain events are fine grained and affect individuals in a population, whereas other disturbances are larger grained and affect assemblages of species in a community. Still larger disturbance events affect entire landscapes and ecosystems

(Forman 1995; Ross et al. 2002). Although spatial scale is an important variable in all the above-mentioned disturbance events, the spatial configuration of landscape elements, and the disturbance history of the landscape are vital to the holistic understanding of disturbance events in ecosystems (Turton & Freiburger 1997).

In human-affected landscapes, the current distribution of forests is largely a result of the spatial and temporal interactions between humans and their environment. One disturbance that is strongly related to human activity is fire. The nature, amount, and spatial distribution of ignitable fuel largely govern the character of the fire in any forest location (Goldammer 1990). Increasing dependence on forests by humans for a variety of uses leads to forest fragmentation that further exacerbates future fire events in the landscape (Cochrane 2003). The spatial juxtaposition of forests and other land covers, derived through anthropogenic land-use, has a large influence on the extent and frequency of fire events on fragmented landscapes (Cochrane et al. 1999; Cochrane 2001). The ecological characteristics of forests, combined with previous disturbance history and prevalent weather patterns, leads to variability in the pattern of burning in forests (Uhl & Kauffman 1990; Glitzenstein et al. 1995; Cochrane & Schulze 1999). The differences among forests in terms of ecological characteristics, climatic factors, and associated disturbance histories translates into differences in susceptibility to and intensity of forest fires among vegetation communities across the landscape. Within the Western Ghats, the remaining forests are fragmented and immersed in a human-dominated landscape, where fire is frequent. Current conservation strategies center on reserves, however, deforestation continues to occur and no account is made of dynamic

processes in assessing the performance or capability of these protected areas to conserve biodiversity or ecosystems. Thus, detailed studies on fragmentation and fires may provide critical insights for sustainable conservation of biodiversity in this region.

As a first step in generating a clearer understanding of landscape-scale spatial and ecological processes in the Western Ghats, I sought to place fire disturbance in perspective as an agent of change and as a potential conservation threat in the Western Ghats. My objectives were to (1) assess the differences in forest fire frequency of different vegetation types in the focus study area in the Western Ghats, (2) delineate current fire-return intervals in a representative sample landscape of the Western Ghats, (3) assess forest fire frequency at various spatial scales, the vegetation type, the landscape, and regional scale in the Western Ghats, (4) explore possible synergisms between forest fires and forest fragmentation and implications of these synergisms to the biodiversity of the Western Ghats.

Study Areas

In order to examine these disturbances at different spatial scales I chose 3 nested study areas. The Western Ghats region in India is a long mountainous massif (8-12°N latitude, 73-77°E longitude) running all along the west coast of peninsular southern India (Fig. 3-1). This distinctive ecoregion covers $1.7 \times 10^5 \text{ km}^2$ and has been categorized as the Western Ghats Moist Forest major habitat type by the World Wildlife Fund (WWF 1998). Experiencing much higher levels of precipitation than the adjoining regions of peninsular India, the natural biota of the region, exhibit a high level of endemism (Subramanyam & Nayar 2001). Variations in topography and climate have created a

highly diverse landscape with land-cover that range from tropical evergreen forest to tropical dry thorn forests. At present, all regions of the Western Ghats exhibit additional diversity, in terms of their landscape components, that range from nearly undisturbed to highly degraded (Nagendra & Gadgil 1999). The current landscape is testimony to the ever-changing social and economic interactions between forest growth and human use of these resources. Although human occupation of the Ghats is ancient, large-scale deforestation and destruction of forests is a more recent phenomenon (Chandran 1997; Subramanyam & Nayar 2001). The increased use of forests and rapid changes in the land-cover of the region have fragmented the remaining forests. This fragmentation and disturbance has led to increased susceptibility of the remaining forests to large and recurrent fires. The present land cover is characterized by the presence of fire-maintained agricultural fields adjacent to forests, large-scale usage of forests for various activities (e.g., firewood and non timber forest product collections), and altered fire regimes of the remaining forests.

The Nilgiri Biosphere Reserve is spread over the states of Karnataka, Kerala, and Tamilnadu in southern India (Fig. 3-1). The forested area of the reserve is 5520 km² (Sukumar 1990). There is a distinct west-east climatic gradient, which gives rise to a diversity of vegetation types. The Nilgiri Biosphere Reserve supports all the major vegetation types of peninsular India (Champion & Seth 1968). These include tropical evergreen and semi evergreen forest, tropical moist deciduous forest, tropical dry deciduous forest, and tropical dry thorn forest. At higher elevations (>1800 m) there are characteristic patches of tropical montane stunted evergreen forest, in the valleys and

folds of the hills and extensive grassland on the hill slopes. More detailed descriptions of the vegetation are available elsewhere (Nair et al. 1977). The Nilgiri Biosphere Reserve is representative of the Western Ghats in terms of its vegetation types and, despite its protections, a region under immense human pressures.

I conducted detailed analysis of fire-return intervals across the different vegetation types in the Mudumalai Wildlife Sanctuary (MWLS). The sanctuary is a part of the Nilgiri Biosphere Reserve and is a long-term ecological research site. The sanctuary is 320 km² and harbors diverse flora and fauna, including 616 vascular plant species. A single 50-ha plot in deciduous forest at the sanctuary contained 71 woody species ≥ 1 cm dbh (Sukumar et al. 1992). Fire maps for the sanctuary were available from 1989 to 2002, and annual areas burned for an earlier period (1909 – 1921) were also available from a working plan for the sanctuary (Hicks 1928).

Methods

I surveyed and mapped fire occurrence in the MWLS on topographic maps of 1:50,000 scale. Two topographic maps covering the entire sanctuary were used, mapsheets 58 A6 and 58 A10 (of the Survey of India). The burnt areas were first located on the ground and then on the topographic maps based on key features such as rivers, swamps or roads. Burned areas were mapped after walking the perimeter of the affected vegetation. Mapping of burned areas was done during the months of April/May, the dry season, before the onset of the south-west monsoon. The mapping activity generally took about 2 weeks, however, there have been occasions when the source and timing of ignitions

increased the amount of time necessary. These detailed maps are available for the years 1989 to 2002. Where available satellite data has been applied to map the spatial extent of fires; data from IRS satellites were used to map forest fires in 1989, 1996, 1997, 1999, and 2001. Fire maps of MWLS from 1989 to 2002 were individually overlaid with the vegetation map of the sanctuary. The resultant map yielded polygons with attributes of vegetation type and presence or absence of fire. I obtained proportion of area burned in the different vegetation types for the different years from this map and used statistical tests to examine differences in fire frequency across the different vegetation types.

Details of satellite data from (Indian Remote Sensing) satellites can be found in chapter 2. Satellite data for the study area were extracted, geo-corrected, and classified into burned and unburned areas with supervised classification. I used ERDAS IMAGINE 8.3 to analyze the satellite data. The raster data for these 5 years were converted into vector form and combined along with the vector data of other years. I assigned unique identity values to the burned and unburned areas. The data for 14 years were combined and a single composite map obtained.

I estimated fire frequency by spatially quantifying the number of fires for each location in the composite map. There were 627 distinct areas in the combined map. Frequency of burning in each location was calculated as the number of years (out of 14) in which fire had been detected. Fire-frequency maps were converted into an estimated fire-return interval map for each forest land-cover type. A fire-return interval is the amount of time required to burn an area, equivalent to the entire forested area, with the understanding

that some areas may not burn whereas others may burn more than once during a cycle (Van Wagner 1978).

I delineated the vegetation map of MWLS using supervised classification of the IRS I-B, (LISS III row 65, path 99, 27 March 1996) satellite data. Six land-cover types were classified: tropical dry deciduous forests, tropical dry deciduous (*Shorea*) type, tropical moist deciduous, tropical moist deciduous (degraded), tropical dry thorn forests, and settlements (Fig. 3-2). The classification system followed by Prabhakar and Pascal (1996) served as a reference in the preparation of the vegetation map. The accuracy of vegetation types ranged from 70% in the case of the tropical moist deciduous (degraded) to 100% in the case of the tropical dry thorn forest type, the overall accuracy with 80%.

To extrapolate the results from the MWLS to both the entire NBR and the larger area of the Western Ghats, I made the conservative assumption that the results from the various vegetation cover types within the MWLS are representative of fire frequency and extent of these vegetation types in these larger regions. This extrapolation is conservative because the MWLS is a protected region of forest, nested within a similar, but larger, protected area, the Nilgiri Biosphere Reserve. Human use of these forests is therefore limited and because 90% of all forest fires are estimated to be anthropogenic in origin (Bahuguna 1999), fire conditions are expected to be less severe than those in the majority of unprotected forests of the Ghats. Unfortunately, the MWLS does not contain two Western Ghats forest types, the tropical evergreen forests and high elevation montane/grassland ecosystems. These forests are known to burn (Hegde et al. 1998) but

because we do not have quantitative data on fire occurrence in these vegetation types I excluded them from the larger scale estimations of regional burning. For the mid-level scale fire extent estimation of the Nilgiri Biosphere Reserve, I used the vegetation cover data from Narendran et al. (2001). Of the total forested area of 5520 km², the tropical dry thorn, tropical dry deciduous, and tropical moist deciduous forest types are the predominant land-covers, accounting for 38%, 29.7% and 16.3% of the area, respectively. The tropical evergreen forests and tropical montane forests/grasslands accounted for the remaining land-cover in the NBR, 10.6% and 5.2% of the area, respectively.

For the regional scale assessment I adapted the vegetation cover information from a recent assessment of terrestrial ecoregions (Olson et al. 2001). In the assessment, distinction was made between northern and southern tropical moist and tropical montane rainforests, but we combined northern and southern classes because we were concerned with physiological conditions and not species compositions. Therefore, for the subsequent analyses, tropical dry deciduous and tropical moist forests respectively accounted for 27% and 42% of the area. Tropical evergreen and grassland/montane rainforests accounted for roughly 31% of the vegetation cover but were excluded from the fire analysis because they do not occur in the MWLS.

Results

Forest fires were common in forests of the MWLS and an average of 27% of the landscape burned in a given year (Table 3-1). The vegetation type with the highest mean

area burned was the tropical dry deciduous (*Shorea*) with 59%, whereas the tropical moist deciduous vegetation had the lowest mean area burned with 10%.

There was variability in the susceptibility to fires across the different vegetation types (Fig. 3-3). Significant differences existed between the tropical dry deciduous vegetation and all other vegetation types in the MWLS ($p < 0.01$). Fire extent in the tropical dry deciduous (*Shorea*) vegetation is significantly higher than the, tropical moist deciduous (degraded), tropical dry thorn, and tropical moist deciduous vegetation types ($p < 0.001 - 0.1$) Table 3-2. To investigate if small sample sizes were responsible for these patterns we also conducted Mann Whitney U tests, which also showed significant differences (all $p < 0.01$) Table 3-3. As these samples are random and independent, we rule out inflation of results due to temporal auto correlation, however, spatio-temporal auto correlation is important in assessing patterns of fire occurrence across a landscape and this is under investigation. Fuel loads from both grasses and leaf litter were also significantly higher in tropical dry deciduous (*Shorea*) and tropical dry deciduous forests compared with all other vegetation types (Dattaraja unpublished data).

The fire-return intervals in the forests of the study region were short. More than 50% of the study site experienced fire between 4 – 10 times in the past 14 years (1989-2002), implying that half the study site will experience a fire every 3.5 or less years (Fig 3-4). The tropical dry deciduous (*Shorea*) had the shortest fire-return interval of 1.7 years, whereas the tropical moist deciduous forests had the longest fire-rotation interval of 11 years. Based on the last 14 years of fire data, the average fire-rotation interval of the

entire sanctuary is 3.7 years. This was considerably shorter than the average 10 year fire-return interval for a similar period 90 years ago (1909-1921). The current fire dynamic is one of rapid return fires and is more frequent than the earlier time period for the study region (Fig. 3-5). Forest fires burned an average of 27% ($90.3 \text{ km}^2/\text{yr}$) of the forests in the landscape of the MWLS each year. At the mid-scale of the Nilgiri Biosphere Reserve, an estimated average of 17% ($895.5 \text{ km}^2/\text{yr}$) of the forests burned every year. At the regional scale of the Western Ghats, average annual burning was estimated to cover 13% ($21,902 \text{ km}^2/\text{yr}$) of the forests.

Discussion

This analysis of forest fires in the Western Ghats demonstrates the importance of forest fires as recurrent disturbance events, with potentially severe consequences for the conservation of biodiversity in the Western Ghats. Our results demonstrate that, although the frequency of forest fires varies across the different vegetation types of the MWLS, all of these ecosystems burned frequently.

Forest fires have been a part of these ecosystems for many thousands of years (Gadgil & Chandran 1988; Gadgil & Guha 1993). In many ecosystems, fire is part of the natural regeneration process, stimulating the germination of certain species, clearing space for the invasion and growth of others, and releasing a periodic flush of nutrients into the soil (Dawson et al. 2002). In the Western Ghats, however, only the grasslands associated with montane forests are considered to be fire-maintained ecosystems (Meher-Homji 1984). What is clear from this study is that the character of fire disturbance events has

been altered fundamentally in recent decades, resulting in much shorter fire-return intervals. This problem has been compounded by land-cover transformations in the surrounding landscape. Large tracts of forests have been lost to plantations of coffee, tea, rubber, eucalyptus, agriculture, and a number of hydroelectric projects (Subramanyam & Nayar 2001; Ramesh 2001). Across the Western Ghats, 40% of the natural vegetation has been lost between 1920-1990. This has resulted in increased fragmentation of the remaining habitats, with the number of patches growing four-fold and a concomitant 83% reduction in mean patch size during this time period (Menon & Bawa 1997). I hypothesize that this fragmentation of the forests makes them more vulnerable to escaped agricultural fires along their extensive edges and that the reduced patch size makes it more likely that entire fragments will burn during each fire event.

With a sanctuary-wide fire-return interval of only 3.7 years, it is doubtful that the current vegetation composition of the MWLS can persist. Although fires were common in the 1910s, it is clear that the fire rotation is accelerating. Fires in tropical forests affect the species composition, demography, structure, and biomass of forests (Holdsworth & Uhl 1997; Cochrane & Schulze 1999; Haugaasen 2000). The mechanism by which, these fires have modified the vegetation cover of the Western Ghats is outlined by Hegde et al. (1998). As the fire-return intervals increase, it becomes more unlikely that the majority of tree species will be able to recruit new trees to a size resistant to mortality from the frequent fires. A study on the demography of woody plant species in the sanctuary found a decline in abundance of many species due to poor recruitment and high mortality from ground fires between 1988-1996, these persistent fires could lead to an increase in

dominance and a decrease in diversity in this forest even on short time scales (John et al. 2002). In general, it is to be expected that fewer species will be able to persist. Already, the effects of the rapid fire-return intervals in the sanctuary has lead to some monodominant and even aged-stands of *Shorea roxburghii* (Kodandapani 2001).

In addition to direct mortality from fire, frequent fires may contribute to the rapid invasion of the sanctuary by exotic fire-adapted species. Already, *Lantana camara* and *Chromolaena odorata* have colonized regions subjected to repeated forest fires (Kodandapani 2001). Besides competing with native species for resources and space, these and other exotic invasive species may also alter the fire behavior in these forests by changing the fuel structure and hence potentially creating more intense fires that could further accelerate the loss of native species.

Although I was unable to include the high elevation grassland/montane forests and the tropical evergreen forests, both of these ecosystems do burn in the Western Ghats. The tropical evergreen forests may be the most threatened by fire. These wetter forests are resistant to fire propagation but they are also very sensitive to fire-related damage and high mortality levels when fires do occur (Uhl & Kauffman 1990; Cochrane & Schulze 1999; Slik et al. 2002). In general, species in tropical evergreen forests are poorly adapted to fire disturbance (Uhl and Kauffman 1990) being characterized by thin bark, buttressed trunks and stilt root systems (Hegde et al. 1998; Barlow 2003).

The results of this study clearly show that there has been a long-standing and strong selective pressure against fire-sensitive species throughout much of the Western Ghats. The environmental changes brought about by forest fragmentation and accelerated fire-return intervals are expected to favor deciduous species at the cost of shade-tolerant and moisture-loving evergreen species (Daniels et al. 1995). Further, as the extent and the frequency of burning increase, the remaining fire-sensitive and even the moderately fire resistant species may be eradicated through both fire-related mortality and recruitment failure. Apart from changing the demography and structure of surviving plant communities, high tree mortality can result in increased fuel loads and more rapid rates of desiccation in the damaged forests. This can result in a positive feedback wherein successive fires become both more likely and more severe until complete deforestation occurs (Cochrane et al. 1999). Depending on the frequency of burning, fires can therefore be expected to either facilitate invasion by more fire-resistant species (e.g., thicker-barked drought deciduous species) or, potentially, lead to total eradication of all trees. Invasion of the evergreen forests by more disturbance tolerant species may be masking the erosion of these ecosystems if they are concurrently being replaced with other vegetation types.

Throughout the Western Ghats, deforestation continues to be a severe problem despite a legal moratorium on deforestation that has been in place since the enactment of the Forest Conservation Act of 1980. Due to a lack of systematically collected spatial data on forest cover change, it is currently unclear how much of this deforestation is actually due to intentional, illegal deforestation. What is clear from the comparative analysis of fire

occurrence during the 1910s and 1990s (Fig. 3-5) is that the landscape fire dynamic has changed, with an approximately 200% increase in the amount of burning. While it is possible that some species of trees may be able to persist under the current fire regime, I have not discovered any forested ecosystems anywhere that regularly experience and thrive in FRI of less than 5 years. The increased amount of fire in these forests is likely due to increased anthropogenic sources of ignition, forest fragmentation, altered fire regimes, and possible synergisms among other factors. These accelerated disturbance cycles may be, in part, responsible for the ongoing deforestation in this region. The observed fire-return intervals in the MWLS are short enough to result in gradual eradication of many of the surviving forests, as are the estimated fire frequencies for the Nilgiri Biosphere Reserve and the Western Ghats. This deforestation would most likely occur as progressive elimination of fragments (Aubreville 1947; Gascon et al. 2000; Cochrane & Laurance 2002) but could also occur rapidly in sections of interior forests that have been severely damaged by previous disturbances (Cochrane et al. 1999). I stress that the fire-return intervals for regions of the Western Ghats outside protected reserves are likely much worse than the conservative estimate of roughly 16% burned per year that we give here. This is because these unprotected forests have much higher human population densities, are more heavily used for various activities, are more fragmented, and are closer in proximity to fire-dependent agriculture. Recent studies on the extent of forest fires in India suggest that about 50% of the country's forests burn yearly (WWF 2003).

Conclusions

Forest fires are frequent across the landscapes of the Western Ghats. Forest fires, fragmentation, and their synergisms may be driving deforestation processes that are fundamentally altering the landscape of the Western Ghats. A number of local, national, and international efforts are underway to conserve the biological wealth of this region but these initiatives will be unsuccessful if the processes driving deforestation and forest degradation are not taken into account. It is imperative to understand these landscape changes and their drivers to promulgate effective strategies for the conservation of the ecosystems and biodiversity of the Western Ghats. The current static policy of nature conservation through demarcation of reserves and legislation against intentional deforestation might not be the only answer to protecting the region's biodiversity. Instead, a more dynamic approach that incorporates policies affecting land-use and land-cover change together with management and monitoring of landscape-level disturbances such as fragmentation and forest fires could prove more beneficial in future conservation programs.

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Table 3-1: Mean area and percentage of area burnt in different vegetation types of the

Vegetation Type	Mean area burned/year (Km²)	(%) Vegetation type	Fire-return interval
Dry Deciduous	59.5 ± 37.7	33.0	3.0
Dry Deciduous <i>Shorea</i> type	17.7 ± 8.5	59.0	1.7
Moist Deciduous (Degraded)	1.5 ± 3.7	9.6	11.1
Dry Thorn	7.5 ± 6.7	12.5	8.3
Moist Deciduous	4.1 ± 7.0	9.9	11.1
Total Area	90.3 ± 53.4	27.4	3.7

Mudumalai Wildlife Sanctuary, Western Ghats between 1989 and 2002

Table 3-2: Parametric paired t tests of temporal pattern of area under fire between vegetation types in the Mudumalai wildlife sanctuary

Pair	Mean Difference	t value	df	p value
Dry deciduous: Dry deciduous (<i>Shorea</i>)	-0.25	-5.42	13	< 0.001
Dry Deciduous:Moist deciduous (Degraded)	.24	4.43	13	< 0.01
Dry deciduous: Dry Thorn	0.21	3.83	13	< 0.01
Dry deciduous: Moist deciduous	0.23	4.7	13	< 0.001
Dry deciduous (<i>Shorea</i>): Moist deciduous (Degraded)	0.49	6.25	13	< 0.001
Dry Deciduous (<i>Shorea</i>): Dry Thorn	0.46	5.5	13	< 0.001
Dry Deciduous (<i>Shorea</i>): Moist deciduous	0.49	6.5	13	< 0.001
Moist deciduous (Degraded) and Dry Thorn	-0.02	-0.42	13	> 0.1
Moist deciduous (degraded) and Moist deciduous	-0.002	-0.063	13	> 0.1
Moist deciduous and Dry Thorn	0.026	0.49	13	> 0.1

Table 3-3: Non Parametric paired t tests of temporal pattern of area under fire between vegetation types in the Mudumalai wildlife sanctuary

Pair	Mann-Whitney U	Wilcoxon W	Z	p value
Dry deciduous: Dry deciduous (<i>Shorea</i>)	49	154	-2.25	< 0.05
Dry Deciduous:Moist deciduous (Degraded)	24	129	-3.4	< 0.01
Dry deciduous: Dry Thorn	35	140	-2.89	< 0.01
Dry deciduous: Moist deciduous	28	133	-3.2	< 0.01
Dry deciduous (<i>Shorea</i>): Moist deciduous (Degraded)	16	121	-3.7	< 0.001
Dry Deciduous (<i>Shorea</i>): Dry Thorn	10	115	-4.04	< 0.001
Dry Deciduous (<i>Shorea</i>): Moist deciduous	9	114	-4.08	< 0.001
Moist deciduous (Degraded) and Dry Thorn	49	154	-2.2	< 0.05
Moist deciduous (Degraded) and Moist deciduous	59	164	-1.8	< 0.1
Moist deciduous and Dry Thorn	69	174	-1.3	> 0.1

Figure 3-1: Map of the Western Ghats showing study regions

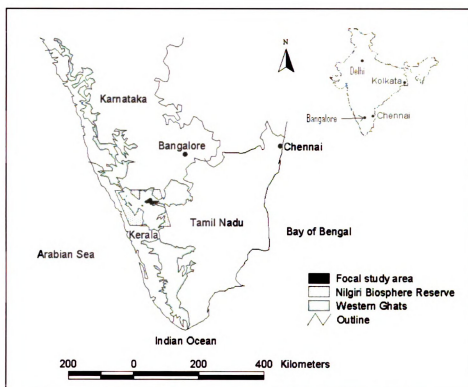


Figure 3-2: Vegetation map of the Mudumalai wildlife sanctuary

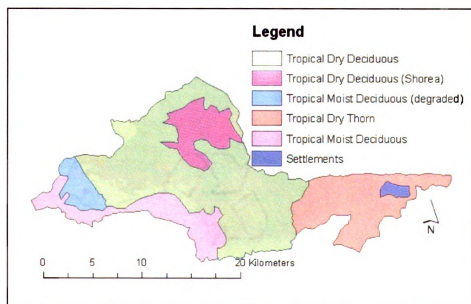


Figure 3-3: Temporal variability in the area burned across the different vegetation types in the Mudumalai Wildlife Sanctuary, Western Ghats

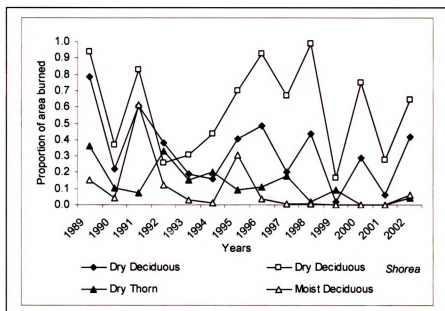


Figure 3-4: Fire-return intervals in the landscape of the Mudumalai Wildlife Sanctuary, Western Ghats

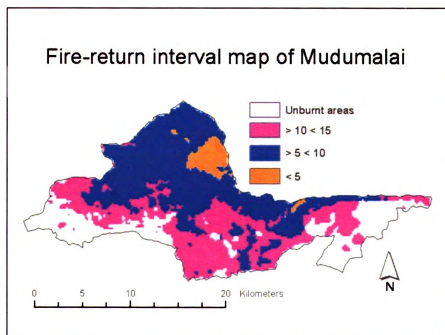
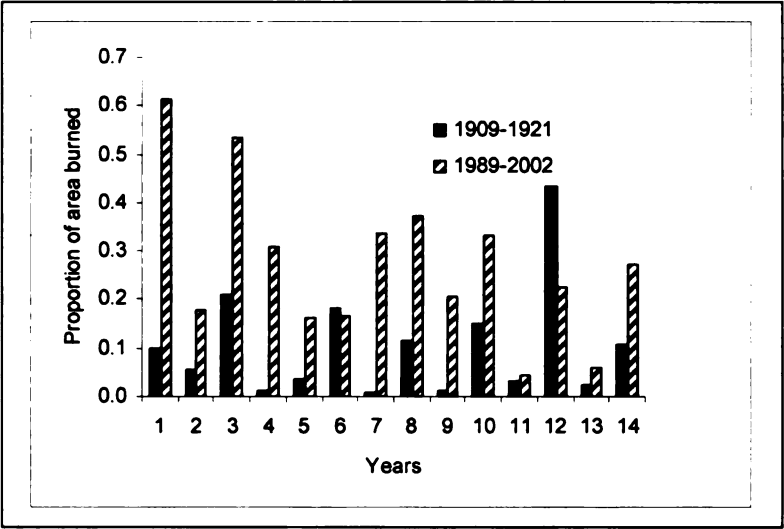


Figure 3-5: Temporal variability of forest fires in the Mudumalai Wildlife Sanctuary, Western Ghats during the 1910s and 1990s



Chapter 4

Forest fires: Spatial and temporal characteristics

Introduction:

The incidence of forest fires in landscapes varies around the globe, while in certain ecosystems the incidence of fires are natural (Whelan 1995), fires in certain other ecosystems are mainly as a consequence of the human presence within and around these forest ecosystems (Kodandapani et al. 2004). Over centuries, India's rural communities have created extensive savannas bordering their farmlands through extraction of woody biomass, grazing by livestock, and annual dry season fires (Gadgil 1993). Studies in the Amazon forests have shown the relationship between fire frequency and the size of forest patches, further these studies have shown the increase in fire-return intervals as function of distance from the forest edge (Cochrane 2001; Cochrane and Laurance 2002). Apart from the relationship between fire-return intervals as a function of distance from forest edge, fires in a landscape depend on a number of landscape characteristics. Fuel, weather and topography drive the behavior of an individual fire (Brown and Davis 1973; Pyne et al. 1996). Macroclimate is less variable temporally and spatially at landscape scales, affected by phenomena such as ENSO and orographic effects; whereas microclimate at the scale of a forest patch is more variable both temporally and spatially. Abiotic factors such as firebreaks, slope, aspect, elevation are spatially variable but temporally constant (Grimm 1984). Biotic factors such as fuel load varies with time, specifically with time since previous fire, and fuel composition varies with vegetation type. Complex models have been derived to explain the spread of fire in landscapes and the behavior of fire (Rothermel 1972; Finney 1998). The spatial pattern of fires over time in the Western Ghats have not

been clearly understood, the relationships between fire and forest boundaries have not been understood, further the relationship between factors such as fuels, climate, and topography have not been assessed.

This chapter describes the spatial pattern of fires in a landscape that harbors the largest contiguous forest area in the Nilgiri Biosphere Reserve (NBR), the region is home to three wildlife sanctuaries, the Mudumali wildlife sanctuary, the Bandipur wildlife sanctuary and the Wynaad wildlife sanctuary, fig 4-1a, hereafter referred to as Nilgiri landscape. The area of the NBR is 5520 km² of which the Nilgiri landscape constitutes 1545 km². Human presence within the forest areas is almost negligible, however the landscape is surrounded by human settlements. The landscape thus offers an unique opportunity to assess the incidence of forest fires as a function of distance from edge of park boundaries. It is unique because of the contiguity of the forests as well as the close juxtaposition with human settlements along most of the park boundaries, however the situation is made possible only when you combine the three wildlife sanctuaries into a single legal unit. Individually, these wildlife sanctuaries have atleast one of their boundaries at a significant distance away from human settlements. Another unique attribute associated with the landscape is that these three wildlife sanctuaries are spread across three different states in southern India, namely Karnataka, Tamil Nadu, and Kerala. The landscape is also extremely diverse in terms of the rainfall pattern, topography, and edaphic factors, which is reflected in vegetation types including the tropical moist deciduous forest ecosystems, tropical dry deciduous forest ecosystems, and the tropical dry thorn ecosystems, fig 4-1b. Detailed information on the other salient aspects of the Nilgiri landscape and the three wildlife sanctuaries are provided in the section on study areas in chapter 1.

I conduct the following analysis at two spatial scales, first I assess the relationship of forest fires as a function of distance from park boundary in the Nilgiri landscape (combined area of the three wildlife sanctuaries, 1545 km²) and follow this up with detailed modeling of the occurrence of fire in the Mudumali wildlife sanctuary (one sanctuary, 321 km²), partly because it encompasses all the three vegetation types, also because of the presence of a relatively longer (1989-2002) dataset on the spatial distribution of fires in these forests.

Objectives:

1. Determine the current fire-return interval in the Nilgiri landscape and to model the relationship as a function of distance from park boundary.
2. Assess measures of temporal and spatial autocorrelation within the Mudumalai wildlife sanctuary.
3. Determine the relative importance of topography, biomass, and climate in explaining fire occurrence in the Mudumali wildlife sanctuary, through the application of weighted logistic regression analysis.
4. Application of universal kriging to predict fire probabilities in the Mudumalai landscape.

Hypothesis testing:

Null hypothesis: *Forest fire occurrence in the landscape is random and there is no relationship between the spatial pattern of fire and the park boundary, further there is no association between landscape variables such as aspect, elevation, slope, rainfall, and biomass and the occurrence or spread of fires in a forest landscape.*

Hypothesis 1: *Fire occurrence in the landscape is spatially dependent on factors such as distance from forest edge, fire-rotation intervals are short close to forest edge and decline with distance from the edge (Cochrane 2001; Cochrane and Laurance 2001). The spatial pattern of fire regimes in human dominated landscapes is influenced by the abilities of humans to alter land cover in the landscape (Johnson et al 1998).*

Hypothesis 2: *The spatial pattern of forest fires is significantly dependent on the characteristics of the landscape (Pyne 1996; Wheelan 1995). The occurrence of fire in a landscape is a function of the fuel, weather, and topography (Brown and Davis 1973; Agee 2000). Landscapes that are South and southwest oriented have a greater probability of fire occurrence due to the longer exposure to solar radiation (Pyne et al. 1996). Ignition and fire occurrence depends on the fuel moisture content, which in turn is affected by the rainfall pattern in the landscape (Finney 1998). Fuel loads and fuel composition especially fine fuels are critical to the spread of fires in landscapes (Rothermel 1972; Agee 2000; Wells et al. 2004).*

Hypothesis 3: *Predicting fire occurrence and spread in a landscape are affected by spatial and temporal autocorrelation that arise due to interactions between forests of similar topography, climate, and fuel composition due to their spatial proximity and previous fire status. Time since last fire and the spatial interaction between landscape variables and fire are important factors in influencing fuel loads, and flammability of forests (Wells et al. 2004; Finney 1998).*

Forest Fire Analysis in the Nilgiri landscape:

Fire maps for these three wildlife sanctuaries have been developed from satellite data. IRS satellite imagery was used to classify the burnt and unburnt forest areas. Data from 1996, 1997, 1999, 2001, 2002, 2004, and 2005 were applied in the analysis. The data was subjected to preliminary processing such as radiometric, atmospheric, and geometric corrections, details provided in chapter 2. Figures 4-2a-g show the fire maps in the Nilgiri landscape.

Spatial analysis of fire pattern in the three wildlife sanctuaries:

Data from the seven years were combined into a fire-return interval (FRI) map (fig 4-3a). The fire pattern revealed by this map indicates numerous areas with short (> 10 yr) FRI close to the Nilgiri boundary, proximal to the human settlements that dot the boundary. There were areas with declining FRI in the interior forest areas, away from the park boundary; and intermediate FRI areas in between. Of the seven years of analyses of fire in the landscape, in years 1996 and 2002, there might be a semblance of the fire progressing from the boundary to the core, in 2004 and 2005 the fire has not progressed to the core, and the remaining three years (1997, 1999, 2001) fire occurrence at the core has no continuity with the periphery of the study area (refer to fire maps).

Modeling FRI as a function of distance from park boundary:

I investigated the spatial pattern of FRI further by performing a buffer analysis on the entire study area. I generated buffers of 1 km from the boundary of the study area upto a distance of 12 kms from the park boundary, by applying methods of buffer creation in ArcInfo (2002) software (fig 4-3b). I plotted the fire-return interval as a

function of distance from park boundary. Unlike in other areas across the globe, the relationship is depicted by long fire-return intervals close to the forest edge, and decreasing with distance from edge (Cochrane and Laurance 2002). The fire-return interval is long (> 10) upto a distance of 5 km. I modeled the relationship, through ordinary least squares, the linear model has an R-square 0.3, the estimate of the coefficient of the buffer distance was -0.43, with F value of 4.3 and a corresponding p value of 0.06 (fig 4-4). Table 4-1 provides the model results and comparison of the different models and their associated statistical values. I also fitted a second order polynomial equation to the fire-return interval as a function of distance from the park boundary and this shows that fire-return interval declines to a distance of 5-10 km. I also fitted cubic and power models to the data, the cubic model provided a higher R-square value compared to the power model. I modeled the relationship to a distance of 6 km from the park boundary to overcome the potential problem of overlapping burnt areas in the core, here the R-square value was much higher for all the models, linear, quadratic, cubic, and power.

Pairwise comparison of means:

I compared the proportion in each fire frequency class across the different buffer distances by performing pairwise t tests. All comparisons were statistically significant at $p < 0.05$. I investigated the bimodal pattern of fire frequency as a function of distance from the park boundary further, I found that in four of the seven fire frequency classes, the proportion of area within each class increased initially, then stabilized, followed by an increase again as a function of distance from the edge (fig 4-5). There is a larger proportion (0.3) of the burnt area in fire frequency class one, close to the park boundary, between 1-5 km from the boundary, and (0.5) at a distance

of 9 km from the boundary, the higher proportion could be partly because of the declining buffer area from the park boundary. The area under the different buffers declines gradually away from the park boundary (fig 4-6), I computed the area within each fire frequency class within each buffer. Similarly the proportion of area in fire frequency three, about (0.1) of the buffer burns at distances of 1-7 km and here again there is a rise in the proportion of area burnt to (0.2) at a distance of 10 km from the park boundary. In the North and East of the Nilgiri landscape, the tropical dry thorn forest prevails, here fuel loads are typically low, however in the West and South of the Nilgiri landscape the tropical moist deciduous forest prevails, here fuel loads maybe higher than the tropical dry thorn forest type, but mean annual rainfall is higher. Interestingly the 5 km distance from park boundary could be the transition from either of these two vegetation types into the tropical dry deciduous forests, where fuel loads are typically high and also rainfall intermediate (refer chapter 5 for fuel load information). I conducted students t tests between the pairs and found significant differences in the proportion of area burnt between the fire frequency classes with $p < 0.05$. This property of the fire frequency points to landscape characteristics that are critical to fire spread, which I shall explore in the following sections.

Modeling fire occurrence in the Mudumalai wildlife sanctuary:

In order to assess fire occurrence in greater detail, in relation with other variables such as topography, climate, and biomass, I conducted spatial regression analysis. I modeled fire spread in the Mudumalai wildlife sanctuary because of the availability of uninterrupted spatial and temporal data of fire occurrence.

Phenomenon such as forest fires and other disturbances in ecosystems are complex and depend on a number of variables in the ecosystem. Predicting the occurrence of forest fires in ecosystems is confounded by spatial and temporal behavior of fires in ecosystems and precludes the use of classical regression analysis in predicting fire spread.

A class of models known as the marginal logistic regression model however permits the presence of spatial and temporal correlation and the use of covariates in predicting the occurrence of an event (Diggle et al. 1994). The model focuses on the probability of an event and the relationship between the explanatory variables and the probability of the occurrence of fire.

Marginal models and logistic regressions:

I developed a marginal logistic regression model for spatially correlated proportions. Unlike the ordinary linear model, the marginal logistic regression model has a logistic form and also permits the use of variance to correct for temporal and spatial autocorrelations, thereby increasing the accuracy of the predictions. The advantage of this model is the application of proportions as dependent variables or variables that have a binary response. The next few lines provide information on generalized estimating equations (GEE) approach for modeling the effect of spatial location and subject-specific covariates on spatially correlated binary data. The interest is on making inferences on the marginal mean structure in the presence of this spatial correlation.

Model (adapted from Albert and McShane 1995)

Let $Y_i(\mathbf{s})$ denote the binary response for subject i at spatial location $\mathbf{s}$, and let $X_i(\mathbf{s})$ denote the corresponding vector containing spatial location information and subject-specific covariates.

I relate both spatial location and subject-specific covariates to marginal response frequency, denoted by $P_i(\mathbf{s}) = P(Y_i(\mathbf{s}) = 1)$, where $i = 1, 2, 3, \dots, K$ is an index of the K subjects, n is the number of spatial observations on each subject, and $\mathbf{s}_1, \mathbf{s}_2, \dots, \mathbf{s}_n$ are two-dimensional vectors containing the spatial locations at which observations are made. The vector of spatially correlated binary random variables for the i th subject is represented by $\mathbf{Y}_i = (Y_i(\mathbf{s}_1), Y_i(\mathbf{s}_2), Y_i(\mathbf{s}_3), \dots, Y_i(\mathbf{s}_n))'$ with the mean vector $\mathbf{P}_i = (P_i(\mathbf{s}_1), P_i(\mathbf{s}_2), \dots, P_i(\mathbf{s}_n))'$.

A GEE approach for modeling the marginal mean structure is proposed by a number of authors (Zeger and Liang 1986; Wedderburn 1974). This approach is a multivariate extension of the quasi-likelihood model, where we parameterize the marginal mean structure as well as the variance structure. For the mean model, I assume

$$h(P_i(\mathbf{s}_j)) = \mathbf{X}_i(\mathbf{s}_j)\boldsymbol{\beta} \quad \text{or} \quad P_i(\mathbf{s}_j) = h^{-1}(\mathbf{X}_i(\mathbf{s}_j)\boldsymbol{\beta}),$$

Where h is a link function, and $\boldsymbol{\beta}$ is a $p \times 1$ dimensional vector of unknown parameters describing the effect of spatial location and subject-specific covariates on the mean.

The link function that I have applied is the logit link function ($h(P) = \text{logit}(P)$)

The covariance structure of $\mathbf{Y}_i$ denoted by $\mathbf{V}_i = \text{Var}(\mathbf{Y}_i)$ can be expressed as

$$\mathbf{V}_i = \mathbf{A}_i^{1/2} \mathbf{R}(\boldsymbol{\alpha}) \mathbf{A}_i^{1/2}$$

Where $\mathbf{A}_i = \text{diag}[P_i(\mathbf{s}_1)(1 - P_i(\mathbf{s}_1)), P_i(\mathbf{s}_2)(1 - P_i(\mathbf{s}_2)), \dots, P_i(\mathbf{s}_n)(1 - P_i(\mathbf{s}_n))]$, and the spatial correlation is specified via the $n \times n$ correlation matrix $\mathbf{R}(\boldsymbol{\alpha}) = \mathbf{Corr}(\mathbf{Y}_i)$, where $\boldsymbol{\alpha}$ is a vector of parameters characterizing the correlation structure. The semivariogram is a useful construct for parameterizing the spatial correlation and is given below in equation (10).

Estimation of the betas

GEE were employed to estimate the parameters of the mean as well as those of the correlation structure. The generalized estimating equation for the mean parameters $\boldsymbol{\beta}$ are given by

$$\sum_{i=1}^K \mathbf{D}_i' \mathbf{V}_i^{-1} (\mathbf{Y}_i - \mathbf{P}_i) = 0 \text{-----(1)}$$

Where $\mathbf{D}_i = \delta \mathbf{P}_i / \delta \boldsymbol{\beta}$ (Zeger and Liang 1986).

Let $\mathbf{Z}_i$ be the sample covariance matrix on the i th subject with (j, k) -th element given by

$$Z_i(j, k) = (Y_i(\mathbf{s}_j) - P_i(\mathbf{s}_j))(Y_i(\mathbf{s}_k) - P_i(\mathbf{s}_k)),$$

And let $\mathbf{z}_i$ and $\mathbf{v}_i$ be the vectors on $n(n - 1)/2$ elements consisting of all entries below the diagonal of the symmetric matrices $\mathbf{Z}_i$ and $\mathbf{V}_i$, respectively. The generalized estimating equations for the correlation model parameters, $\boldsymbol{\alpha}$, are given by

$$\sum_{i=1}^K \mathbf{E}_i' \mathbf{W}_i^{-1} (\mathbf{z}_i - \mathbf{v}_i) = 0 \text{-----(2)}$$

Where $\mathbf{E}_i = \delta \mathbf{v}_i / \delta \boldsymbol{\alpha}$ and $\mathbf{W}_i$ is the working variance matrix of $\mathbf{z}_i$ (Prentice, 1988). As suggesting by Prentice (1988), $\mathbf{W}_i = \text{diag}(w_{i21}, w_{i31}, w_{i41}, \dots)$, where

$$w_{ijk} = \text{var}(Z_i(j, k)) = (1 - 2P_i(s_j))(1 - 2P_i(s_k))v_{ijk} + v_{ijj}v_{ikk} - v_{ijk}^2,$$

And where v_{ijk} is the (j, k) -th element in V_i .

Starting with initial estimates of β , and $\alpha(\beta_0$, and $\alpha_0)$, equations 1 and 2 can be solved by iterating between modified scoring algorithms for β and α .

First, given α_m , an update β_{m+1} is given by

$$\beta_{m+1} = \beta_m + \left[\sum_{i=1}^K \hat{D}_i' \hat{V}_i^{-1} \hat{D}_i \right]^{-1} \sum_{i=1}^K \hat{D}_i' \hat{V}_i^{-1} (Y_i - \hat{P}_i) \text{-----}(3)$$

Where $D_i = D_i(\beta_m)$, $V_i = V_i(\beta_m, \alpha_m)$, and $P_i = P_i(\beta_m)$. Second, given β_{m+1} , an update α_{m+1} is obtained by

$$\alpha_{m+1} = \alpha_m + \left[\sum_{i=1}^K \hat{E}_i' \hat{W}_i^{-1} \hat{E}_i \right]^{-1} \sum_{i=1}^K \hat{E}_i' \hat{W}_i^{-1} (\hat{z}_i - \hat{v}_i), \text{-----}(4)$$

Where $E_i = E_i(\beta_{m+1}, \alpha_m)$, $W_i = W_i(\beta_{m+1}, \alpha_m)$, $z_i = z(\beta_{m+1})$, and $v_i = v_i(\beta_{m+1}, \alpha_m)$.

Estimates of β , and α are obtained by iterating between equations (3) and (4) until convergence ($m=1, 2, 3, \dots$).

The asymptotic properties of the parameter estimators for the mean and correlation structure follow the work of Liang and Zeger (1986), Zeger and Liang (1986), and Prentice (1988). The estimator β is consistent (fixed n , $K \rightarrow \infty$) even for misspecified spatial correlation structure (semivariogram model). The estimator α is consistent under a correctly specified mean and semivariogram model. If the semivariogram model is correctly specified, a model-based consistent estimator of the variance of β is

$$\text{var}_{\text{mod}}(\hat{\beta}) = \left[\sum_{i=1}^K \hat{D}_i' \hat{V}_i^{-1} \hat{D}_i \right]^{-1} \text{-----}(5)$$

As an alternative, the robust estimator of variance,

$$\text{var}_{\text{rob}}(\hat{\beta}) = \left[\sum_{i=1}^K \hat{D}_i' \hat{V}_i^{-1} \hat{D}_i \right]^{-1} \left[\sum_{i=1}^K \hat{D}_i' \hat{V}_i^{-1} Z_i \hat{V}_i^{-1} \hat{D}_i \right] \left[\sum_{i=1}^K \hat{D}_i' \hat{V}_i^{-1} \hat{D}_i \right]^{-1} \quad (6)$$

is consistent even when the semivariogram model is misspecified. Estimators of $\text{var}(\alpha)$ can be obtained as described in Prentice (1988).

Starting values for the mean structure parameters (β_0) can be obtained by assuming spatial independence and fitting a generalized linear model with a logit link function (logistic regression). The logistic regression model can be fit using GLIM (McCullagh and Nelder 1989), and it is identical to GEE under spatial independence. These estimates are consistent and have asymptotic variance.

$$\text{var}_{\text{indep}}(\hat{\beta}) = \left[\sum_{i=1}^K D_i' \Sigma_i^{-1} D_i \right]^{-1} \left[\sum_{i=1}^K D_i' \Sigma_i^{-1} V_i \Sigma_i^{-1} D_i \right] \left[\sum_{i=1}^K D_i' \Sigma_i^{-1} D_i \right]^{-1} \quad (7)$$

Where $\Sigma_i = A_i$ and V_i is the true spatial covariance structure.

In order for simplicity the logistic model can be rewritten as a general form of the mean,

$$\text{Logit}(p_i) = x_i' \beta \quad (8)$$

Where p_i is the probability of fire occurrence at the i th point, a model for the variances and covariances among the points. The explanatory variables x_i are measurements of different topographical and environmental variables at these points.

As suggested earlier, the above model permits the incorporation of explicit temporal and spatial autocorrelation in modeling the occurrence of fire in the Mudumalai

wildlife sanctuary. The explanatory variables that I have included in the model can be categorized into three broad classes, topographical, biophysical, and climatological.

Fire Frequency map of the Mudumalai wildlife sanctuary:

I generated the fire frequency map for the Mudumalai wildlife sanctuary as indicated in chapter 3. Figure 4-7a shows the FRI map of the wildlife sanctuary from 1989 to 2002. The fire frequency map contained 627 polygons, with frequency ranging from no fire (0 frequency) to 11 fires during the 14 year time period, I randomly sampled 150 points from this map. I conducted ordinary logistic regressions with the proportion of fire occurrence as the dependent variable and explanatory variables that included corresponding elevation, slope, aspect, mean annual rainfall, and forest fractional coverage. Ordinary logistic regression uses maximum likelihood to fit the logistic model for the mean in equation (8), however it assumes that the observations are uncorrelated and it also assumes that the response variable (fire proportion) is a binomial random variable with variance $p_i(1-p_i)/n$, where $n=14$ yr and p_i =probability of fire occurrence at point i .

The explanatory variables:

Elevation: The elevation ranges from 470 to 1251 m asl. The Moyar gorge falls of steeply to low elevation values in the North of the Mudumalai wildlife sanctuary (fig 4-8a). There are two areas in the Mudumalai landscape where elevations are higher than 1000 m asl. One in the eastern part called Narlae betta, and another in the northwest.

Aspect: A large part of the study area has South and southwest facing aspects. The aspect ranges from 0 to 360 degrees (fig 4-8b). I transformed the aspect data such

that the North was at 20 degrees and South at 200 degrees. I obtained the absolute value of each aspect data point after subtracting 180 and set the aspect range from 1 to 180 degrees.

Slope: Slopes are gentle for most of the landscape, however in the Moyar gorge, the landscape falls precipitously leading to high slopes. The slope ranges from 0 to 45 degrees (fig 4-8c).

Forest fractional coverage: Forest fractional coverage was obtained from Landsat ETM+ data as described in chapter 2. The forest fractional coverage is high on the western side, in the tropical moist deciduous forests, intermediate in the dry deciduous forests, and low in the tropical dry thorn forests, however in certain areas in the dry thorn forests values are high due to the influence of grasses and shrub leaves in these forests (Wang et al. 2005), figure 4-8d.

Mean Annual Rainfall: The climate variable includes rainfall obtained from a ordinary kriging surface of a set of points obtained from data of 347 rainfall stations for the Nilgiris (Lengerke 1977). The Nilgiris is the largest and highest mountain massif, which connects to a number of distinct topographic features, in the North it is connected to the Mysore plateau and the Nigiri-Wynaad region, to the East are the Coimbatore plains, to the South is the Silent valley, and to the West are the Naduvattam hills and Nilambur plains. The data are annual means over a 30 year time period from 1941 to 1970, data ceased to be collected beyond this point in time at these locations.

Kriging and variogram of rainfall data for the Nilgiris

The omnidirectional variogram for the rainfall data was obtained. A spherical model was fitted to the variogram. The variogram has a nugget of 100000 and a psill of

1050000. A nugget:sill ratio of 0.095 is obtained suggesting that about 10% of the data are random or some degree of independence is present in the data. But there is clear spatial dependence at distances up to 60 km and the remaining 90% of the data has a spatial structure. I applied ordinary kriging on the rainfall data. Values for the prediction surface range from 400 to 6000 mm and the standard errors from the global ordinary kriging estimate ranged from 381 to about 934 mm.

I extracted the rainfall pattern for the Mudumalai wildlife sanctuary from the ordinary kriging surface (fig 4-8 e). I compared the predicted values to a set of values obtained from the 1990s. The predicted and actual values are plotted in (fig 4-9), the plot shows the percentage error in the predicted rainfall at each point, with reference to the observed rainfall data; the standard errors range from 0.04 to 1.

Estimation of deviance residuals: To evaluate the fit of the model, I computed the deviance residuals, which is given by the following formula (Collett 1991).

$$d_i = \sqrt{2n_i Y_i \log(Y_i / p_i) + 2(n_i - n_i Y_i) \log\{(1 - Y_i) / (1 - p_i)\}} \text{-----(9)}$$

Where

d_i is the deviance residual

Y_i is the observed fire occurrence

p_i is the predicted fire occurrence

$n = 14$ yr

Next I obtained the standardized deviance residuals (Collet 1991), this is the deviance residual divided by its asymptotic standard error.

I computed an empirical semivariogram from the standardized deviance residuals, d_i (Cressie 1991).

$$\gamma(h) = 1/2N(h) \sum_{i,i^*}^{N(h)} (d_i - d_{i^*})^2 \text{ -----(10)}$$

where h = distance between two points, $N(h)$ is the number of pairs of points h meters apart, and the subscripts i and i^* indicate two different points that are h units apart. The variogram of the deviance residuals shows that the points of fire occurrence are spatially correlated to a distance of 7 km. The sill of the variogram estimates the variance of the response variable (Gumpertz et al 2000). Since the residuals are standardized the sill of the variogram should be close to 1. A sill much greater than one would be an indication that there is overdispersion in the dataset and one possible reason for that is temporal autocorrelation in the dataset (Gumpertz et al. 2000).

Variograms and over dispersion

I fitted an exponential model to the variogram, shown in fig 4-10, the estimate of the sill is 1.1 and the nugget is 0.2. The nugget effect is 18%. This suggests that there is some degree of independence or random noise in the data. The exponential model fitted to the empirical variogram is

$$\gamma(h) = 0.2 + 0.9(1 - e^{-3h/7000})$$

The above semivariogram of standardized deviance residuals shows that there is spatial correlation upto a distance of 7 km, beyond this distance fire occurrence is uncorrelated.

In an exponential model the range is the distance at which the semivariogram is 95% of the sill. Since the sill in the semivariogram is 1.1, the results suggest that the overdispersion in the data is negligible and spatial autocorrelation is important more than temporal autocorrelation in the study of fire in the study area, (fig 4-10).

Temporal autocorrelation analysis of forest fires:

I determined the autocorrelation coefficient applying the following formula, the formula provides an estimate for the coefficient under a first-order Markov model. The model can be changed to provide second and third order models.

$$\rho_i = \left[\frac{1}{n_i - 1} \sum_{j=1}^{n_i-1} (U_{ij} - Y_i)(U_{i,j+1} - Y_i) \right] / Y_i(1 - Y_i) \text{-----(11)}$$

Where:

ρ_i is the autocorrelation coefficient

U_{ij} is the fire occurrence (0 or 1) in polygon i in year j

Y_i is the observed fire occurrence

$n = 14$ yr

I examined the likelihood of a fire event based on previous fire occurrences in a polygon. I computed a first order, second order, and third order Markov estimate.

The results suggest that for a first order Markov process, probability of fire occurrence at a polygon depends on the fire status only in the previous year, the autocorrelation coefficients in the tropical dry deciduous forest ranged from -0.86 to 0.53, with 50% of the polygons having autocorrelation coefficients > -0.11 ; in the tropical moist deciduous forest the temporal autocorrelation coefficients ranged from -0.35 to 0.24, with 50% of the polygons having autocorrelation coefficients > -0.08 ; in the tropical dry thorn forest the temporal autocorrelation coefficients ranged from –

0.35 to 0.69, with 50% of the polygons having autocorrelation coefficients > -0.08. For a second order Markov process, probability of fire occurrence depends on fire status in the previous two years, the autocorrelation coefficients in the tropical dry deciduous forest ranged from -0.26 to 0.23, with 50% of the polygons having autocorrelation coefficients > 0.017; in the tropical moist deciduous forest autocorrelation coefficients ranged from -0.12 to 0.054, with 50% of the polygons having autocorrelation coefficients > 0.0008; in the tropical dry thorn forest autocorrelation coefficients range from -0.31 to 0.16, with 50% of the polygons having autocorrelation coefficients > 0.01. For a third order Markov process, probability of fire occurrence depends on fire events in the previous three years, the autocorrelation coefficients in the tropical dry deciduous forest ranged from -0.13 to 0.17 with 50% of the polygons having autocorrelation coefficients > 0.0; in the tropical moist deciduous forest the autocorrelation coefficients ranged from -0.009 to 0.075, with 50% of the polygons having autocorrelation coefficients > -0.0007; in the tropical dry thorn forest autocorrelation coefficients ranged from -0.08 to 0.096, with 50% of the polygons having autocorrelation coefficients > -0.001.

Incorporating Spatial Autocorrelation

I use weighted logistic regressions to incorporate spatial autocorrelation within the model of fire occurrence in the Mudumalai wildlife sanctuary. I computed the correlation parameters from the semivariogram between points

$$R_{i,j} = \text{CORR}(Y_i, Y_j) = \frac{C_1}{C_0 + C_1} * e^{(-3h/a)} \text{-----}(12)$$

Where

C_0 is the nugget; commonly called the nugget effect; a is the range, which provides a distance beyond which the variogram or covariance value remains essentially

constant, $C_0 + C_1$, commonly called the sill, which is the variogram value for very large distances, $\gamma(\infty)$. It is also the covariance value for $|h| = 0$, and the variance of the random variables (Isaaks and Srivastava 1989; Bailey and Gatrell 1995).

I applied the correlations obtained in the above equation as a weight and performed a weighted logistic regression analysis. The new variance of Y_i now incorporated in the model is

$$p_i(1-p_i)/n_i * (1 + \text{TOTCORR}_i/n_i).$$

Where TOTCORR_i is the total correlation obtained from the covariances of 150 points from variogram modeling.

This is computed in SAS by specifying the weights as a new variable in Proc GENMOD (SAS 2005), I have regressed Y_i on the explanatory variables and specified weights $w_i = 1/(1 + \text{TOTCORR}_i/n_i)$, where w_i is a multiplier for the inverse of the variance (Gumpertz et al. 2000). Figure 4-11 shows the variogram after correction for spatial autocorrelation.

Parameterization of variables

Table 4-2 gives the five variables and their statistical significance after the weighted logistic regression analysis, from the p values it is clear that forest fractional coverage, elevation, and mean annual rainfall are the most important variables in predicting fire occurrence in the Mudumalai wildlife sanctuary. The slope and aspect are not statistically significant in explaining the fire occurrence in the study site. The probability of fire occurrence in the wildlife sanctuary decreases with increase in rainfall, fire occurrence with reference to forest fractional coverage is mainly

restricted to the intermediate levels, that is forest fractional coverage values ranging from 40 to 60%, which corresponds well with the tropical dry deciduous forest ecosystem. At high forest fractional coverage in the tropical moist deciduous forest rainfall is also high, in the dry thorn forests fuels are discontinuous. Further topographical parameters such as elevation emerge as significant predictors of fire in the Mudumalai wildlife sanctuary. Fire probability increases with elevation; this could be one of the reasons for the short FRI in the North of the sanctuary.

Predicting fire occurrence in the Mudumalai landscape

The last objective of assessing fire in the landscape is to predict fire occurrence in the landscape. I performed Universal kriging (UK). UK is a useful prediction tool when there exists a broad first order trend in the dataset. In UK the trend is removed from the dataset by obtaining a variogram model of the covariance structure of the residuals of the trend surface. However when performing the UK the trend is refitted into the prediction surface. The kriging surface in this case provides an estimate of the probability of fire occurrence at a location for any given year. The universal kriging surface provides the probability of fire occurrence at a given location for a particular year.

Removal of first order trend and Universal kriging

Since the data showed a broad first order trend (fig 4-12) I removed it by fitting a trend surface to the dataset. Table 4-3 shows that the x and x^2 are statistically significant and the R-square value is 0.42 and the F value is significant at $p < 0.001$.

The exponential model obtained from the residuals of the trends surface has a nugget of 0.009 and a sill of 0.02, with a range of 5 km (fig 4-13). I applied this model to obtain the universal kriging surface of fire probability in the Mudumalai landscape.

Fig 4-14 shows the universal kriging surface of the fire probabilities in the Mudumalai landscape. The map reveals low fire probabilities in the East, West, and South of the landscape. Higher probabilities occur in the North of the study area. The predicted surface is smoother than the actual data, however it honors actual data, the overall pattern of fire probability in the landscape is captured by the kriging estimate. The kriged surface reveals a few values below zero, this is an artifact of the universal kriging process. A large part of the moist deciduous (57.17%) and dry thorn (48.5%) forests have not burnt even once in the 14 year time period, in contrast 100% of the dry deciduous forests have burnt one or more times in my analyses. In addition to this predominance of fires in the dry deciduous forests, fires in the other two forest types are mostly restricted to the ecotone areas, thus parameterization and predictions of fire spread in the Mudumalai landscape by including the three vegetation types in a single model would not significantly alter the results.

Model diagnostics

The universal kriging variance range from 0.012 to 0.02 with a median value of 0.01458 and a mean of 0.01481, figure 4-15 shows the kriging standard error surface for the Mudumalai landscape. The map of the kriging standard error shows that the predictions become less reliable where sampling is sparse, that is, near the edges of the study area. Figure 4-16 shows the cross validation map of the fire residuals, some of the errors are large, the largest was 0.42. However half the errors lie between –

0.05 and 0.059. The median value for zscores was -0.05 about half the values lie between 0.39 and 0.49. The RMSE value was 0.124. Finally I checked for autocorrelation in the residuals of the UK predictions (fig 4-17), there appears to be very little spatial structure, suggesting that the model is appropriate.

Discussion:

The linear model of FRI and distance steadily decreases with distance from park boundary, it provides an indication of the importance of landscape variables in driving the spatial pattern of fire. The bimodal pattern of the proportion of area under fire in the different fire frequency classes suggests ignitions originating from within the park boundary. The presence of relatively large proportion of area in fire frequency classes 3 & 4 at the center of the study area is an indication that landscape characteristics such as rainfall, topography, and fuels are important in fire spread in the Nilgiri landscape.

The contiguous forest area of the Nilgiri landscape is currently experiencing short fire-return intervals ($\text{FRI} < 4 \text{ yr}$). All models reveal a declining trend in the FRI as a function of distance from the park boundary. The linear model captures 30% of the variation in the fire-return interval, suggesting that factors other than the presence of human settlements along the fringes of the study area are important in determining the current spatial pattern of fire occurrence in the Nilgiri landscape. Part of this declining trend in the FRI can be attributed to the presence of the tropical moist deciduous forests and tropical dry thorn forests in the 1 – 5 km buffer classes, where fire occurrence is limited by climatic conditions as well as fuel loads and fuel composition. Within the 5 – 12 km buffer classes, the predominant vegetation type in

the tropical dry deciduous where the climate and fuel load, composition is favorable to the occurrence of fire.

The occurrence of fire at a given location is a function of the fuel load and its characteristics, the topographic features of the landscape, as well as the climate. A significant portion of the landscape in the deciduous forests have elevation values greater than 900 m asl, at these high elevations the landscape is subjected to increased drying, and curing of fuels occurs here. Thus the available fuels are greater at higher elevations. At higher elevations and in the deciduous forest trees are bereft of foliage, winds have a desiccating effect, increasing available fuels in the process (Freifelder et al. 1998).

The rainfall pattern in the landscape effects the fuel moisture content and hence fire occurrence and spread. Changes in fuel state can be caused by abrupt changes in weather, diurnal changes, seasonal changes, annual changes, and successional changes. In the Mudumalai landscape the second and third causes are relatively more important. The moisture content of fine fuels changes throughout the day in response to weather conditions, such as solar radiation, temperature and relative humidity. Seasonal changes in climate is another important factor in the landscape, the three most important months for fire are January, February, and March, this corresponds to the fire season, the total rainfall received during the fire season is only 8% of the mean annual rainfall for the Mudumalai wildlife sanctuary. Apart from the fuel state changes, fuel type changes in response to the season, deciduous leaf drop lead to increase of litter, curing of the grasses also takes place during this season (Agee 1993).

A significant part of the surface fuels in the Mudumalai landscape comes from grasses, both annual and perennial species. The presence of an overstorey canopy layer and the understorey grass layer leads to a reasonable assessment of biomass with fractional coverage. In the West of the Mudumalai landscape, high fractional coverage, compare with a closed canopy, with leaf litter and negligible grass fuels, and corresponds with no or very little fire, in the East low fractional coverage, compare with grass fuels and negligible leaf litter fuels, correspond to very little or no fire. In the North and central parts where fractional coverage is intermediate, it corresponds to high incidence of fire.

In the dry deciduous forests, fire dependence on time is sensitive to the previous year, whereas in the dry thorn and moist deciduous forests it is negatively (mean autocorrelation coefficient = -0.001) auto correlated and positively (mean autocorrelation coefficient = 0.0068) auto correlated respectively over a four-year period, although the strength of the coefficients are minute. Differences in fuel loads, fuel compositions and the decay rates (Sundarapandian and Swamy 1999) could be reasons for the temporal autocorrelation patterns in the three vegetation types. Although fuel loads vary with time since fire, this is important with respect to large fuel sizes (10 hr, 100 hr, and 1000 hr fuels), whereas with fine fuels and in the Mudumalai landscape, the deciduous nature of trees and the perennial character of the grass fuels might ensure fuel loads from the fine fuels (1 hr) are less variable from one year to the next. However it is clear that spatial autocorrelation is a significant parameter in assessing and predicting fire occurrence in this ecosystem. The weighted logistic regression model was able to reduce the extent of spatial autocorrelation in the prediction of the occurrence of fire in the study area. The range

of spatial continuity after the correction for spatial auto correlation is 3.5 km suggesting that fire spread is dependent on spatial effects due to similarities in topography, fuels, and climate at this scale.

Universal kriging provides a smooth surface of the fire probabilities in the Mudumalai landscape in a given year. The RMSE is low and the standard errors are also low for the entire study area. However at the edges of the study area errors are relatively high, here the numbers of points are few. The overall kriging surface captures the spatial pattern of forest fires in the landscape with reasonable accuracy as revealed in the cross validation exercise, except for a few points on the boundary of the study area, predicted values are close to the observed values.

Implications for conservation:

A significant area in India are under deciduous ecosystems, about 25% of the forests in the Western Ghats are classified as dry deciduous ecosystems, further within these forested areas, a number of wildlife sanctuaries and national parks have been created and areas set aside. It is reasonable to expect similar spatial patterns of fires in these forests, active conservation measures will be required to obviate the negative effects of these short-fire return intervals on the ecology of these forests (chapter 5). A number of wild animals such as the Asian elephant and other large mammals inhabit these forests (Sukumar et al. 2004), the current conservation strategies of merely delineating National parks and wildlife sanctuaries will require rethinking. A more holistic conservation strategy of fire management in the landscape, with reference to sources of ignitions, and landscape characteristics, biodiversity monitoring, and local community involvement in conservation is important.

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Figure 4-1 a. Location of Mudumalai, Wynaad and Bandipur wildlife sanctuaries
b. Vegetation types in the Nilgiri landscape

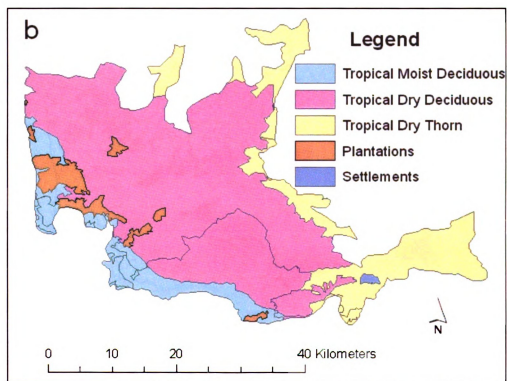
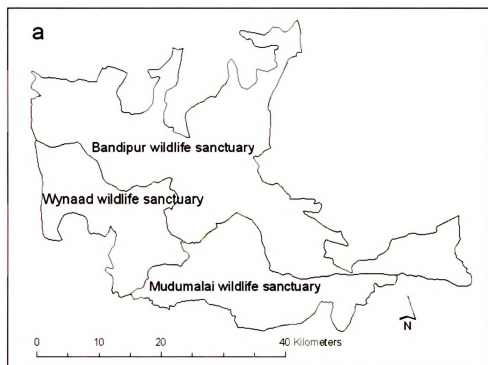


Figure 4-2 a. Fire map of Nilgiri landscape in 1996
b. Fire map of Nilgiri landscape in 1997

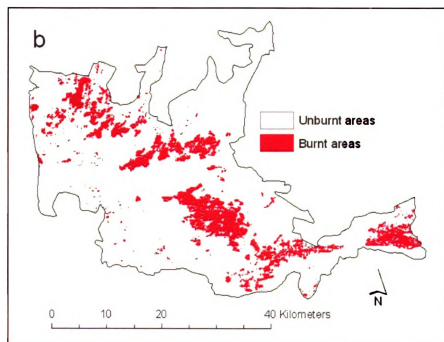
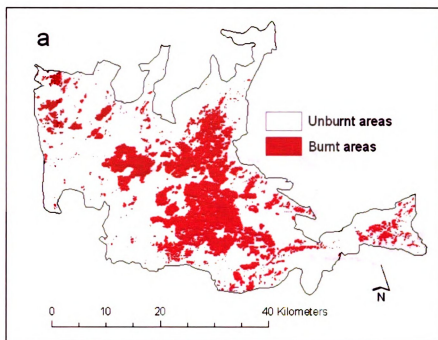


Figure 4-2 c. Fire map of Nilgiri landscape in 1999
d. Fire map of Nilgiri landscape in 2001

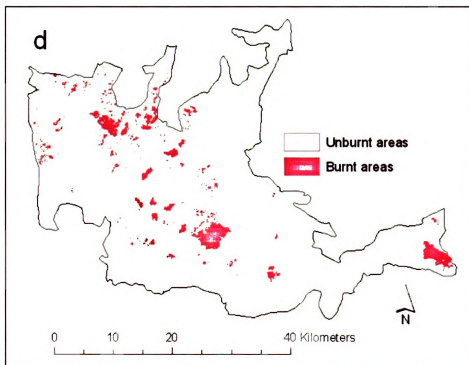
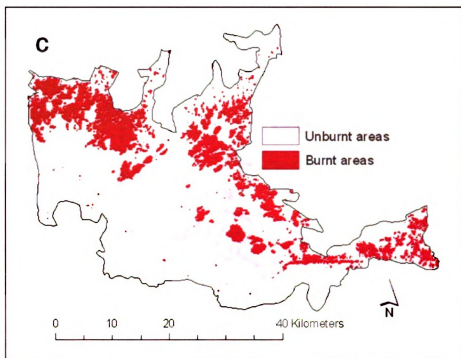


Figure 4-2 e. Fire map of Nilgiri landscape in 2002
f. Fire map of Nilgiri landscape in 2004

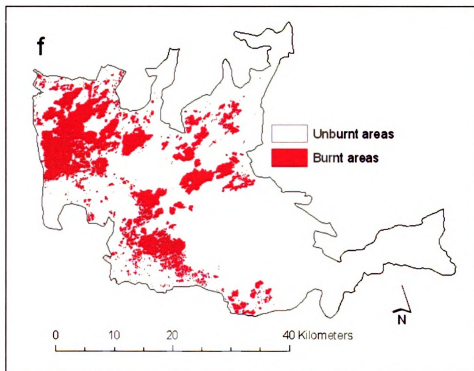
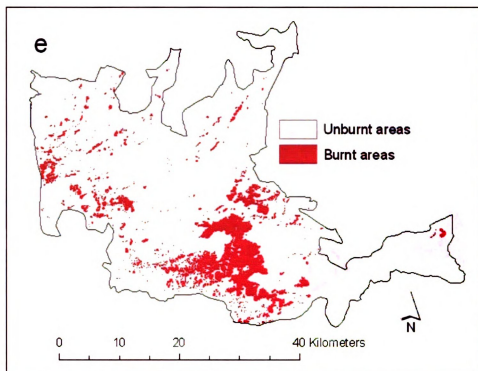


Figure 4-2 g. Fire map of Nigiri landscape in 2005

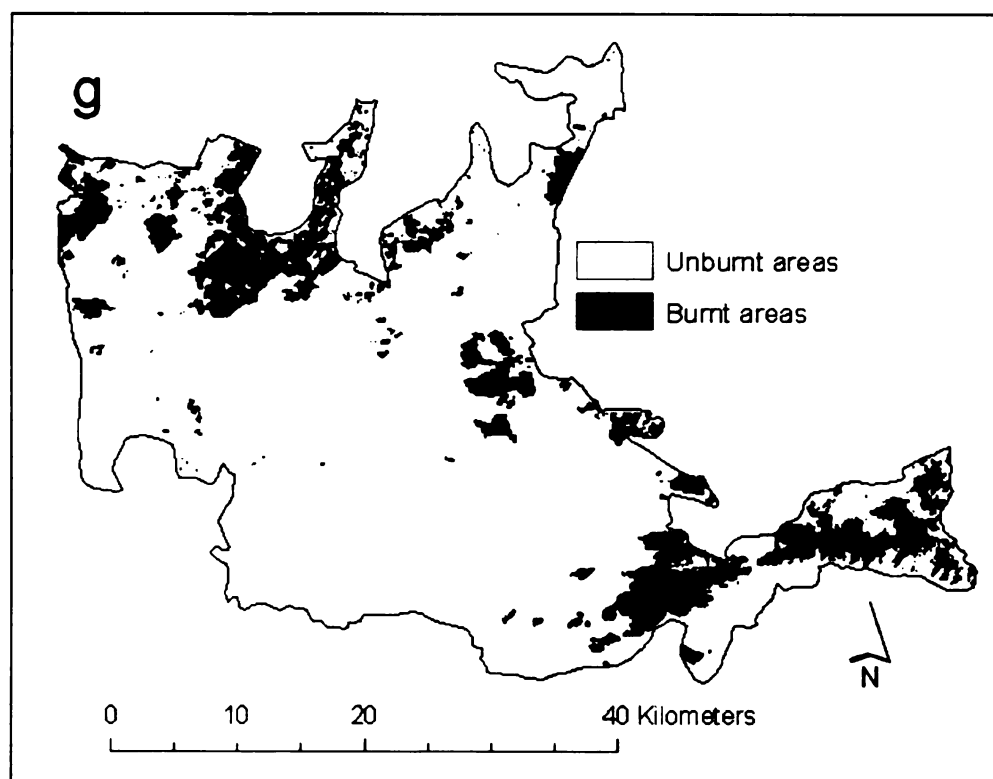


Figure 4-3 a. Fire-return interval of the Nilgiri landscape
 b. Buffer distances from the park boundary in the Nilgiri landscape

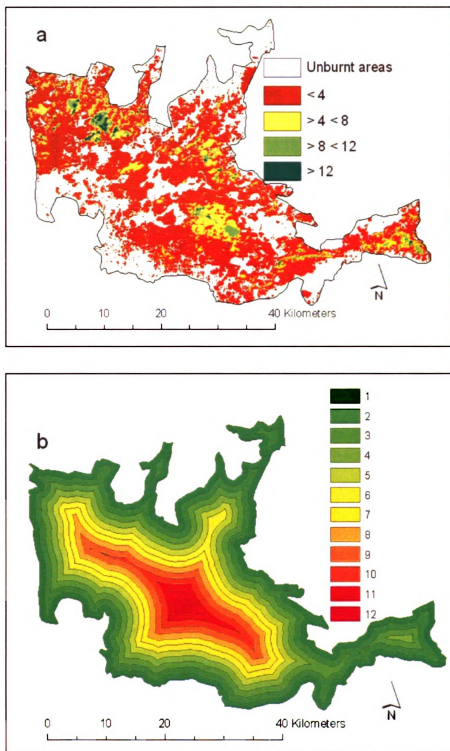


Figure 4-4: Relationship between mean fire-return interval as a function of distance from edge

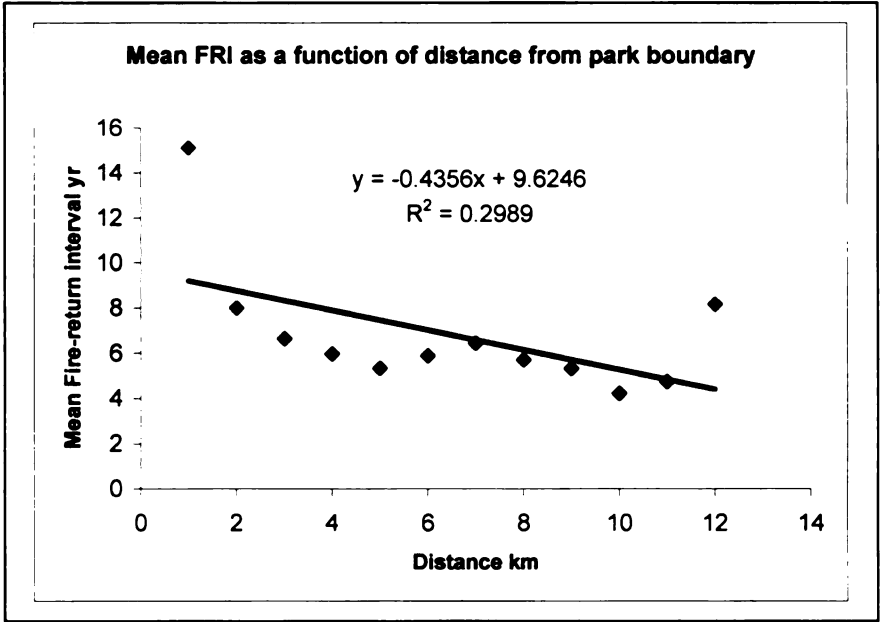


Figure 4-5: Proportion of fire burnt in different fire classes as a function of distance from edge

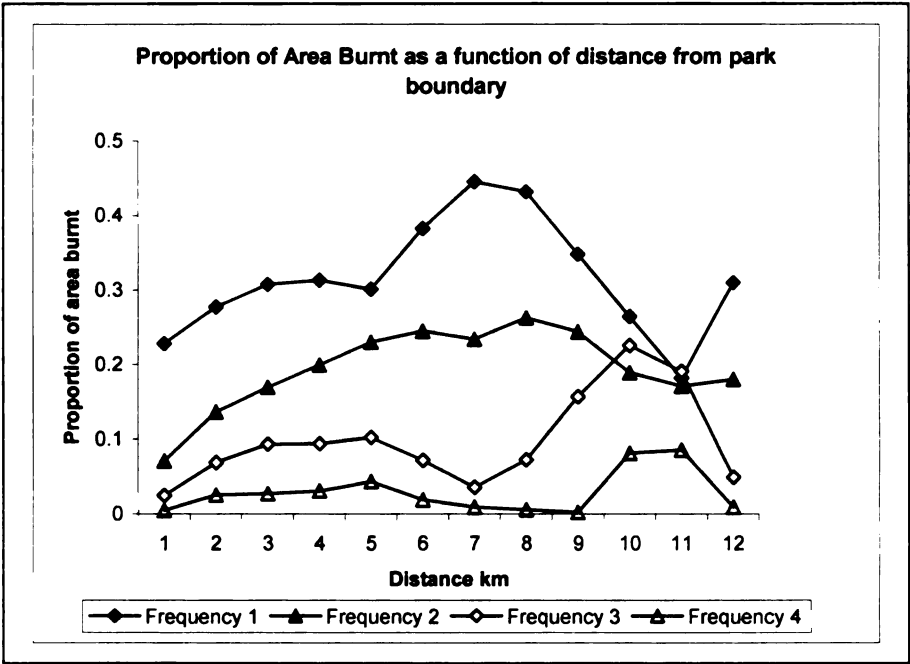


Figure 4-6: Plot of declining area with distance from park boundary in the Nilgiri landscape

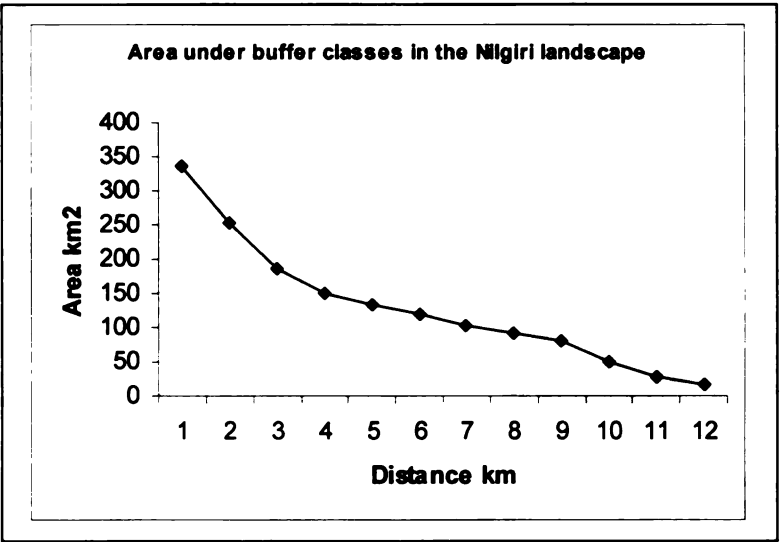


Table 4-1: Results of the models for the relationship between fire-return interval and distance from park boundary in the Nilgiri landscape

	Model	R square	F	p	B ₀	B ₁	B ₂	B ₃
overall	Linear	0.29	4.27	0.06	9.62	-0.43		
overall	Quadratic	0.7	10.8	< 0.01	14.6	-2.5	0.16	
overall	Cubic	0.76	8.6	< 0.01	17.5	-4.8	0.57	-.02
overall	Power	0.57	13.4	< 0.01	11.1	-0.33		

Table 4-2: Final weighted logistic regression model with spatial autocorrelation

Parameter	DF	Estimate	Wald (5% Error)	Confidence Limits		Chi Square	p value
Intercept	1	-5.2	1.86	-8.93	-1.63	8.06	0.0045
Slope (degrees)	1	-0.02	0.026	-0.081	0.023	1.13	0.28
Rainfall mm	1	-0.0005	0.0003	-0.0011	0.0001	3.2	0.07
Elevation m	1	0.0064	0.002	0.0028	0.01	12.57	0.0004
Aspect (degrees)	1	0.002	0.0019	-0.002	0.005	0.83	0.3
Fc (%)	1	-0.029	0.013	-0.05	-0.004	5.12	0.02
C ₀ , nugget		0.1					
C ₁ , sill		1.0					
a, range m		3500					

Table 4-3: Results of the trend surface analysis

Coefficient	Estimate	T/F value	pvalue
Intercept	-1.275	-1.27	< 0.1
x	-3.14 * 10 ⁻⁵	-9.92	< 0.001
x*y ²	2.24 * 10 ⁻¹⁷	10.28	< 0.001
R ²	0.42	53.79	< 0.001
Adj R ²	0.41		

Figure 4-7 a. Fire-return interval map of Mudumalai wildlife sanctuary 1989 to 2002

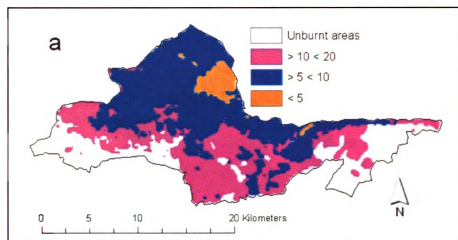


Figure 4-8 a. Elevation map of Mudumalai wildlife sanctuary
b. Aspect map of Mudumalai wildlife sanctuary

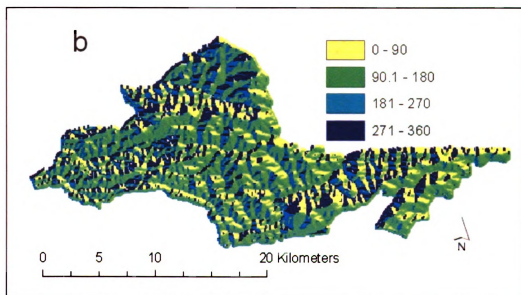
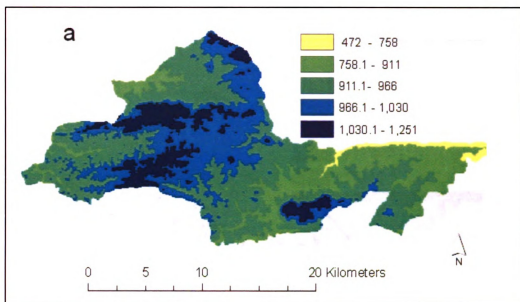


Figure 4-8 c. Slope map of the Mudumalai wildlife sanctuary
 d. Forest fractional cover of Mudumalai wildlife sanctuary

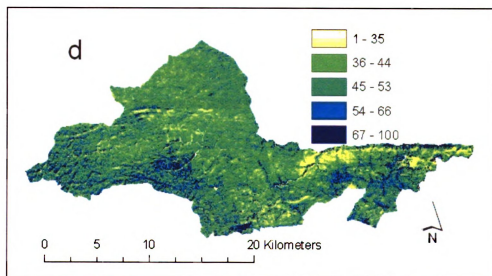
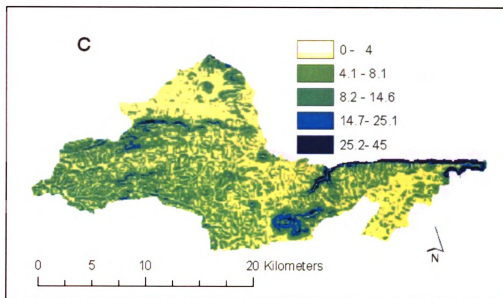


Figure 4-8 e. Mean annual rainfall (mm) of the Mudumalai wildlife sanctuary

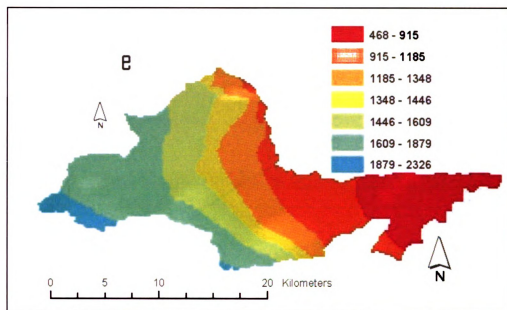


Figure 4-9: Percentage error in predicted values of rainfall

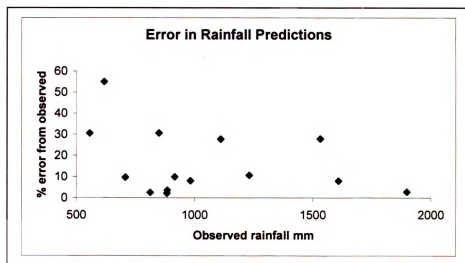


Figure 4-10 Variogram of the standardized residuals of ordinary logistic regression analysis

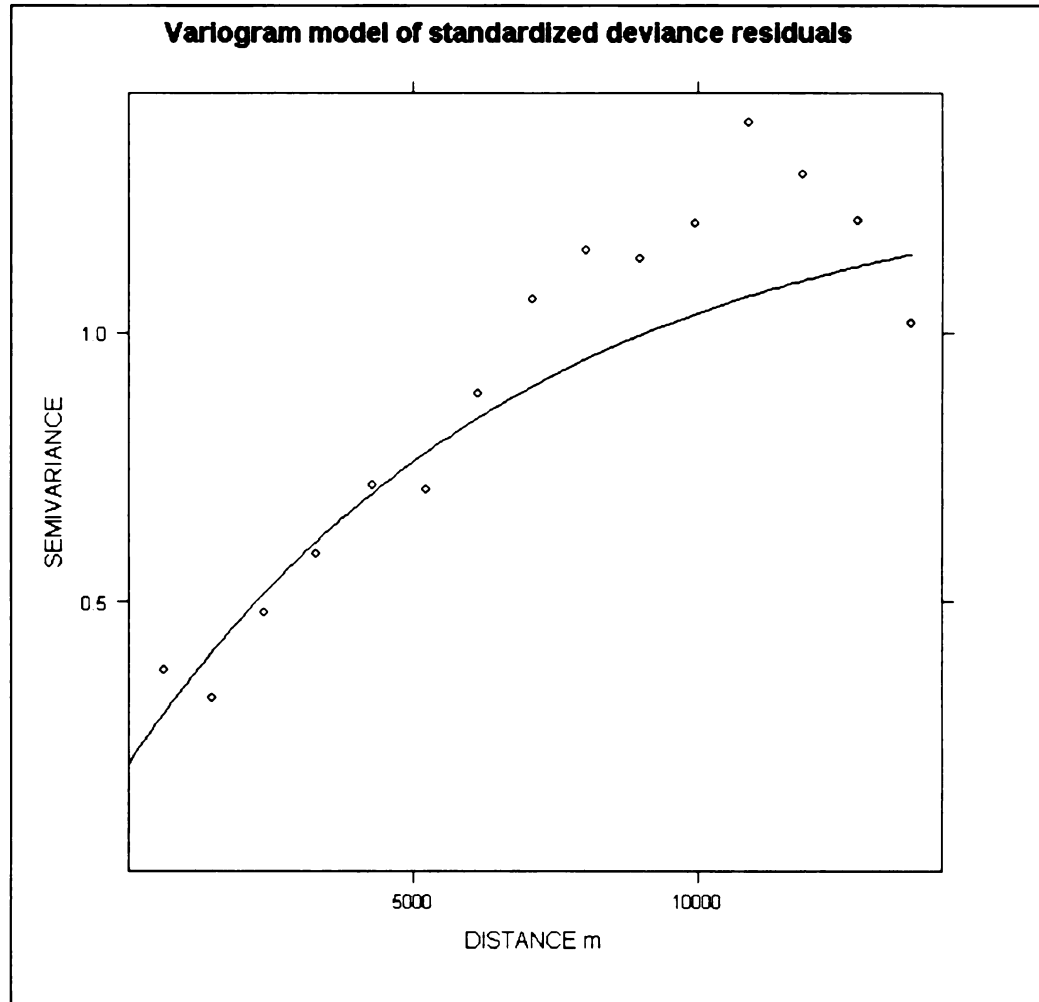


Figure 4-11: Variogram of standardized deviance residuals of weighted logistic regression analysis for spatial autocorrelation

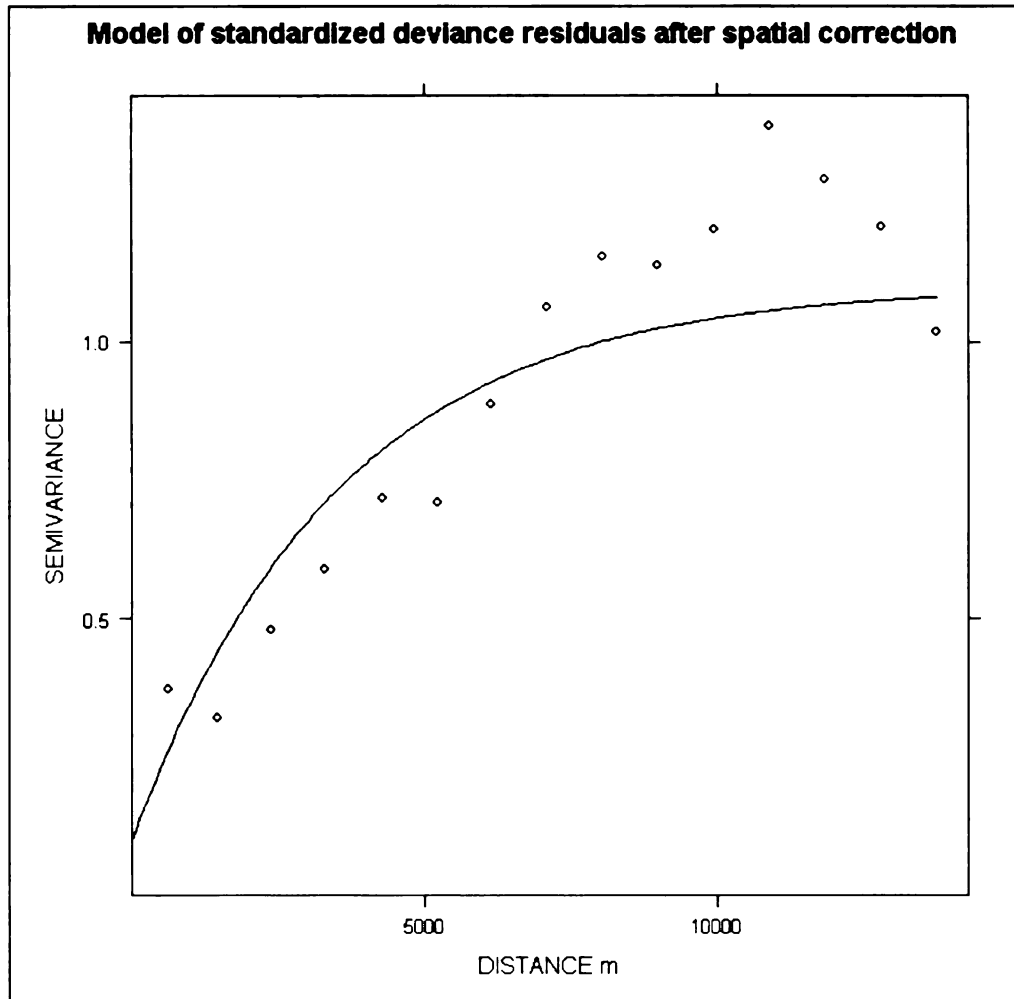


Figure 4-12: Observed spatial pattern of proportion of years in fire

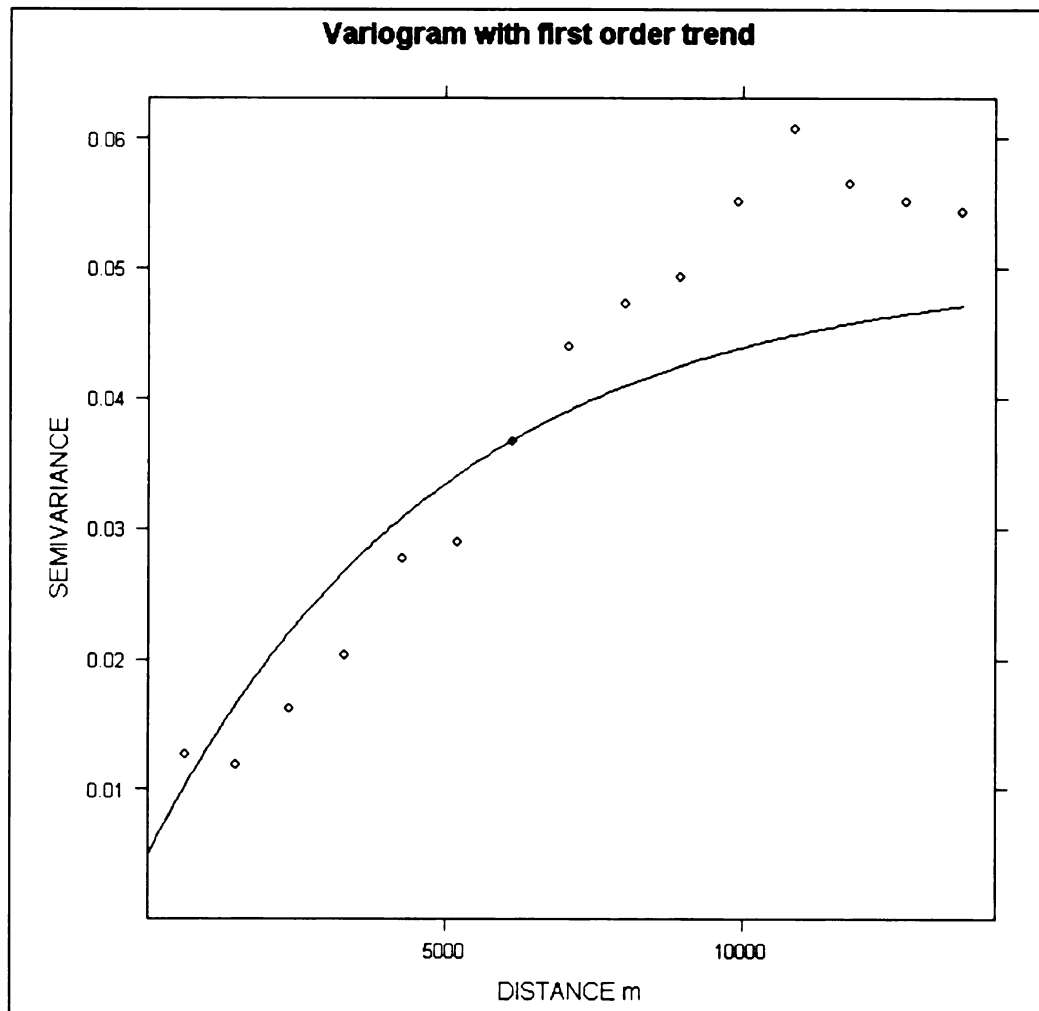


Figure 4-13 Variogram from the residuals of the trend surface

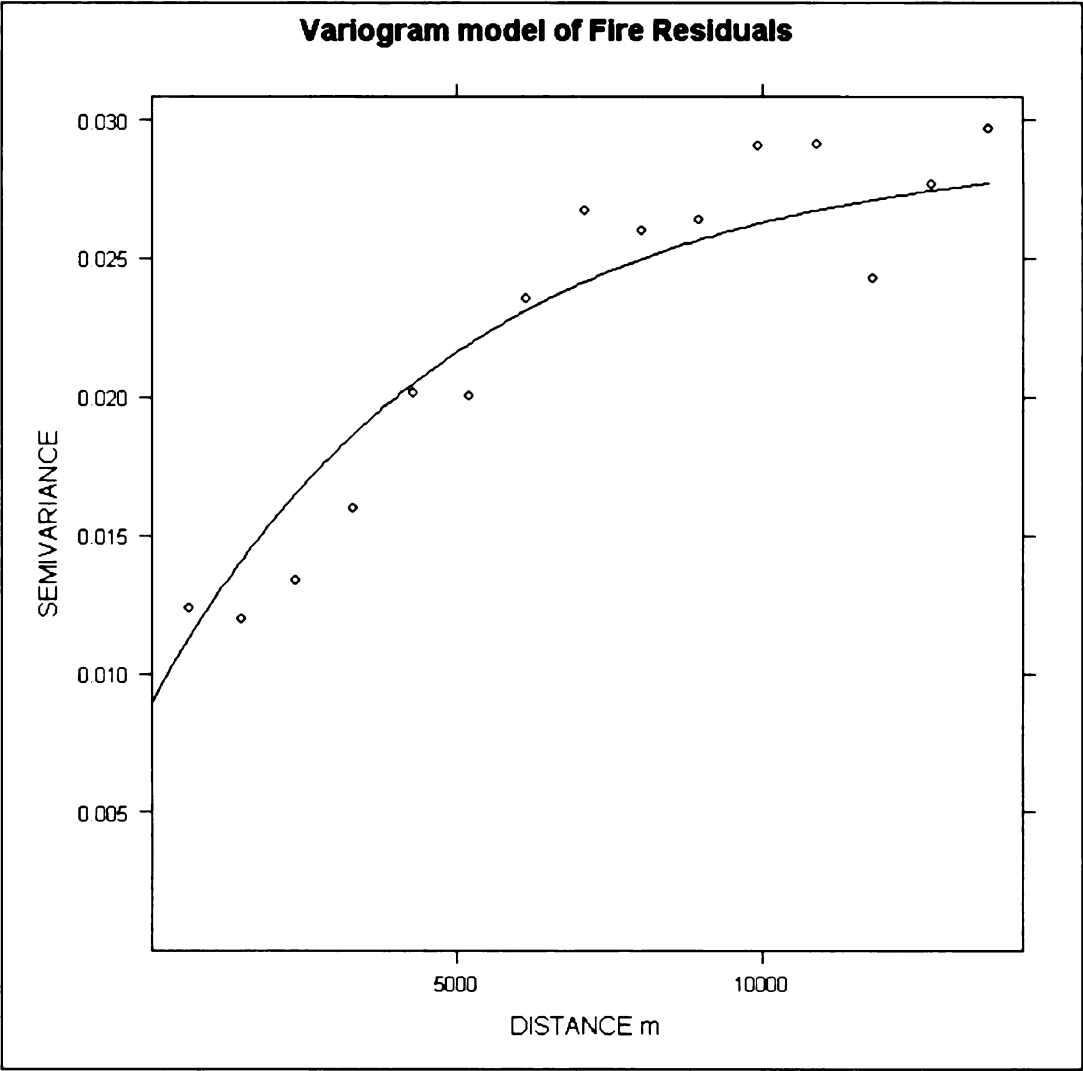


Figure 4-14: Universal kriging predictions for the Mudumalai landscape

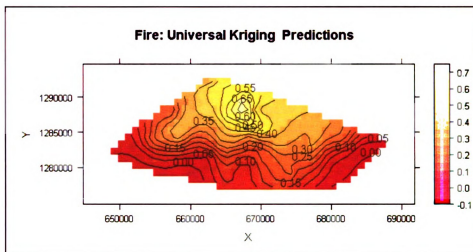


Figure 4-15: Universal kriging standard error map

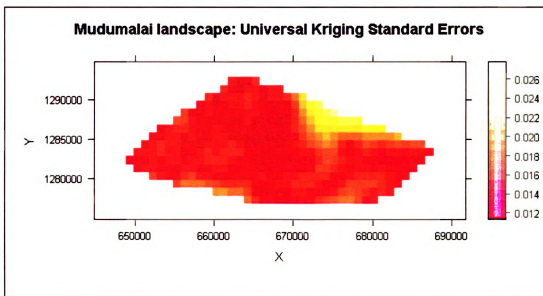


Figure 4-16: Cross validation of Universal kriging predictions

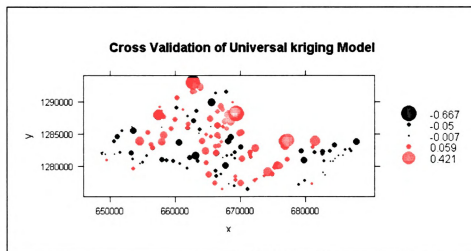
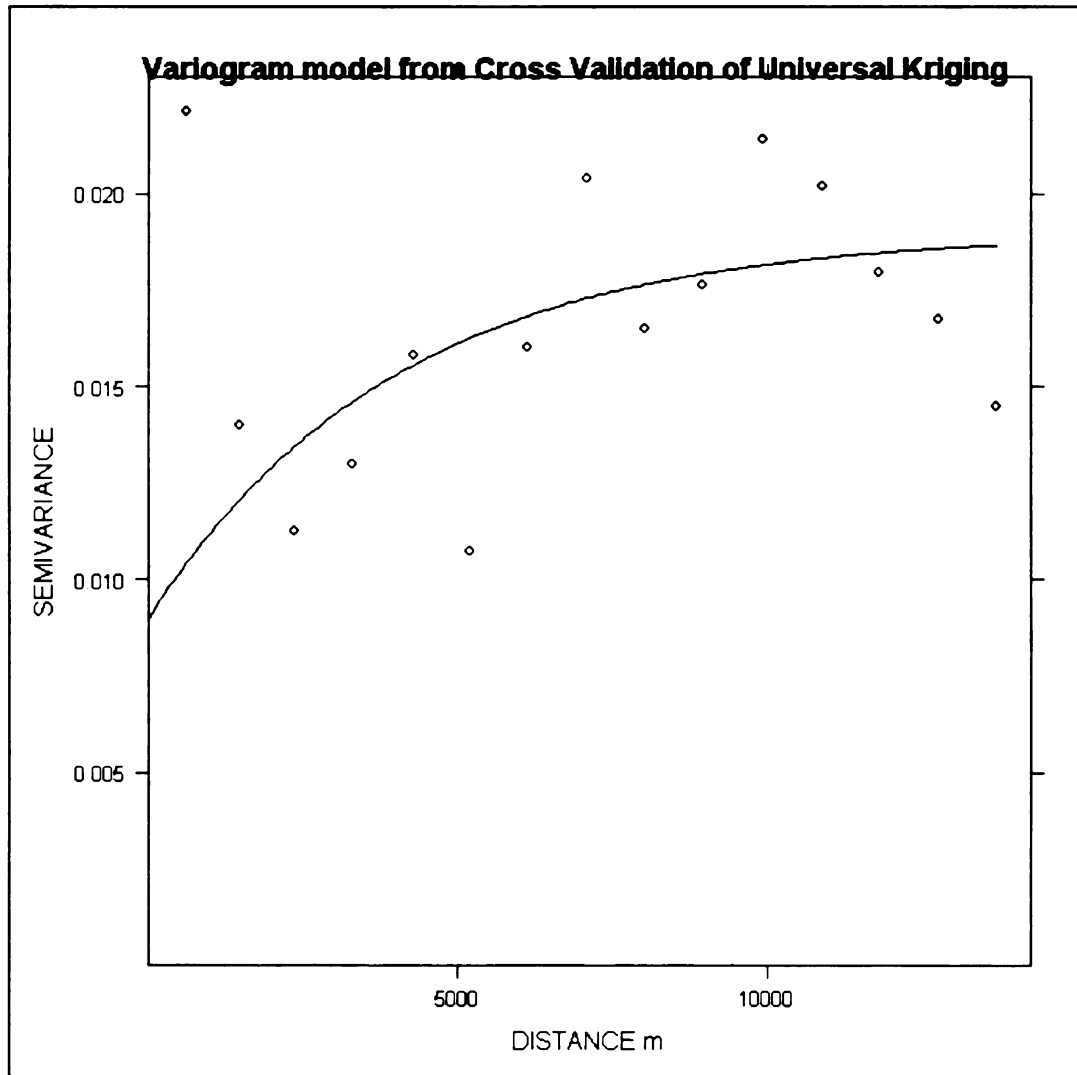


Figure 4-17: Variogram of cross validation of residuals of Universal kriging



Chapter 5

The fire regime and effects of fire in the landscape

Introduction

Humans have consciously used fire as a tool to alter landscape functions and perpetuate preferred ecosystems to suite their diverse needs (Pyne 1996 et al.; Roberts 2000; Gadgil 1993). In the Western Ghats in India, forest fires have been used in slash and burn agriculture for the past 3000 years (Hegde et al. 1998). Forest fires in lowlands of tropical Asia have been incorporated into the ecosystem over many thousands of years and vegetation in these ecosystems has responded to fire occurrence in the environment (Stott et al. 1990). Humans have had a large impact on the dynamics of many ecosystems through a number of activities. In the Nilgiris in India, humans have been known to practice agriculture, collection of non-timber forest products, and graze cattle within forests (Prabhakar 1994; Silori and Mishra 2001; Narendran et al. 2001).

The use of fire in ecosystems has significantly altered the composition of forests in the tropics (Bond and van Wilgen 1996). Forest ecosystems around the world are experiencing a number of disturbance events. While disturbances are integral to ecosystems and perform a number of functions (Huston 1979), key parameters of disturbance regimes in the Nilgiris, such as the intensity, frequency, extent, and its severity (Souza 1984) have not been clearly defined. Disturbance plays an important role in ecosystem dynamics by preventing competitive exclusion (Gause 1934) and thereby increasing resource availability for seedlings.

The magnitude and frequency of the disturbance have important consequences for the number and character of species in ecosystems. When disturbance is extremely

infrequent, it results in dominance by a few species in the ecosystem. While at the other end of the disturbance spectrum, extremely frequent disturbances results in species unable to successfully reproduce and establish within the window of opportunity (Hoffmann 1998). When the spatial and temporal components of disturbances are at an intermediate level, the highest species richness occurs in an ecosystem (Connell 1978; Souza 1979; Lubchenco 1978). Demographic and land use changes in the tropics have significantly altered fire regimes in the tropics (Cochrane 2003). However, in the tropics, monitoring and assessment of current fire regimes is poor. Thus, there is an urgent need to monitor current fire regimes and assess the effects of altered fire regimes on various ecological aspects of the forests. It is in this backdrop that I ask the following questions in this chapter.

Objectives:

1. What are the fire regimes in the three main forest types of the Nilgiri landscape? Specifically, I examine the fuel loads, frequency, extent of fires, and fire severity in these three vegetation types.
2. What are the effects of these fire regimes on the regeneration, structure, composition, and diversity of these forests?
3. What are the differences in mortality across different dbh size classes among the dominant species in the landscape?

Study area

The study was located in the Nilgiri landscape, which is a part of the Nilgiri biosphere Reserve. Transects were located in the Mudumalai wildlife sanctuary and surrounding areas and also in the Bandipur tiger reserve. The Mudumalai wildlife

sanctuary is located in the northern part of the Nilgiri mountain range and is contiguous with the Bandipur and Wynaad wildlife sanctuaries. The predominant vegetation types in these three wildlife sanctuaries are the tropical moist deciduous forests, tropical dry deciduous forests, and the tropical dry thorn forests. Details of the study areas can be found in chapter 1.

Ecological history of the Mudumalai landscape

The Mudumalai landscape where most of this study has been located has witnessed a number of disturbances in the past, anthropological studies have shown that the Nilgiris landscape was settled as early as 100 A.D, this is evident from remains of megalithic cultures present in various sites in the Nilgiris (Hockings 1989, Prabhakar 1994; Sukumar et al. 2004). However, in the absence of any numbers it would be hard to speculate on the exact effects of these human communities. A number of hunter-gatherer societies also inhabited the forests of the Nilgiris and Mudumalai, they include the *Kurubas*, *Irulas*, *Paniyas*, and *Kotas*, these indigenous communities continue to inhabit the fringes of the forest and predominantly obtain a living from the collection of forest products and other activities associated with the forest (Narendran et al. 2001). The presence of malaria in the hill tracts discouraged the people from the plains from settling during the 18th and early 19th century. However, the arrival of the British changed the demographic profile of these regions. The British set up a number of plantations of tea in the upper elevations of the Nilgiris and also began systematic logging operations in these forests (Tucker 1988; Hicks 1928). A number of migrant workers moved from the plains into the hills of the Nilgiris. However population densities were lower than current levels in almost all places.(Sukumar et al. 2004).

The Mudumalai forests were declared reserved land as early as 1889, in 1894 a working plan was formulated to log these forest to make sleepers used in railway tracks (John 2000). The forests were initially owned by the royal family of Travancore, the ownership came under state control in 1927 and in 1940, when 60 km² was set aside as the Mudumalai wildlife sanctuary. It was expanded in 1958 and again to the current area of 321 km² in 1977 (Sukumar et al. 2004). Logging and forest operations began in Mudumalai in the early 19th century. Working plans were drawn out by the mid 19th century to systematically harvest species such as *Tectona grandis*, *Pterocarpus marsupium*, *Terminalia crenulata*, *Lagerstroemia microcarpa*, and *Dalbergia latifolia* (John 2000; Sukumar et al. 2004; Ranganathan 1941). During the Second World War, some core areas of the sanctuary were used for jungle warfare training, to train soldiers for deployment in Burma (John 2000).

Methods:

Transects: Data on woody plant species was collected from belt transects of 500 x 10 m in the study areas. 35 transects were enumerated in the study area, of which 17 transects were from the tropical dry deciduous forests, 10 from the tropical dry thorn forests, and 8 transects from the tropical moist deciduous forests (Appendix 1). Most of the transects were located in the Mudumalai wildlife sanctuary area, however 5 transects were located in the adjoining Bandipur Tiger Reserve, 3 transects were located in a tropical dry thorn forest outside the Mudumalai wildlife forest area, 6 transects in Nilambur and 1 transect in Wynaad. I did this because much of the tropical dry thorn forest within the Mudumalai wildlife sanctuary is exposed to grazing by livestock and also illegal logging of trees. However, in the sampled areas, this was controlled. Transects in the disturbed moist deciduous forests were located

in Nilambur as I did not have permissions of sample within the Nilgiri landscape.

Methods were adapted from (Cochrane and Schultz 1999).

Canopy cover: Along each transect, canopy cover was estimated by collecting canopy pictures using a fish eye lens. Pictures were taken at intervals of 25 m along the transect. The images were later downloaded and analyzed for the canopy cover using gap analysis software (Steege 1996).

Regeneration of woody plant species: At intervals of 25 m on transects, regeneration of woody plant species were estimated along transects in quadrats of 10 x 10 m², there were 20 quadrats in each transect. Data was collected in two size classes 0-5 cm dbh and 5-10 cm dbh in four sub quadrats of 25 m² each. Mortality of individuals in these two size classes was also recorded along transects.

Species composition: All individuals $\geq$ 10 cm dbh were enumerated along each transect and identified to species. Species that could not be identified in the field were sampled and pressed for collection and these have been deposited in the herbarium of the field station of the Center for Ecological Sciences, Indian Institute of Science, Masinagudi. These specimens were later identified with the help of botanists at the institute (Appendix 2).

Fuel composition and fuel load estimation:

At intervals of 25 m along the transect, fuel load was estimated by applying the modified planar intercept method (Cochrane et al. 1999; Uhl and Kauffman 1990). Fuels data was collected in four size classes 0-0.6 cm, 0.6-2.5 cm, 2.5-7.6 cm, and

>7.6 cm along a 10.5 m line laid randomly at intervals of 25 m along the transect.

These fuel classes correspond to 1 hr, 10 hr, 100 hr, and 1000 hr time lags (Pyne et al. 1996). In the fuel size class > 7.6 cm, apart from recording the number of these logs that intercept the fuel line, diameter measurements of the fuels were also collected. Finally the condition of these fuels, in terms of whether they were sound or rotten, was recorded.

Estimating Grass and Leaf litter:

In the different vegetation types, I collected grass and leaf litter in 1x1 m quadrats. The grass and leaf litter were weighed in the field using a balance, they were next packed and transported to the field station where they were oven dried at 100 °C for 24 hours and weighed again. In 2005, there were fires in the dry thorn and dry deciduous forests, I collected samples within these burnt forests after the fires, in addition to sampling the unburnt forests.

The Fire regime of the Nilgiri Biosphere Areas

The fire regime characterizes various components of a forest fire in an ecosystem. It includes the frequency, intensity of fire, season of burning, extent of fire and the type of fire, these components together characterize the fire regime (Whelan 1995).

Although each of these parameters can be characterized for each vegetation type, there is significant variability in frequency, intensity, and extent; it is also clear that synergistic effects are an important component of the fire regime (Agee 2000; Cochrane 2001). I shall provide a description of each of these components below and, where possible, quantify these components.

Time/Season of Fire: Forest fires in the Nilgiri landscape and the Western Ghats in general are restricted to the dry season, between January and April of each year. The eastern slopes of the Western Ghats, which lie in the rain shadow receive very little rainfall. The total rainfall at Mudumalai during these four months is 222 mm. This is approximately 17% of the total annual precipitation, data correspond to mean monthly rainfall collected between 1990-2000 (Sukumar et al. 2004). Table 5-1 provides a summary of the climate data for the Mudumalai landscape. The fire season corresponds to a phase wherein trees in the deciduous forests are in the leaf fall stage and is the dormant phase in this ecosystem.

Test of Hypotheses

Null Hypothesis: *Fire regimes are similar in the entire Nilgiri landscape. There are no differences in the frequency, intensity, areal extent of fires, fuel composition and loads. As a consequence, ecological effects of forest fires are similar in the different forests of the Nilgiri Biosphere Reserve.*

Hypothesis 1: *Fuel load and fuel composition are a function of the ecosystem, and characteristics of species within the ecosystem. Fuels are dynamic and depend on the productivity of ecosystems, past disturbance history and decomposition (Agee 1990; Murphy and Lugo 1986; Sundarapandian and Swamy 1999). Differences in the ratio of the surface area to volume of a fuel lead to differences in the amount of time required to gain or lose water fuel, thereby leading to differences in flammability.*

Hypothesis 2: *Fire regimes are also characterized by the size and extent of the fires and they vary by vegetation type. Fuel continuity and barriers to the spread of fire in*

a forest are important factors contributing to the size of a fire (Pyne et al 1996; Miller and Urban 1999). In certain vegetation types, frequent fires lead to relatively larger burnt areas compared with infrequent fires (Swetnam 1993).

Hypothesis 3: *Fire severity varies by ecosystem type and leads to different effects on the regeneration, mortality, diversity, structure, and composition in the three forest types in the Nilgiri landscape. Intensity of fires is characterized by flame characteristics and is a function of energy content, mass consumed and the rate of spread (Agee 1993). Cell death in plants is a function of both the temperature and duration of exposure (Whelan 1995).*

Hypothesis 4: *The morphological characteristics of trees such as bark thickness and flammability of bark leads to differences in the mortality patterns of trees. Protection from cambial kill increases as the square of the bark thickness; thermal diffusivity of the bark depends on thermal conductivity, heat capacity, and density of the bark (Agee 1993).*

Results:

Frequency: From information obtained from local forest records, remote sensing data and also the fire mapping exercise described in chapters three and four, I obtained the fire frequency of each of the 35 transects in the Nilgiri Biosphere Reserve. The frequency is low in the moist deciduous forest and the dry thorn forests in the East. The mean FRI computed for the Nilgiri landscape shows that the tropical dry deciduous forest have short mean FRI of 6 years, the tropical dry thorn forest has a mean FRI of 10 years, and the tropical moist deciduous forest has a mean FRI of 20 years.

Fuel loads:

I compared fuel size composition in three vegetation types in the NBR. Figures 5-1, 5-2, 5-3 show the quantity of fuel in the three vegetation types. I conducted one-way analysis of variance (ANOVA), with Bonferroni multiple comparison test, to examine the means in each size class by vegetation type. Table 5-2 provides the results of the ANOVA.

1 hr, 10 hr, and 100 hr fuels were significantly higher for tropical moist deciduous forests than tropical dry deciduous, and the tropical dry thorn forests; total fuel loads in the dry deciduous and moist deciduous were not significantly different however they were significantly different when compared with the dry thorn forests. Total fuel loads and the size composition of fuels vary within vegetation types as well as across vegetation types (Agee 2000).

Grass and Leaf Litter estimates

The fine fuel estimate for the tropical dry thorn forest was $0.38 \pm 0.17 \text{ Mgha}^{-1}$ ($n=21$), leaf litter was absent along the transect; in another transect grass fuel estimate was $0.34 \pm 0.18 \text{ Mgha}^{-1}$ ($n=11$), while the leaf litter estimate was $0.19 \pm 0.21 \text{ Mgha}^{-1}$ ($n=11$). In the tropical dry deciduous forest, the grass fuel estimate is $2.56 \pm 1.13 \text{ Mgha}^{-1}$ ($n=21$) and the litter fuel estimate is $1.53 \pm 0.9 \text{ Mgha}^{-1}$ ($n=21$). In the moist deciduous forest, grass fuels estimate is $0.06 \pm 0.1 \text{ Mgha}^{-1}$ ($n=21$) and leaf litter fuel estimate is $4.95 \pm 1.26 \text{ Mgha}^{-1}$ ($n=21$). I also obtained opportunistic estimates of fine fuels after fire in the forest. Fire had swept through certain areas of forests in 2005. In the dry deciduous forest, the estimate for grass fuels was $0.13 \pm 0.1 \text{ Mgha}^{-1}$ ($n=11$), leaf litter was absent. In the dry thorn forest, the estimate of grass fuel after the fire was $0.1 \pm 0.074 \text{ Mgha}^{-1}$ ($n=20$), leaf litter was absent.

Size of forest fires in the three forest types

The size and extent of fires are important components of the fire regime. Fire sizes are variable and are influenced by a number of factors including the weather and, fuel conditions in the forest (Agee 2000). The size of a disturbance determines the ability of species to recolonize after the disturbance, which in turn is a function of the presence of seed sources and their modes of dispersal and germination within the species range (Agee 1993; Souza 1994). I compared total area burnt in each vegetation type in the Nilgiri landscape over the years (1996 – 2005).

The area burnt in each forest type, normalized by the total forest area was significantly higher in the tropical dry deciduous forest than the tropical moist deciduous and the tropical dry thorn forests. However area burnt was not significantly different between tropical moist deciduous and tropical dry thorn forests (Table 5-3). Similarly the mean fire size was significantly larger in the dry deciduous forest than in the tropical moist deciduous and tropical dry thorn forests. However it was not significantly different between the tropical moist deciduous forest and the tropical dry thorn forest (table 5-4). The maximum fire size was significantly larger in the dry deciduous forest than the tropical moist deciduous and tropical dry thorn forests. However it was not significantly different between the tropical moist deciduous forest and the tropical dry thorn forest (table 5-5). Total burnt areas in the dry deciduous forests are the largest from year to year, the mean fire size is also the largest, and the largest burnt areas also occur within this forest type, about 100 km². Fires are of intermediate sizes, (largest fire < 40 km²) in the dry thorn forest and small within the moist deciduous forests (largest fire < 20 km²) figures 5-4, 5-5, 5-6. Fires sizes are also extremely variable within the dry deciduous forests. There are a few large fires in extent, but many small fires simultaneously. However, fires in the dry

thorn forests and moist deciduous forest are generally smaller in size, and less variable (fig 5-7).

Fire Intensity:

I obtained fireline intensity through indirect measurements. I collected data on scorch heights for 56 burnt trees in the forest and used the following equation to obtain fireline intensity. Since the data were collected from the height of burnt marks on the tree bole, it should be treated as a conservative estimate as certain assumptions were made regarding wind speeds.

$$I^{2/3} = h_s/0.148$$

Where I is the fireline intensity and h_s is the scorch height (Rothermel and Deeming, 1980).

The fireline intensity ranges from 1.57 to 86 kWm^{-1} for the various trees measured in the dry deciduous forest of Mudumalai wildlife sanctuary. Fireline intensity relates well to flame length and is a good predictor of the effect of fire on trees or other canopy components of a forest in the convective column above (Rothermel and Deeming 1980). Fire intensities in these forests can be classified as low intensity (0-25 kWm^{-1}), moderate intensity (25-50 kWm^{-1}) and high intensity (> 50 kWm^{-1}).

Fire severity

The Nilgiri landscape is characterized by fires that are mainly low-intensity surface fires and occasionally by medium to high-intensity surface fires, which lead to “torching” of individual tree crowns. Mortality of trees ≥ 10 cm dbh varies across the

different vegetation types in the Nilgiri landscape. This in part could be due to the properties of tree species in the three vegetation types (Uhl and Kauffman 1990; Pinnard and Huffman 1997). In the tropical dry thorn forest tree species are fire hardy, in the tropical dry deciduous forest trees are a combination of fire sensitive and fire hardy species; trees in the moist deciduous forest are mainly fire sensitive species. This has resulted in differences in mortality in the three different vegetation types. I analyzed the abundance of seedlings (0-5 cm dbh) and saplings (5-10 cm dbh) in the different vegetation types across fire severity classes. I defined fire severity by the percentage basal area mortality of woody plants in the transect.

I first examined the differences in mortality among the fire severity classes in the three vegetation types. In the tropical dry deciduous forest there were 4 classes (table 5-6) 0-5%, 5-10%, 10-15% and > 15%, I conducted an ANOVA, the mortality is statistically different in the four classes, $F_{3,17}=20.9$, $p < 0.001$. In the tropical dry thorn forest there were three class (table 5-7), 0-5%, 5-10%, and 10-15%, here again the ANOVA revealed significant differences between mortality in the three classes, $F_{2,10}=41.7$ $p < 0.001$. In the tropical moist deciduous forest there were three classes, unburned, 0-2%, 2-5% mortality, the ANOVA reveals statistically significant differences, $F_{2,7}=13.2$, $p = 0.017$.

Fire severity and regeneration:

Fire severity and seedlings: I examined mean density of seedlings/100 m² in each transect. In the dry deciduous forest, density of seedlings is significantly higher in the 0-5% mortality class than the 5-10% and > 15% mortality classes, however density is not significantly different between the 0-5% and 10-15% mortality classes (Table 5-8). Figure 5-8 shows the differences in seedling density across fire severity classes in

the dry deciduous forests, the primary Y-axis refers to the mean density of seedlings, the density of seedlings declines with increasing fire severity; the secondary Y-axis refers to the coefficient of variation, the 0-5% severity class shows the least variability whereas the 10-15% class has the most variability. Similarly, in the tropical moist deciduous forest, mean density of seedlings/100 m² is significantly higher in unburnt forest than the two mortality classes in the burned forests (table 5-9). Figure 5-9 shows the effect of fire on seedling recruitment in the tropical moist deciduous forest along the primary Y-axis; the secondary Y-axis indicates that the seedling density is less variable in the unburned than the burned forests. However, in the dry thorn forests the mean density of seedlings on the primary Y-axis is high, in the low as well as high fire severity classes, and the mean density is low in the moderate fire severity class; the secondary Y-axis indicates the coefficient of variation, the seedling density is less variable in the high severity classes than the other two classes figure 5-10. The results of the ANOVA did not show significant differences between fire severity in the tropical dry thorn forests $p > 0.05$, (table 5-10).

Fire severity and saplings: I examined saplings abundance in each transect. In the dry deciduous forest ANOVA results do not reveal differences between sapling abundance between the different fire severity classes, $p > 0.05$ (table 5-11), similarly sapling abundance is not different between fire severity classes in the tropical dry thorn forest $p > 0.05$ (table 5-12). However the mean sapling abundance/0.2 ha was significantly higher in unburned forest than the two burned classes; the sapling abundances were not significantly different between the two burned classes in the moist deciduous forests (table 5-9).

Fire and forest structure

I classified forest transects under three classes of fire frequency. Transects that burned only once under the category low, transects with frequencies 2, 3, 4, and 5 under moderate fire frequency class, and greater than 5 in high fire frequency class. The fire frequency data corresponds to a 16 yr time period. In the tropical moist deciduous forest this classification was not possible, only burned and unburned was possible. Table 5-13 provides the total number of stems and basal area in the tropical dry deciduous forests. I computed the mean number of individual trees in dbh mid point classes at intervals of 5 cm beginning at 10 cm dbh. I conducted pairwise t tests to examine differences in forest structure among these fire frequency classes. In the dry deciduous forest the mean number of stems in the lower size classes (< 20 cm dbh) is higher in the low fire frequency class, however in the moderate and high frequency classes, the mean number of stems is greater in higher size classes (> 20 cm dbh), fig 5-11. In the dry deciduous forest paired t tests showed significant differences between low and moderate $p < 0.05$; low and high $p < 0.05$; however, moderate and high were not significantly different, $p=0.46$, table 5-14. I also conducted paired tests for related samples, the Wilcoxon signed ranks test showed significant differences only between low and moderate frequency classes, $p < 0.05$; however the differences were not significant between low and high and moderate and high frequency classes. The Kolmogorov Smirnov Z test was significant between low and moderate, $Z=1.42$ $p < 0.05$, however they were not significant between low and high and moderate and high frequency classes, $p > 0.1$.

The structure of woody plant species in the dry thorn forest is different in three fire severity classes, in the low frequency class and the high frequency class the mean

number of trees in the (10-20 cm dbh) are greater when compared with the moderate frequency class. However in the moderate frequency class, the mean number of stems in the larger size classes (> 15 cm dbh) is greater than the low frequency class, fig 5-12. Trees in the large size class (> 50 cm dbh) are completely absent in the high frequency class. Table 5-15 provides the total number of stems and basal area by size class. I conducted paired t tests in the dry thorn forest, paired tests between low and moderate frequencies are not significant different $p > 0.1$; however they are significantly different between low and high ($p < 0.1$) and moderate and high ($p < 0.05$) frequency classes (table 5-16). I also conducted Wilcoxon signed ranks test, the tests were significant between low and moderate and moderate and high frequency classes, $p < 0.1$, but were not significant between low and high frequency classes. However the Kolmogorov Smirnov Z test was not significant between any of the fire frequency classes.

In the unburned moist deciduous forest, the structure of trees follows a typical reverse J distribution, whereas in the burned forests the larger sized trees (> 15 cm dbh) are more abundant, fig 5-13. Table 5-17 provides the total stem numbers and the basal area by size class in the two categories. The statistical analysis in the tropical moist deciduous forests does not reveal significant differences between burned and unburned forests due to the large difference in stem numbers between the lower size classes and the higher size classes. In order to overcome this problem, I truncated the analysis at 55 cm dbh and conducted the statistical analysis. The results show significant differences at $p < 0.1$, I also conducted nonparametric tests of related paired samples, the results show significant differences between the burned and

unburned moist deciduous forests $p < 0.1$, (table 5-18). The Kolmogorov Smirnov Z test did not show significant differences $p > 0.1$.

Fire and species diversity

I computed the reciprocal of the Simpson's index of diversity for woody plants along each transect, I also computed the equitability index for each transect. Diversity is relatively higher in the tropical dry thorn forest, followed by the tropical moist deciduous forest and tropical dry deciduous forests. There is a decline in the diversity index with an increase in the fire frequency in the dry deciduous forests; R-square is 0.245, $F=4.87$, $p < 0.05$, fig 5-14. However there does not exist statistically significant effects of fire frequency on species diversity in the dry thorn (fig 5-15) and the moist deciduous forests (fig 5-16). The equitability index, which is a measure of the evenness in the abundance of species in the transect, reveals relatively low values for the tropical dry deciduous, interesting there is an increasing trend in equitability with an increase in fire frequency (fig 5-17). Similarly, in the tropical moist deciduous forest the equitability index is low (fig 5-18). However in the dry thorn forest the equitability index is relatively higher and the highest values from among the three vegetation types occurs in this forest type (fig 5-19). I computed ANOVA to examine differences between the diversity indices between the three vegetation types, $F=3.7$, $p=0.03$.

Fire hardy species and dominance in the landscape:

A number of mechanisms have been evolved by plants to reduce mortality of trees in a population. The presence of a thick bark has been shown to confer protection against cambial kill in different forests (Pinard and Huffman 1997; Uhl and Kauffman

1990). Insulation from heat by the presence of subterranean stems and meristematic cells has also been shown to confer protection from thermal injury (Whelan 1995). In certain species growth spurts just before the onset of surface fires, thereby increasing the height of stems and protecting the sensitive tissues from thermal kill has also been shown to reduce injury to plant tissues (Whelan 1995). I examined the effects of fire on three species in the tropical deciduous forests in the Mudumalai landscape, they are *Anogeissus latifolia*, *Tectona grandis*, and *Terminalia crenulata*. Mortality in *Tectona grandis* ranges from 60% of the stems in the 10-15 cm dbh class to 5% in the > 60 cm dbh class, with a mean of 21% (fig 5-20). However in *Anogeissus latifolia* mortality ranges from 4.8% in the 25-30 cm dbh class to 0 in the > 60 cm dbh class with a mean of 2.7% (fig 5-21), in the case of *Terminalia crenulata* mortality ranges from 5.2% in the 35-40 cm dbh class to 0 in the > 60 cm dbh class, with a mean of 1.8% (fig 5-22). Thus there is a striking difference in the percentage of mortality among species in the landscape. Paired t-test did not show significant differences in mortality across the dbh classes between *Anogeissus latifolia* and *Terminalia crenulata*, however there were significant differences in mortality across the dbh classes between *Anogeissus latifolia* and *Tectona grandis*, as well as between *Terminalia crenulata* and *Tectona grandis*, $p < 0.01$, table 5-19.

Discussion:

The differences in fire pattern and its effects in the three different vegetation types represent variations in the climate, fuel distributions, topographic variations, and species compositions in the ecosystem. There not only exists differences in the fire pattern across vegetation types, there are also differences within a vegetation type, indicating the spatial and temporal variability in the occurrence of fire in the Nilgiri

landscape. The characteristic dry period in deciduous ecosystems increases fuel flammability over large parts of the Nilgiri landscape.

Frequency and incidence

The results suggest that the fire frequency is different among the three vegetation types, these variations reflect the differences in the vegetation types in terms of the fuels, topographic features, climatic patterns, and the proximity to sources of ignitions.

The size and composition of fuels influences the flammability as well as the severity of fires. The surface area to volume ratio of fuels is important in determining the sensitivity of the fuel to equilibrium moisture conditions and thus the flammability of fuels (Agee 1993). There is a larger proportion of the 1, 10, and 100 hr fuels in the dry thorn and moist deciduous forests, however in terms of the total fuel loads, it is highest in the dry deciduous forests. The fine fuels, in the form of the grasses and the leaf litter are also extremely sensitive to changes in the ambient relative humidity of the atmosphere. This could be one of the reasons driving the short fire-return intervals in the dry deciduous forests. However in the moist deciduous forests leaf litter forms a large component of the fine fuels, but the high ambient humidity within the forest requires a prolonged desiccation period to render the fuels flammable, this might explain the long fire-rotation intervals in these forests (Barlow and Peres 2003; Cochrane and Schulze 1999; Siegert et al. 2001). In 2004 a large fire broke out in the moist deciduous forests in the Nilgiri landscape (c. 14 km²), increased drying could be one of the reasons behind this anomalous fire event in this forest. In the dry thorn forests the low fuel load requires a longer time period for the lower fuel sizes to

accumulate before they could effectively support a low intensity surface fire, hence fire-rotation intervals are long.

The extent of fire in the ecosystem has important implications for the species composition and abundance of species, subsequent to the disturbance event (Agee 1993; Souza 1984). The large areal extent of fires in the dry deciduous forest could be one of the reasons for the declining diversity as well as low equitability index within these forests. Only species that can be dispersed over large distances either through wind or bird dispersal might be able to successfully reproduce (Barlow and Peres 2003). However, species with resprouting mechanisms could also thrive under the current fire regime, *Cassia fistula*, *Lagerstroemia microcarpa*, and *Tectona grandis* have a large number of recruits and reproduce vegetatively (John 2000). Both the mean fire size as well as the largest fires occurs in the dry deciduous forests. However the small size of disturbance in the dry thorn forest and the moist deciduous forests could actually be promoting the coexistence of species and hence promoting diversity in these two forest types (Connel 1978; Huston 1979).

Intensity:

Forest fires in the Nilgiri landscape can be extremely variable, fuel size and arrangement is variable in the landscape, the fuel loads are variable across the three vegetation types. In the dry thorn forest the surface fires are of low intensities however occasionally they could lead to moderate intensity fires in places where fuels have accumulated. The fuel bed in some areas is discontinuous almost barren with exposure to barren soil, thus fire intensity and spread is variable in this vegetation type. In the dry deciduous forests the fuel bed is continuous and intensity could range from moderate to high intensities. On occasions the torching of the crown in these

forests can be observed (Kodandapani pers obs). The high proportion of the 1000 hr fuel class within this forest could be another reason for the high intensity within the forest. Fires in the moist deciduous forest are mostly low intensity and creeping fires. Although the fuel bed is continuous, high ambient humidity conditions attenuate the rapid spread of fire within these forests.

Fire severity:

Fire severity has effects on the regeneration of seedlings in the study area. In the dry deciduous forest there is a decline in the density of seedlings with increasing fire severity. Earlier studies in a 50-ha plot have reported increased mortality among seedlings (1 – 5 cm dbh) due to fire. Mean annual mortality during 1988-1996 was almost 30%, while it was lower than 1% for trees > 30 cm dbh (John 2000). In the dry thorn forests there is a marginal decline in regeneration, however, there is also an increase in regeneration in the high fire severity class. The differences are not statistically significantly different, perhaps an indication of the patchy nature of fire in this ecosystem, as well as the fire hardy characteristics of species within these ecosystems, long fire-return intervals, and low intensity of fires in the ecosystem. In the moist deciduous forests, the fire severity has a significant effect on the regeneration of seedlings, these forests have long fire-return intervals, species are also fire sensitive and a number of species are evergreen and might have evolved in the absence of fire (Cochrane and Schultz 1999; Barlow et al. 2003; Hegde et al. 1998; Kilgore and Taylor 1979).

Fire and forest structure:

Fire frequency has a significant effect on the stand structure in the dry deciduous forests, as expected the structure demonstrates a right skewed distribution pattern, however in the moderate and high frequency classes there are a larger number of stems in the > 20 cm dbh classes. These results are consistent with studies on stand structure in fire impacted forests (Cochane and Schultz 1999; Barlow 2003; Peres 1999; Chappell and Agee 1996). Large trees require higher temperatures to bring about cambial kill and there exists a positive correlation between bark thickness and dbh size, thick barks provide insulation against the high temperatures (Pinard and Huffman 1997). In the dry thorn forest the structure of trees in the low and moderate fire frequency classes is not very different, however stems > 50 cm dbh are absent in transects of high fire frequency. The lack of structural differences in the low and moderate frequency classes can be explained by the fire hardy nature of species within this forest type. The growth of trees in the high frequency class might be curtailed due to the repeated fires, further the presence of larger number of stems in the lower size classes can be explained if delayed mortality is invoked in this forest, such delayed mortality has been shown in the forests of Amazonia (Cochrane and Schulz 1999; Barlow et al. 2003; Peres 1999). In the moist deciduous forests the large sized trees dominate the burned forests, however the large number of stems in the lower size classes in the unburned moist deciduous are canceling out the differences, the truncation of the analysis to < 55 cm dbh provided significant differences in structure between the burned and unburned forests.

Fires and species diversity:

In the dry deciduous forests, species diversity as obtained by the reciprocal of the Simpson's index declines with increasing fire frequency, at low and intermediate fire frequencies disturbance in the ecosystem prevents competitive displacement and/or competitive exclusion (Connel 1978; Souza 1984; Huston 1979), however at high fire frequencies there could be competitive displacement with only a few species that can thrive in the environment. Interestingly, the equitability index increases with increasing frequency. What this means is that at high frequencies, a few species, which are fire hardy, are more evenly abundant whereas at low frequencies there are more species, the low disturbance levels permits the presence of many more rare species.

In dry thorn forest, both diversity and equitability are high, perhaps an indication of the fire hardy nature of tree species in these forests; in the dry deciduous forest, diversity is at intermediate levels and equitability low perhaps an indication of the ongoing changes in these forests with a few species well adapted to the occurrence of fire; finally both diversity and equitability are low in the moist deciduous forest, an indication of the fire sensitive trait of species in these forests.

In the moist deciduous forests, the unburnt forest has a lower diversity as well as lower equitability index, and the burned forest shows higher diversity and equitability. Canopy gaps created by the death of trees provide an opportunity to shade intolerant species to germinate and survive (Denslow 1980; Daniels et al. 1995). Although in some forests fire does promote regeneration by the clearing of the substrate and providing seeds an opportunity to come in contact with the soil (Chappel and Agee 1996; Slik et al. 2002), the reduced regeneration of seedlings and saplings in the

burned moist deciduous forests can be explained as the grasses have invaded the undergrowth and also the altered microclimatic conditions in the forests (Freifelder et al. 1998). The fire sensitive nature of species indicated by the structure of the burned forests suggests high mortality among the lower size classes, besides a larger proportion of stems in the higher dbh classes could be an indication of the positive effects of thick bark in mitigating mortality due to cambial kill (Pinard and Huffman 1997; Uhl and Kauffman 1990). Studies in the Western Ghats have shown an increased tendency of trees in disturbed environments to possess thick barks in certain species. Species sorting during the long human occupation, which included the use of fire for a number of activities such as slash and burn agriculture over the past 3500 years could have resulted in an abundance of fire hardy species with thick barks; many of these species are now found in climax evergreen and semi evergreen forests of the Western Ghats (Hegde et al. 1998).

The three vegetation types provide a continuum along a disturbance gradient, the moist deciduous forests and the dry deciduous forests form the extremes of the disturbance spectrum and the dry thorn forests lie in between these two extremes of disturbance. This study shows the importance of not merely a one-dimensional approach to the study of disturbance in the landscape, examining various components of a disturbance regime is important in assessing its effects on species diversity. The fire frequency, the fire-rotation interval, the size fires, the intensity of the fire, and the fuel load all require careful examination in order to assess the effects of fire in an ecosystem.

Figure 5-23 shows a conceptual model of the effects of disturbance on diversity in the Nilgiri landscape. In the moist deciduous forests the diversity index is low, the disturbance is characterized by long fire-return intervals, small size of fires, and low fire frequencies; dry deciduous forests reveal low diversity values, the disturbance is characterized by short-fire return intervals, large size of fires, and high fire frequencies; dry thorn forests have the highest diversity indices, the disturbance is characterized by intermediate fire-return intervals, intermediate fire frequencies, and intermediate size of fires. Similar studies have been conducted in various ecosystems and these studies have reported the highest diversity values at intermediate levels of disturbance (Souza 1984; Lubchenco 1978). Apart from these differences in the fire regime of the three vegetation types, the invasion of grasses and exotic species such as *Lantana camara* and *Eupatorium odorata* have effected these ecosystems, both directly, by preventing the regeneration of seedlings, and indirectly, by increasing the magnitude of the various components of the fire regime in the dry deciduous and moist deciduous forests. Fire removes trees, the absence of trees permits high wind speeds at the grass canopy (Freifelder et al. 1998), leading to increased curing of fine fuels. The fuel bed is almost continuous over larger distances, the packing ratios of the fine fuels (Agee 1993) also aids the spread of fire in these forests. The grass-fire cycle maybe an important driver of the fire regime especially in the dry deciduous and disturbed moist deciduous forests, the invaded grasses promote fire for the reasons given above, following this grasses out compete plants. The increase in grass fuels leads to increasing fire in the landscape due to altered microclimatic conditions and increase in the availability of fine fuels (D'Antonio and Vitousek, 1992).

Fire hardy species in the landscape

While the presence of a thick bark in *Terminalia crenulata* could be a reason for the low mortality among trees of all size classes (Pinard and Huffman 1997), the lack of a thick bark in the case of *Tectona grandis* could be invoked to explain the high mortality in the species. However, the lack of a thick back and low mortality in *Anogeissus latifolia* poses a problem in explaining low mortality in this species. An interesting and possible fire hardy trait of the species could explain the low mortality among *Anogeissus latifolia*, it sheds its bark during the fire season and has a smooth pale bark. In certain trees in Australia, species with similar barks have been shown to have a delayed charring time, further they do not ignite (Gill and Ashton 1968). In addition to the bark non-flammability, it is also possible that the smooth pale bark reflects more energy than it actually absorbs leading to lower temperatures at the bark (Whelan 1995). Besides the smooth bark reduces the surface area volume ratio thereby altering the rate at which energy transfer occurs (Agee 1993).

Conclusions:

The fire regime in terms of the fuel loads, size of fires, and the intensity of fires are significantly larger in the tropical dry deciduous forest. This has resulted in significant impacts on the regeneration, structure, and diversity within this forest type. Species that are well adapted to fire, either in terms of bark characteristics or sprouting behavior currently dominate the landscape. The current fire regime poses a threat to the regeneration and composition within the dry deciduous forest type. In the tropical moist deciduous forest the lack of fire in the past has resulted in a few species that are well adapted to fire, the current fire regime in this forest has impacted the regeneration and composition of species in the forest. Under current fire regimes,

the deciduous forests of the Western Ghats which are the predominant vegetation type will be converted into species poor ecosystems. In the dry thorn forests a combination of the fire regime, climate as well as the prolonged use of the forests for different activities has resulted in tree sorting, with a number of species that are drought and disturbance tolerant, this has resulted in relatively higher species diversity in this forest type. Thus each of the alternate hypothesis laid out provide insights to the fire regime and its effects in the different vegetation types and the landscape as a whole in the Nilgiris.

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Figure 5-1: Fuel size composition in the dry deciduous forest

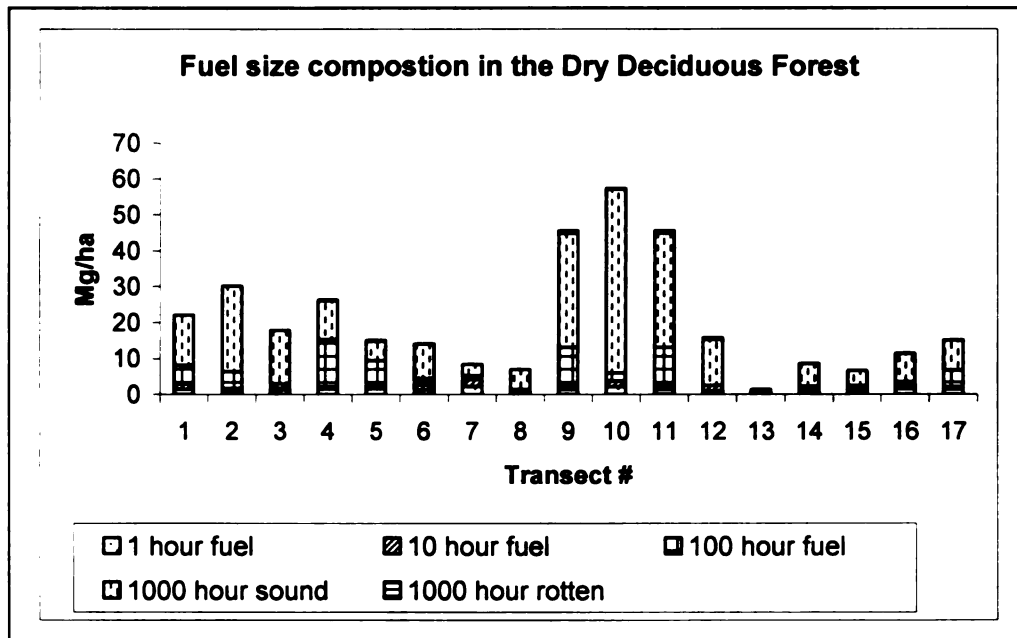


Figure 5-2: Fuel size composition in the moist deciduous forest

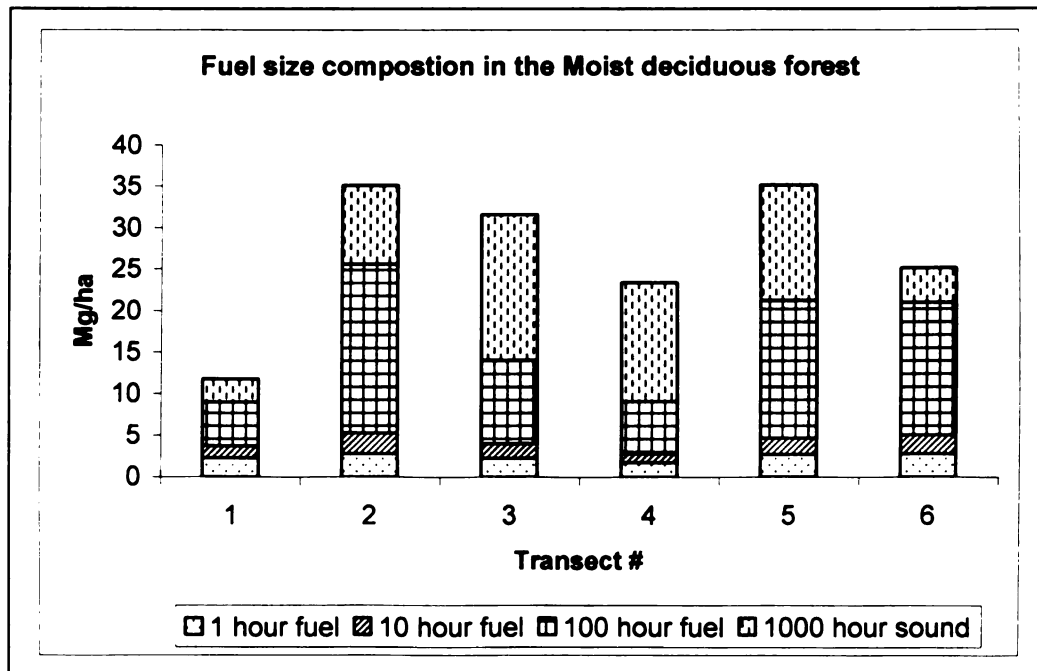


Figure 5-3: Fuel size composition in the dry thorn forest

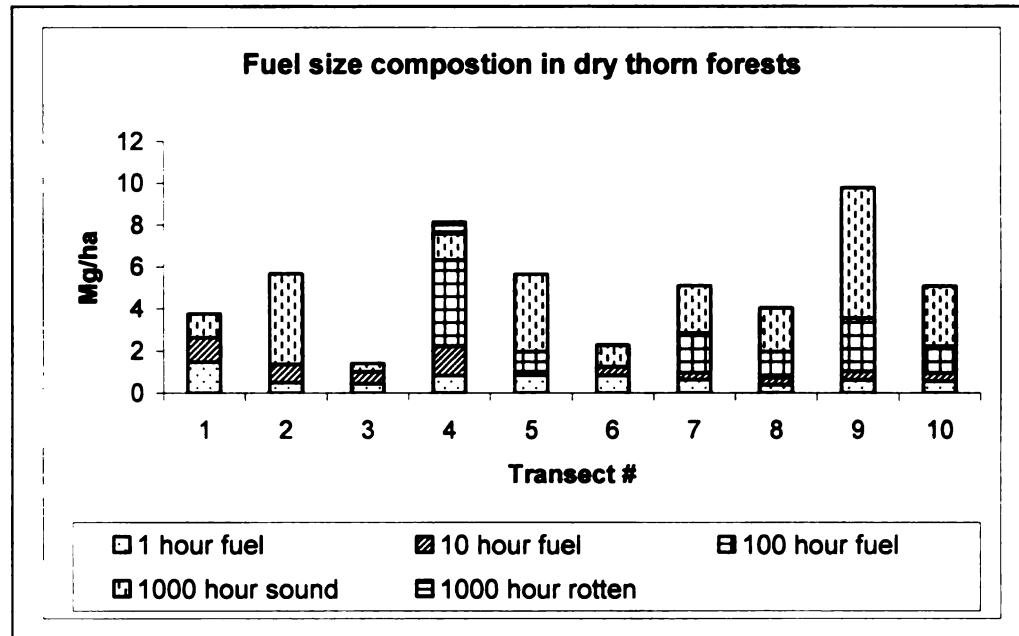


Figure 5-4: Temporal pattern of fire in the Nilgiri landscape

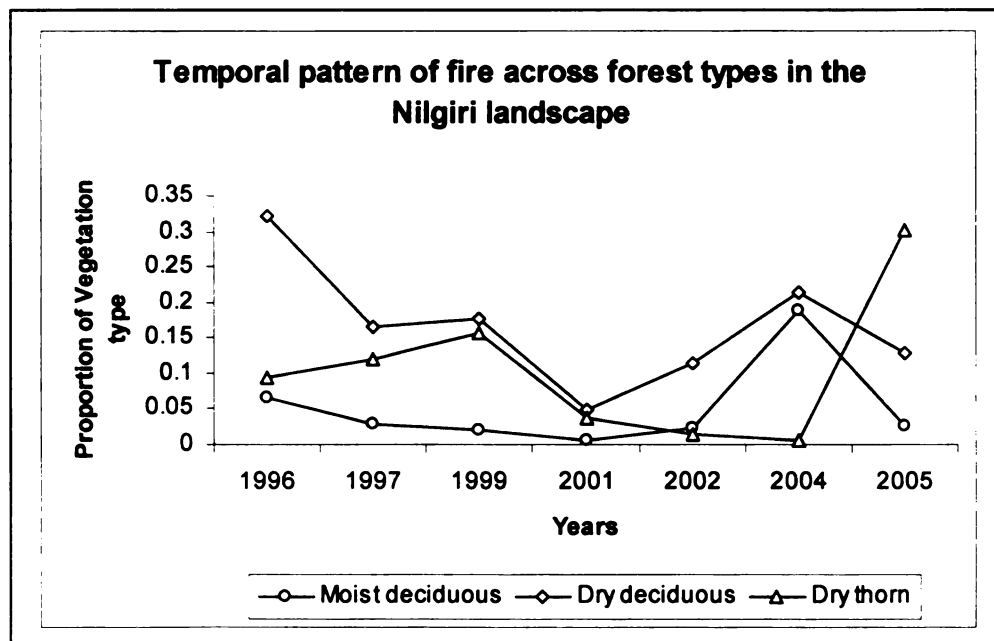


Figure 5-5: Maximum fire size of burnt areas in the Nilgiri landscape

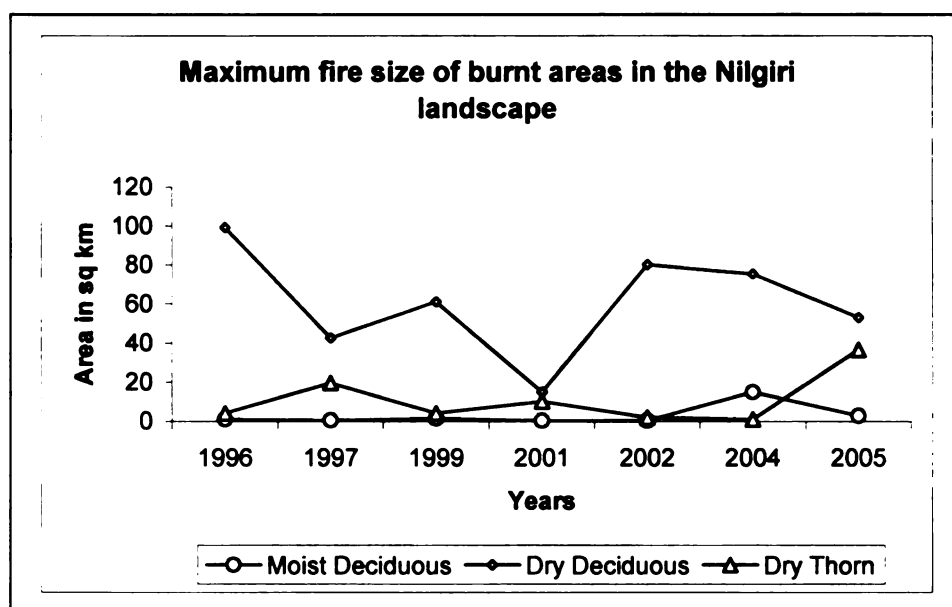


Figure 5-6: Mean fire size of burnt areas in the Nilgiri landscape

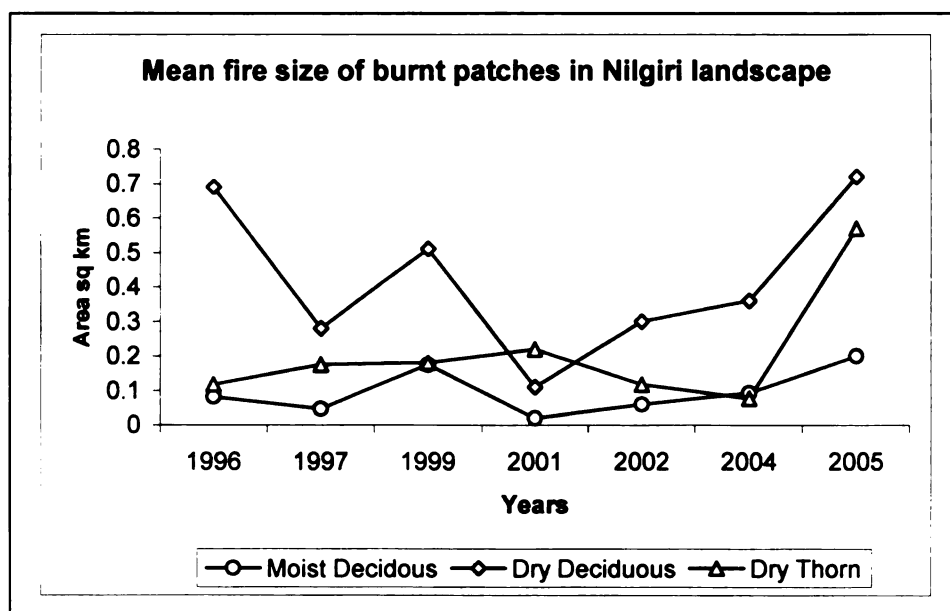


Figure 5-7: Variability in burnt areas in the Nilgiri landscape

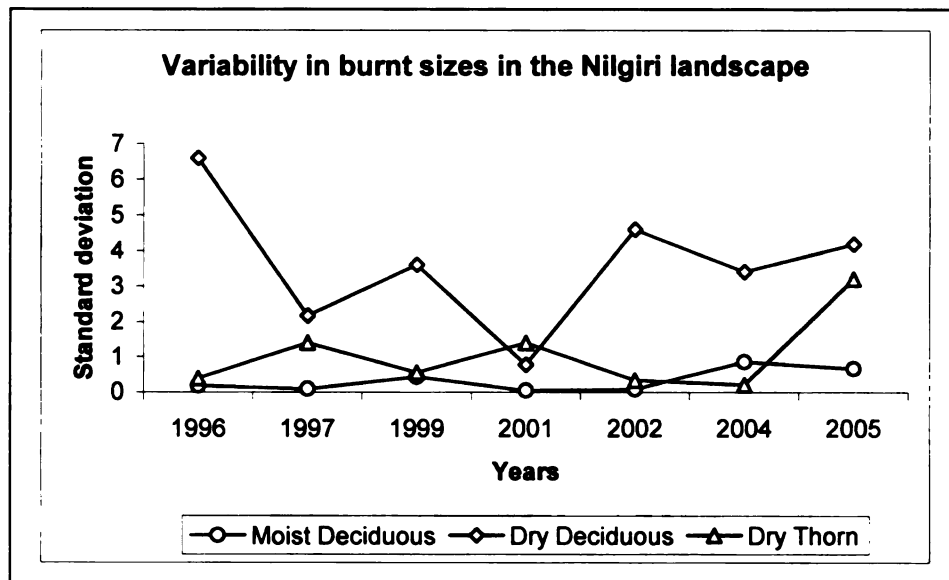


Figure 5-8: Fire severity and seedling density in dry deciduous forest

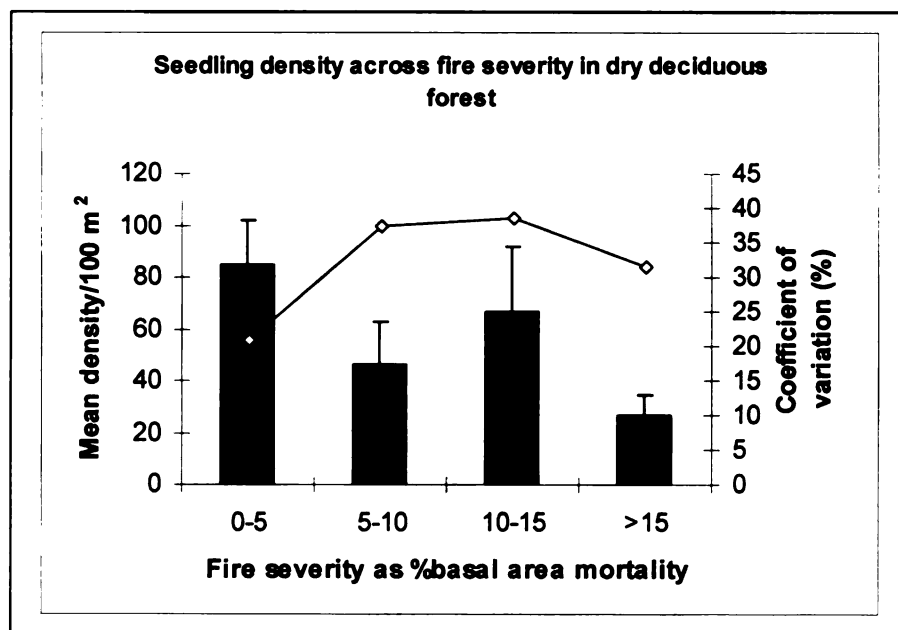


Figure 5-9: Fire severity and seedling density in moist deciduous forest

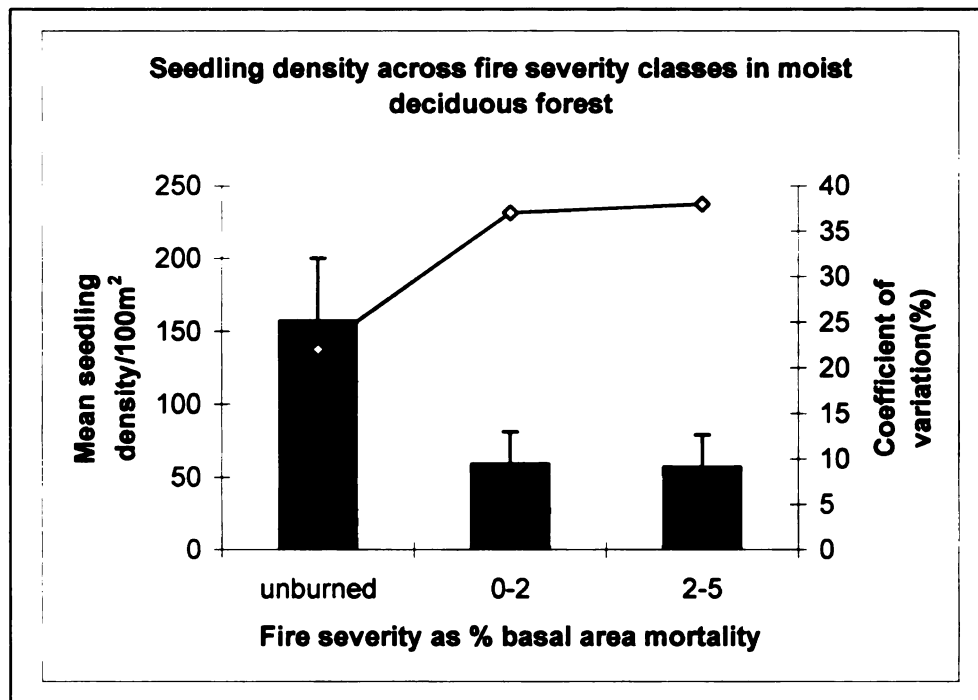


Figure 5-10: Fire severity and seedling density in dry thorn forest

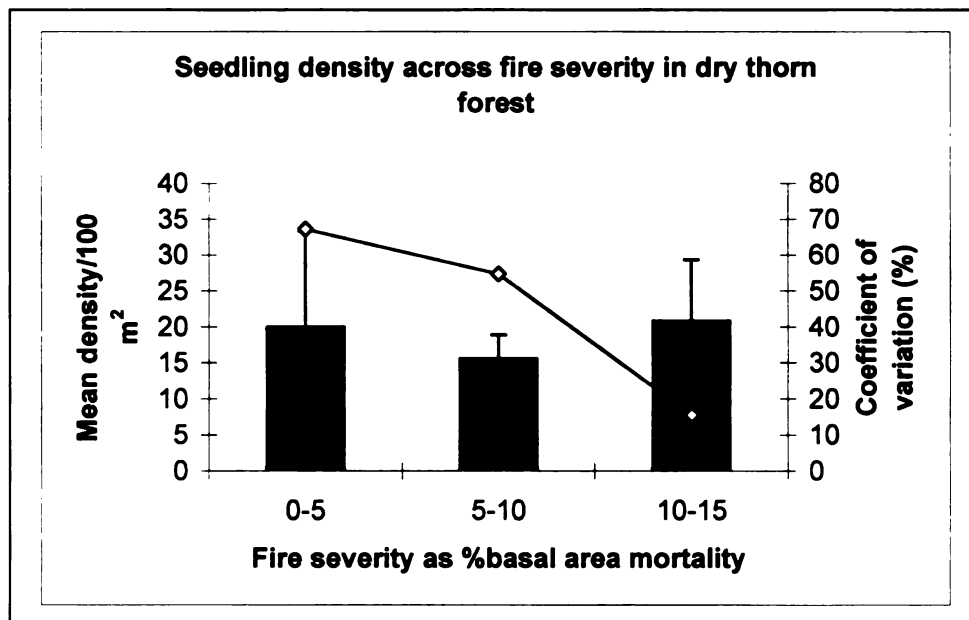


Figure 5-11: Woody plant structure and fire frequency in dry deciduous forests

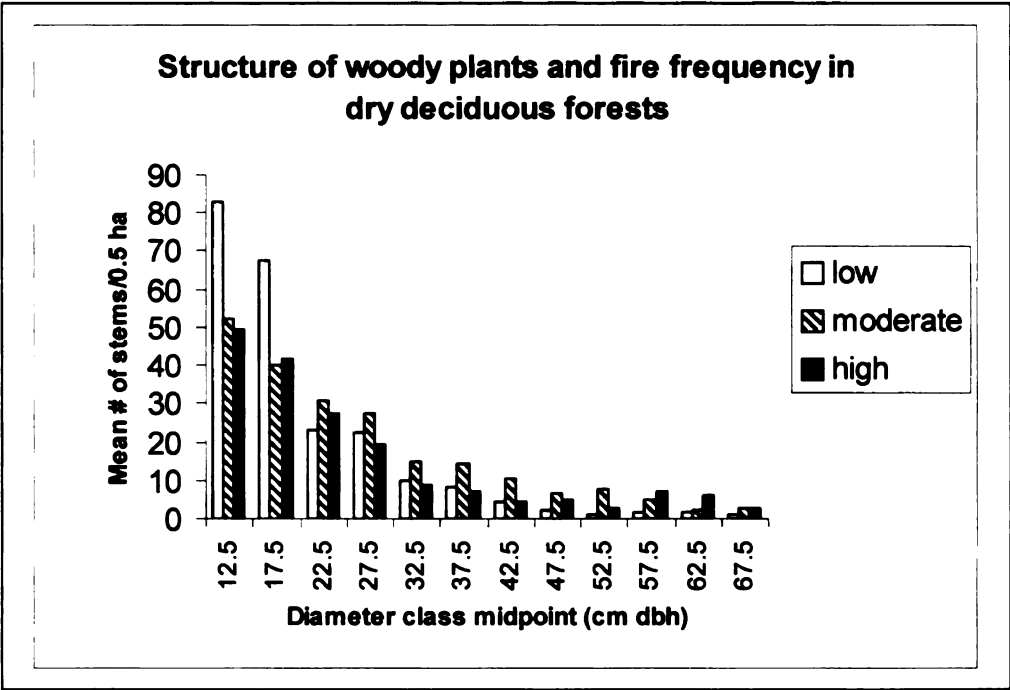


Figure 5-12: Woody plant species structure in dry thorn forest

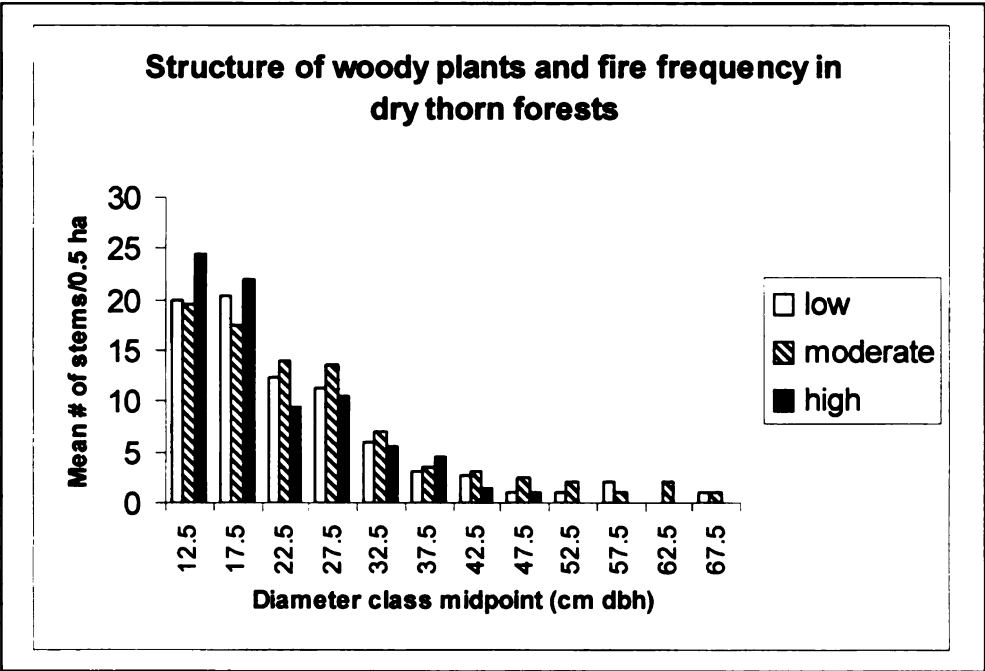


Figure 5-13: Woody plant species structure in moist deciduous forest

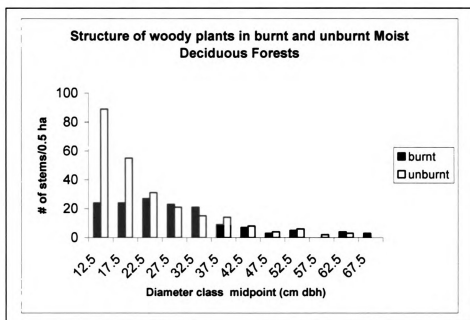


Figure 5-14: Fire frequency and species diversity in dry deciduous forest

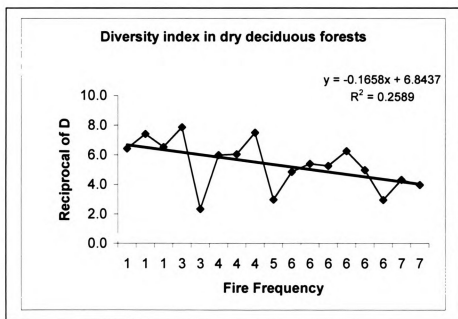


Figure 5-15: Fire frequency and species diversity in dry thorn forest

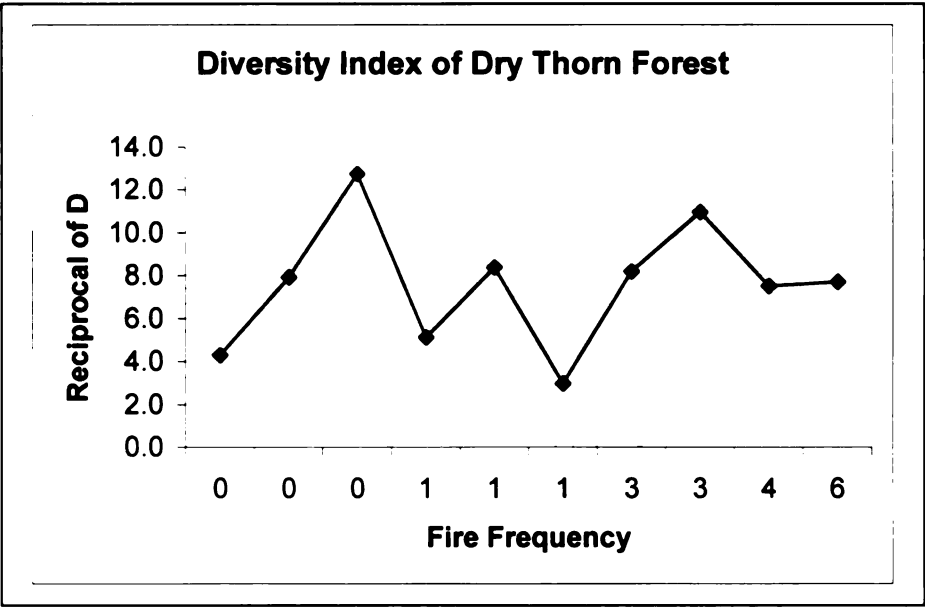


Figure 5-16: Fire frequency and species diversity in moist deciduous forest

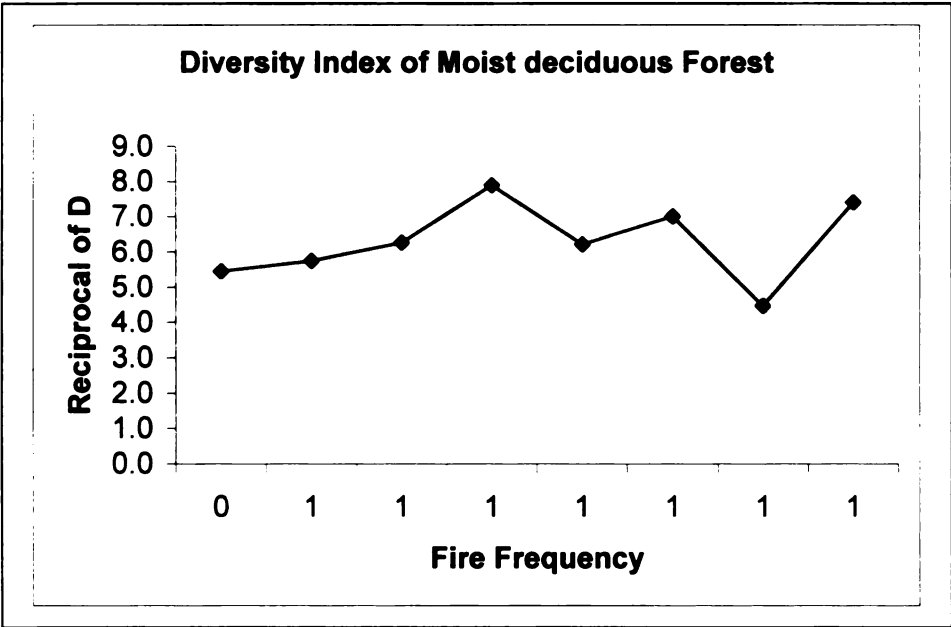


Figure 5-17: Fire frequency and equitability index in dry deciduous forest

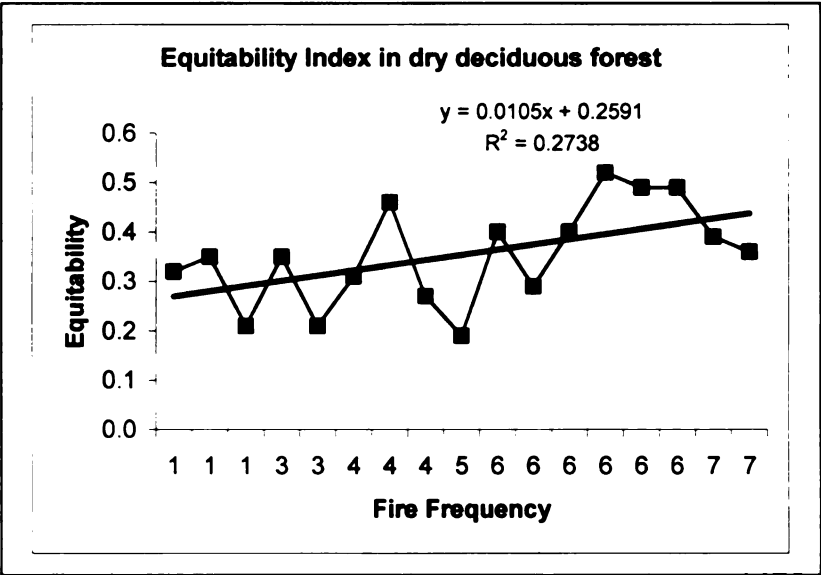


Figure 5-18: Fire frequency and equitability in moist deciduous forest

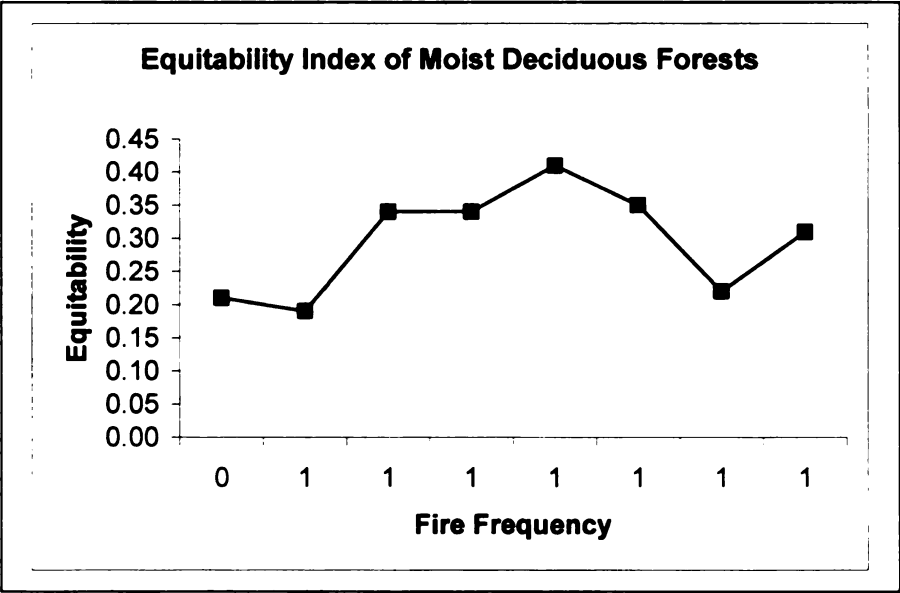


Figure 5-19: Fire frequency and equitability in dry thorn forest

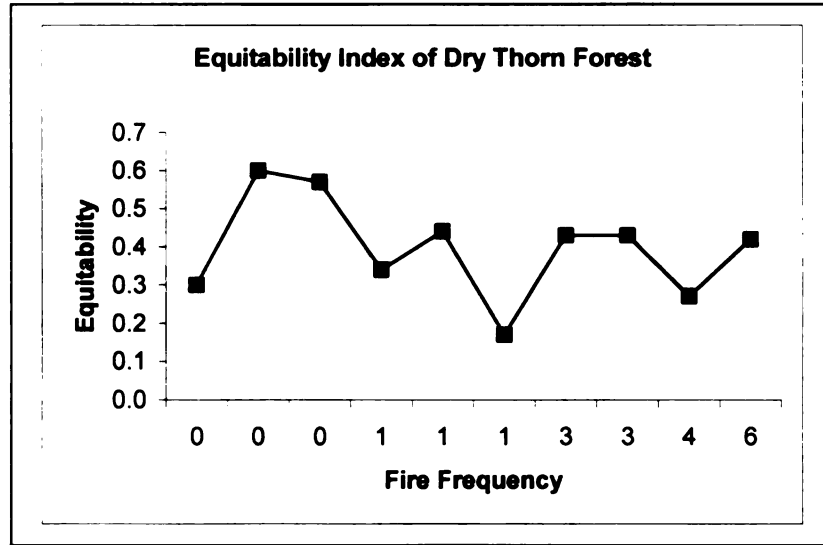


Figure 5-20: Stem distribution by size of *Tectona grandis* in Mudumalai

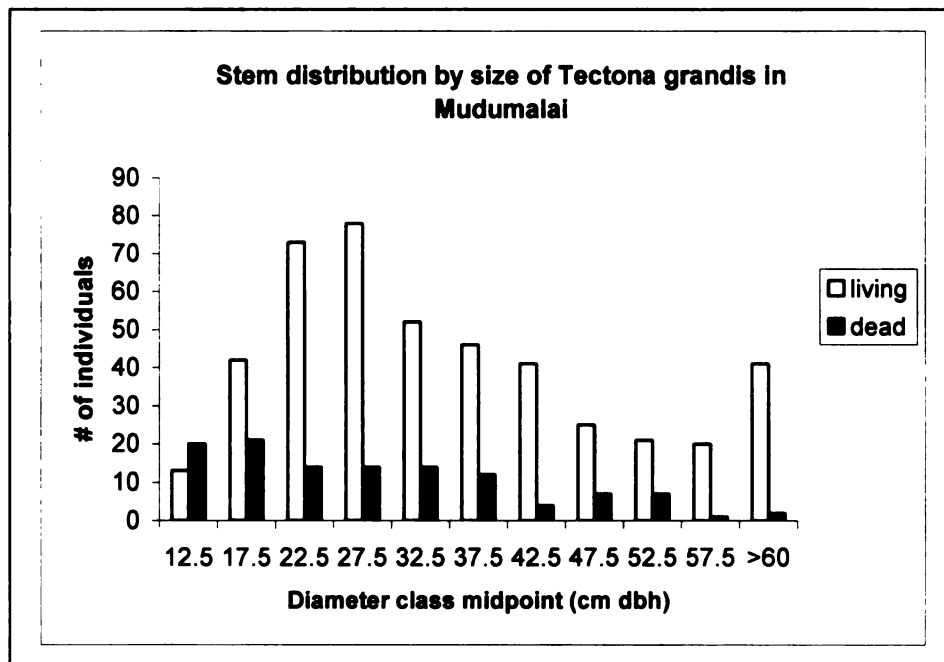


Figure 5-21: Stem distribution of *Anogeissus latifolia* in Mudumalai

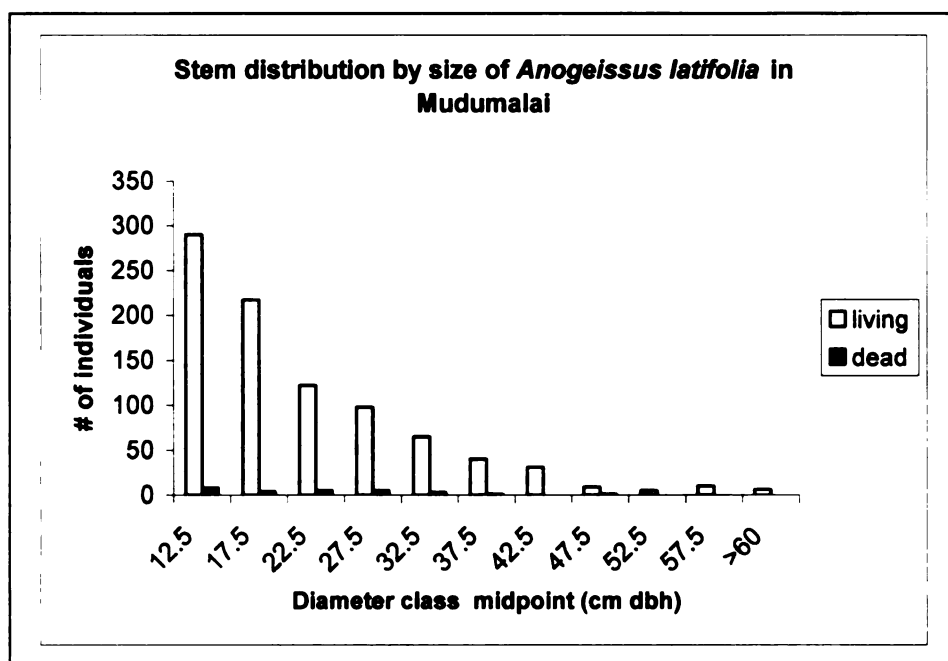


Figure 5-22: Stem distribution of *Terminalia crenulata* in Mudumalai

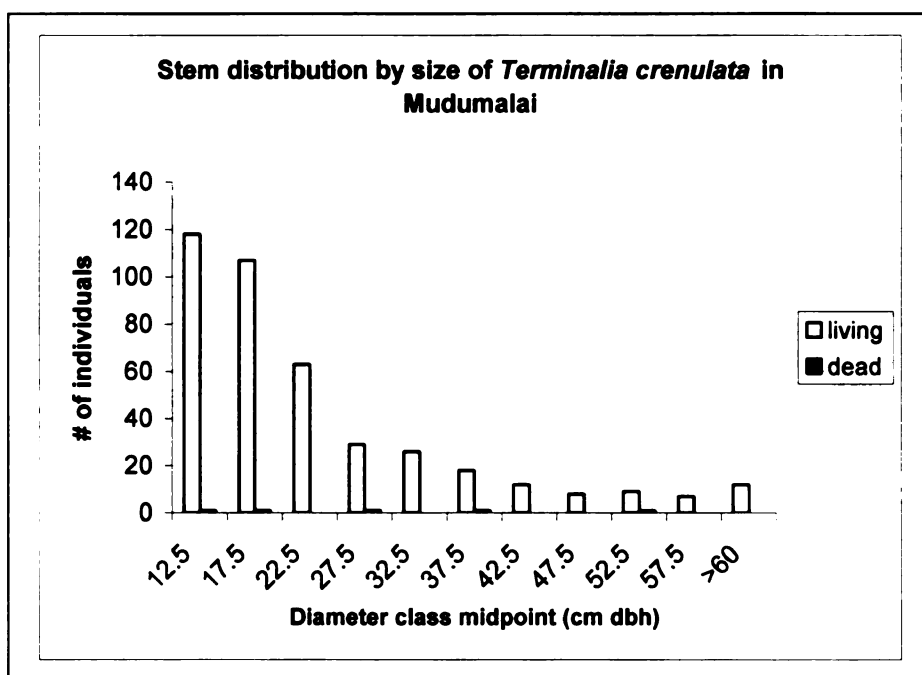


Figure 5-23: Disturbance and Diversity in the Nilgiri landscape

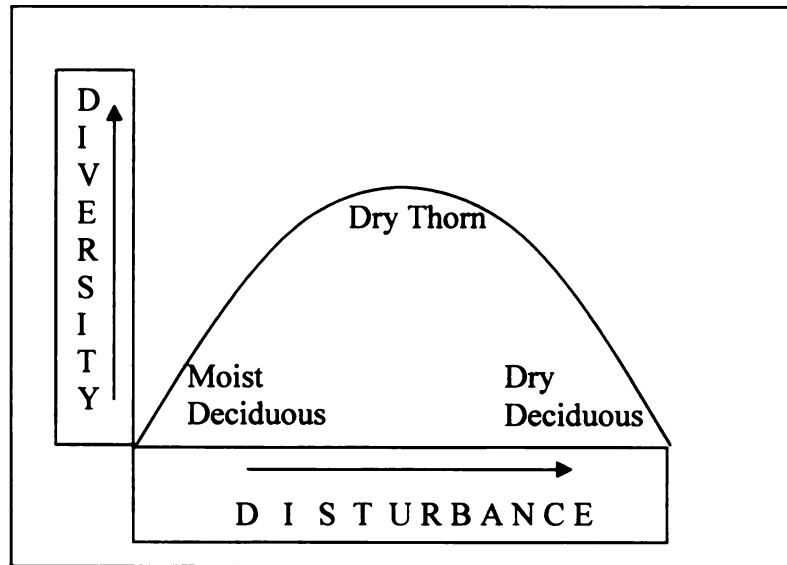


Table 5-1: Mudumalai climate data (adapted from Sukumar et al. 2004)

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total/ Average
Rain mm	7	9	88	119	129	160	146	130	126	170	115	50	1250
ADTMx °C	26.7	28.8	34.0	30.4	29.5	27.0	28.3	29.4	23.1	26.5	27.2	27.8	28.2
ADTMn °C	16.1	16.3	19.8	18.0	16.2	18.2	20.1	20.1	14.7	18.4	15.2	14.2	17.3
Solar Radiation Wm ⁻²	308	367	352	349	345	251	177	179	266	252	312	255	284

ADTMx: Average maximum daily temperature

ADTMn: Average minimum daily temperature

Table 5-2: Fuel size and composition in the Nilgiri landscape (Units Mgha⁻¹)

	Transects															Mean	
	1.42	1.13	1.24	1.56	1.58	1.26	2.22	.82	2.12	1.4	1.03	.24	.68	1.13	1.53	1.44	
1 hr ¹	1.5	.56	.47	.69	1.44	1.41	2.03	.28	1.69	1.09	1.53	.25	.97	1.34	.69	.59	1 ^a
10 hr ¹	5.32	4.7	1.25	13.2	6.6	1.9	.94	.31	2.2	10.7	0	.31	.63	0	1.3	4.7	3.4 ^a
100 hr ¹	13.9	23.9	14.9	10.6	5.3	9.6	3.3	5.6	51.2	31.7	12.7	.54	6.4	4.2	8	8.4	13.1 ^a
Sound ¹	0	0	0	.34	.23	0	0	0	.23	.71	.6	0	0	0	0	0	.13 ^a
Rotten ¹	22.1	30.3	17.8	26.3	15.1	14.1	8.5	7	45.6	57.4	15.9	1.3	8.67	6.7	11.5	15.2	18.9 ^a
Total ¹	1.47	.49	.43	.83	.83	.82	.62	.37	.6	.55							.7 ^b
1 hr ²	1.16	.88	.59	1.41	.19	.44	.34	.34	.41	.44							.62 ^{ab}
10 hr ²	0	0	0	4.07	.94	0	1.88	1.25	2.51	1.25							1.19 ^{ab}
100 hr ²	1.13	4.32	.39	1.26	3.68	1.02	2.24	2.06	6.23	2.86							2.5 ^b
Sound ²	0	0	0	.57	0	0	0	0	0	0							.05 ^{ab}
Rotten ²	3.76	5.69	1.41	8.14	5.64	2.28	5.08	4.02	9.78	5.07							5.1 ^b
Total ²	2.28	2.77	2.23	1.72	2.68	2.78											2.41 ^c
1 hr ³	1.41	2.47	1.75	1.09	1.94	2.25											1.81 ^c
10 hr ³	5.32	20.4	10	6.3	16.6	15.9											12.4 ^c
100 hr ³	2.75	9.51	17.5	14.3	13.9	4.1											10.35 ^{ac}
Sound ³	0	0	0	0	0	0											0 ^{ab}
Rotten ³	11.8	35.1	31.57	23.4	35.1	25.1											27 ^{ac}
Total ³																	

Note: superscripts in the first column indicate different vegetation types, 1 indicates dry deciduous, 2 indicates dry thorn forest, and 3 indicates moist deciduous forests. Group membership within columns (one way ANOVA, with Bonferroni multiple comparison test) $p < 0.05$

Table 5-3: Comparison between total areas burnt in the three vegetation types

	Total Area km ²	1996	1997	1999	2001	2002	2004	2005	Mean Area km ²
Dry Deciduous	1018	326.28	167.61	179.5	50.4	117.1	217	129.3	169.5 ^a
Moist Deciduous	148	9.63	4.2	2.97	.88	3.36	27.8	3.78	7.5 ^b
Dry Thorn	306	29.07	36.5	47.7	10.9	4.69	1.7	92.7	31.9 ^{bc}

Group membership within columns (one way ANOVA, with Bonferroni multiple comparison test) $p < 0.05$

Table 5-4: Comparison between mean fire sizes in the three vegetation types

	1996	1997	1999	2001	2002	2004	2005	Mean fire size km ²
Dry Deciduous	.69	.28	.51	.11	.3	.36	.72	.42 ^a
Moist Deciduous	.08	.05	.18	.02	.06	.09	.2	.1 ^b
Dry Thorn	.12	.18	.18	.22	.12	.08	.57	.21 ^{ab}

Group membership within columns (one way ANOVA, with Bonferroni multiple comparison test) $p < 0.05$

Table 5-5: Comparison between maximum fire sizes in the three vegetation types

	1996	1997	1999	2001	2002	2004	2005	Maximum fire size km ²
Dry Deciduous	99.2	42.7	61.04	14.8	80.2	75.3	53	60.9 ^a
Moist Deciduous	1.04	.46	1.49	.29	.4	14.8	2.9	3.05 ^b
Dry Thorn	4.2	19.5	4.2	10.04	2.02	.96	36.7	11.08 ^{bc}

Group membership within columns (one way ANOVA, with Bonferroni multiple comparison test) $p < 0.05$

Table 5-6: Data on seedling density and sapling abundance in burnt dry deciduous forests

Fire severity class	% mortality of total BA	Mean seedling density/100 m ²	Sapling abundance/0.2ha
0-5	4.58	109.35	52
0-5	1.88	66.05	42
0-5	1.02	76.3	13
0-5	0.96	90.3	1
0-5	2.25	66.6	5
0-5	4.03	98.2	32
5-10	8.55	60.7	41
5-10	6.97	43.05	7
5-10	8.73	19.25	3
5-10	8.04	45.7	35
5-10	8.52	61.3	90
10-15	10.94	84.7	90
10-15	11.94	37.15	6
10-15	12.91	77.3	71
>15	30.03	19.63	11
>15	15.17	35.8	27
>15	15.42	23.9	2

Basal Area refers to trees ≥ 10 cm dbh; Each row represents a transect

Table 5-7: Data on seedling density and sapling abundance in dry thorn forests

Fire severity class	% mortality of total basal area	Mean seedling density/100 m ²	Sapling abundance/0.2 ha
0-5	2.42	11.4	55
0-5	1.71	17.8	64
0-5	0	23.15	98
0-5	2.6	10	5
0-5	3.5	34.7	40
5-10	8.33	10	125
5-10	8.37	25.45	153
5-10	6.01	32.7	53
10-15	10.57	4.7	33
10-15	10.73	18.55	102

Basal Area refers to trees ≥ 10 cm dbh; Each row represents a transect

Table 5-8: Comparison of the effect of fire on seedlings in dry deciduous forests

	Seedlings /100 m ²	Seedlings /100 m ²	Seedlings /100 m ²	Seedlings /100 m ²	Seedlings /100 m ²	Seedlings /100 m ²	Mean
0-5	109.35	66.05	76.3	90.3	66.6	98.2	84.47 ^a
5-10	60.7	43.05	19.25	45.7	61.3		46 ^b
10-15	84.7	37.15	77.3				66.38 ^{ab}
> 15	19.63	35.8	23.9				26.4 ^{bc}

Group membership within columns (one way ANOVA, with Bonferroni multiple comparison test) $p < 0.05$

Table 5-9: Comparison of the effect of fire on seedlings and saplings in moist deciduous forests

Fire severity class	% mortality of total basal area	Mean seedling density/100 m ²	Mean Sapling abundance/0.2 ha
unburned	1.75	157 ^a	13.2 ^a
0-2	0.53	59 ^b	3.6 ^b
2-4	2.4	56.6 ^{bc}	4.7 ^{bc}

Group membership within columns (one way ANOVA, with Bonferroni multiple comparison test)

Table 5-10: Comparison of the effect of fire on seedlings in dry thorn forests

	Seedlings/ 100 m ²	Seedlings/ 100 m ²	Seedlings/ 100 m ²	Seedlings/ 100 m ²	Seedlings/ 100 m ²	Mean
0-5	9.85	17.8	23.015	9.85	34.7	19.95 ^a
5-10	10	25.45	32.7			15.62 ^a
10-15	4.7	18.55				20.85 ^a

Group membership within columns (one way ANOVA, with Bonferroni multiple comparison test)

Table 5-11: Comparison of the effect of fire on saplings in dry deciduous forests

	Saplings /0.2 ha	Saplings /0.2 ha	Saplings /0.2 ha	Saplings /0.2 ha	Saplings /0.2 ha	Saplings /0.2 ha	Mean
0-5	52	42	13	1	5	32	24.2 ^a
5-10	41	7	3	35	90		35.2 ^a
10-15	90	6	71				55.67 ^a
> 15	11	27	2				13.3 ^a

Table 5-12: Comparison of the effect of fire on saplings in dry thorn forests

	Saplings/ 0.2 ha	Saplings/ 0.2 ha	Saplings/ 0.2 ha	Saplings /0.2 ha	Saplings/ 0.2 ha	Mean
0-5	55	64	98	5	40	52.4 ^a
5-10	125	153	53			110.33 ^a
10-15	33	102				67.35 ^a

Group membership within columns (one way ANOVA, with Bonferroni multiple comparison test)

Table 5-13: Comparison of forest structure and fire frequency in dry deciduous forests

DBH mid point	Low Frequency N=3		Moderate Frequency N=3		High Frequency N=3	
	Total # of stems/0.5 ha	Total BA m ² /0.5 ha	Total # of stems/0.5 ha	Total BA m ² /0.5 ha	Total # of stems/0.5 ha	Total BA m ² /0.5 ha
12.5	249	2.61	156	2.05	148	2
17.5	203	3.08	121	2.88	125	3.1
22.5	69	2.85	93	3.74	83	3.1
27.5	67	3.78	83	4.69	57	3.3
32.5	29	2.7	45	3.74	27	2.1
37.5	24	2.11	42	4.79	22	2.4
42.5	13	1.34	32	4.94	13	1.7
47.5	4	.9	20	3.2	14	2.4
52.5	1	0	15	3.5	9	1.9
57.5	5	.53	10	2.9	7	2
62.5	3	.61	7	1.9	6	1.6
67.5	1	.33	6	1.8	3	1

Note: Entire forest type burned at least once during the past 16 yr period of analysis.

Table 5-14: Paired t tests between fire frequency classes in the dry deciduous forest

Table 5-14: Paired t tests between fire frequency classes in the dry deciduous forest

Pair	Mean difference	T value	df	P value
Low and moderate	-0.17	-2.86	11	< 0.05
Low and High	-0.14	-2.39	11	< 0.05
Moderate and High	0.031	0.75	11	> 0.1

Table 5-15: Comparison of forest structure and fire frequency in dry thorn forest

DBH mid point	Low Frequency N=3		Moderate Frequency N=2		High Frequency N=2	
	Total # of stems/0.5 ha	Total BA m ² /0.5 ha	Total # of stems/0.5 ha	Total BA m ² /0.5 ha	Total # of stems/0.5 ha	Total BA m ² /0.5 ha
12.5	60	1.02	39	1.58	49	.75
17.5	61	1.54	35	1.88	44	1.13
22.5	37	1.82	28	1.97	19	.97
27.5	34	2.17	27	2.23	21	1.19
32.5	18	1.7	14	1.85	11	.84
37.5	9	.84	7	1.8	9	.77
42.5	8	.98	6	1.9	3	.42
47.5	2	.56	5	1.28	2	.44
52.5	3	.22	4	1.9	0	0
57.5	2	.68	1	.78	0	0
62.5	0	0	2	1.51	0	0
67.5	1	.33	1	.35	0	0

Table 5-16: Paired t tests between fire frequency classes in the dry thorn forest

Pair	Mean difference	t value	df	P value
Low and moderate	-0.12	-1.36	11	> 0.1
Low and High	0.12	1.83	11	< 0.1
Moderate and High	0.25	2.68	11	< 0.05

Table 5-17: Comparison of forest structure and fire frequency in moist deciduous forests

DBH class mid point	Unburnt Forest N=1		Burnt Forest N=1	
	Total # of stems/0.5 ha	Total BA m ² /0.5 ha	Total # of stems/0.5 ha	Total BA m ² /0.5 ha
12.5	89	1.3	24	.33
17.5	55	1.5	24	0.6
22.5	31	1.5	27	1.08
27.5	21	1.4	23	1.53
32.5	15	1.18	21	1.75
37.5	14	1.46	9	1.03
42.5	8	1.34	7	1
47.5	4	.55	3	0.51
52.5	6	1.28	5	1.06
57.5	2	.54	0	0
62.5	3	.92	4	1.24
67.5	0	0	3	1.1

Table 5-18: Paired t tests between fire frequency classes in the moist deciduous forest

Pair	Mean	T/Z value	df	P value
Burned and Unburned	-0.15	-2.01	8	< 0.1
Nonparametric Wilcoxon test		-1.83		< 0.1

Table 5-19: Paired Samples Test between mortality across size classes in *Anogeissus latifolia*, *Tectona grandis*, and *Terminalia crenulata*

Pair	Mean difference	T value	df	P value
TERC-ANOL	-0.0088	-.6	10	> 0.1
TERC-TECG	-0.19	-4.08	10	< 0.01
ANOL-TECG	-0.18	-3.94	10	< 0.01

Chapter 6

Conclusions and future directions

I examined forest fires as a landscape disturbance event in a seasonally dry tropical ecosystem in the Western Ghats, India. Forest fires are almost annual disturbance events in the deciduous forests, my research on the spatial and temporal nature of these fires and their ecological effects has provided new insights into forest fires in this region.

My research has provided a synthesis of disturbances such as forest fires, past ecological history, variables such as topography, climate and vegetation patterns at the landscape scale. In the dry thorn forests, the past use of these forests by humans, the low rainfall, and low productivity has resulted in small scale disturbances that could be driving ecological patterns and species diversity. In the dry deciduous forests, the systematic logging operations have opened these forests, leading to the invasion by dense grasses, this combined with the high productivity rates, the grass-fire cycle, and lack of regeneration is leading to the impoverization of the forest and resulting in species poor forests. In the undisturbed moist deciduous forests, the microclimate and the dense fuel beds are leading to the absence of fire in the forests, however in the disturbed forests, the invasions by grasses has altered the fire regime and the species composition. Further the moist deciduous forests are extremely sensitive to changes in climatic patterns, years of below average rainfall and drought conditions could potentially lead to extensive fires.

Different methods are currently adopted to map and delineate fires in forests in India. The application of remote sensing data to the analysis of forest fires as demonstrated in chapter two is advantages as it provides accurate spatial information on fire in the landscape. Further the presence of large datasets could be harnessed to understand the temporal pattern of fires in the landscape. The application of simple supervised classification algorithms to detect and classify burned areas was possible due to the deciduous nature of trees in these forests. A large component of forest area in the Western Ghats is under deciduous forests and similar techniques could be adopted to map, assess, and monitor disturbances within these forests.

In order to assess the fire-return interval across different vegetation types, I compiled spatial information on the extent of fires in the Mudumalai wildlife sanctuary between 1989 and 2002. The fire-return interval is between 3 and 4 years for the sanctuary area, I compared this with an earlier time period in the 1920s and found the fire-return interval to be about 10 years, suggesting a three fold increase in fire-return intervals in the 80 year time period. Pairwise comparisons of the temporal pattern of fires in the sanctuary showed significant differences in the areas under fire in the different vegetation types ($p < 0.05$).

The spatial and temporal characteristics of forests fires in the landscape are important to assess the fire regime. Chapter four provides information on the occurrence of fire with reference to the park boundary. The linear model shows that fire-return interval is relatively long close to the park boundary, but decreases with distance from the park

boundary ($R^2=0.3$). In this chapter I also present a statistical model that determines the important variables critical to fire occurrence and spread in the landscape. Elevation, forest fractional coverage, and rainfall are important to explain the occurrence of fire in the landscape. In developing the statistical model, two important correlations confound the predictions. I incorporated spatial data analysis techniques and variogram modeling to determine the extent of temporal and spatial autocorrelations in the dataset. Spatial autocorrelation is more important in modeling fire in the Nilgiri landscape than temporal autocorrelation.

Apart from the spatial and temporal characteristics of fires at the broad landscape scale, I also assessed the fire regime in terms of the sizes of the fires, the intensity of the fires, the severity of the fires in terms of its effects on the regeneration, structure, and diversity of forests, chapter five provides information on these aspects of fire in the Nilgiri landscape. The study of fires between 1996 and 2005 shows that the mean fire sizes in tropical dry deciduous forests were four fold larger than mean fire sizes in tropical moist deciduous forests and two fold larger than tropical dry thorn forests. Total fuel loads are significantly higher in the tropical moist and tropical dry deciduous forests compared with tropical dry thorn forests. Maximum fire size in the tropical dry deciduous forest was 20 fold larger than moist deciduous forest while it was only 6 fold larger than the tropical dry thorn forest. Fire severity is especially significant in the tropical moist deciduous and tropical dry deciduous forests, with effects on the regeneration, structure, and diversity within these two forest types.

Future Challenges

Apart from answering many of the questions laid out in the research, my dissertation research has opened up a number of issues and challenges for conservation in the tropics.

Some important research questions, as a direct consequence of this research, include the understanding of the fire regime in the tropical evergreen ecosystems. Although spatial extent of fires similar to the deciduous ecosystems in the Western Ghats are not feasible in evergreen forests, the small fires that have been reported in disturbed and derived forests may be worth exploring further. The ecological effects of these fires could have large effects due to the absence of fire in the evolution of these forests.

As the consequences of global climate change and warming become a reality, changes in climatic patterns and alterations in the microclimate of forests especially in human dominated ecosystems presents threats to biodiversity due to changes in fire regimes. Future studies will have to closely examine the link between climate changes and possible exacerbation of wildfires in the Western Ghats. About 80% of the biomass burning occurs in the tropics and are directly related to anthropogenic sources. Emissions from these forest fires which include methane, carbon dioxide, oxides of nitrogen could have significant impacts on the concentration of greenhouse gases. Further research on climate change, fire regimes and their feedbacks will be an important line of research for the future.

Due to time and other constraints this research has focused on the effects of fire on woody plant species. The Western Ghats are home to a rich assemblage of fauna, the consequences of the current fire regime to different aspects of the ecology of these animals is certainly an area of potential future research. In parts of Africa, there exists a close relationship between vegetation dynamics and fires and grazing by wildlife present in these ecosystems. It will be interesting to understand these interactions in the Western Ghats in India.

Synergistic components of the fire regime have not been quantified in this research. Synergistic interactions between forest fragmentation and forest fires have been shown to alter the fire regime. Future research efforts in the Western Ghats will require an understanding of how these processes are playing out in the Western Ghats and affecting fire regimes and vegetation.

Fire has been incorporated into the ecosystem for many millennia; conservation practices will require an acknowledgement of the use of fire in the landscape. About 80 million people in South Asia live within or on the fringes of forests and some of these communities obtain a significant proportion of annual income directly from forest resources. Integrating both the social and ecological components of fires into fire management could provide new insights into conservation practices and result in better management of natural resources in the Western Ghats.

This dissertation provides a conceptual framework for future studies to build on in assessing forest fires and their ecological effects in the Western Ghats in India. The application of analysis at the landscape scale has provided many key insights into the spread of fire and its impacts on the landscape vegetation pattern. The application of novel techniques in remote sensing and GIS has provided a method to arrive at the temporal and spatial characteristics of disturbances in the ecosystem. Above all this research demonstrates the importance of a multi disciplinary approach in examining questions at the landscape scale.

Appendix 1: List of transects and locations in the three vegetation types in the Nilgiri landscape

Transect	# of species	Longitude	Latitude	Forest Type
dd1	6	76 35' 47"	11 34' 57"	Dry Deciduous
dd2	11	76 31' 29"	11 38' 17"	Dry Deciduous
dd3	10	76 35' 51"	11 33' 56"	Dry Deciduous
dd4	12	76 36' 3"	11 34' 8"	Dry Deciduous
dd5	13	76 35' 51"	11 33' 56"	Dry Deciduous
dd6	11	76 36' 19"	11 33' 54"	Dry Deciduous
dd7	22	76 25' 51"	11 36' 34"	Dry Deciduous
dd8	20	76 27' 4"	11 36' 52"	Dry Deciduous
dd9	18	76 27' 3"	11 36' 55"	Dry Deciduous
dd10	13	76 37' 3"	11 36' 14"	Dry Deciduous
dd11	15	76 35' 15"	11 36' 47"	Dry Deciduous
dd12	11	76 31' 27"	11 37' 45"	Dry Deciduous
dd13	12	76 28' 14"	11 37' 2"	Dry Deciduous
dd14	19	76 28' 30"	11 36' 58"	Dry Deciduous
dd15	20	76 38' 12"	11 38' 2"	Dry Deciduous
dd16	21	76 38' 4"	11 38' 5"	Dry Deciduous
dd17	31	76 38' 18"	11 37' 52"	Dry Deciduous
dt1	19	76 49' 26"	11 35' 30"	Dry Thorn
dt2	25	76 49' 43"	11 35' 34"	Dry Thorn
dt3	19	76 41' 4"	11 36' 21"	Dry Thorn
dt4	22	76 40' 53"	11 36' 37"	Dry Thorn
dt5	17	76 39' 36"	11 36' 36"	Dry Thorn
dt6	13	76 47' 15"	11 34' 6"	Dry Thorn
dt7	18	76 39' 25"	11 33' 32"	Dry Thorn
dt8	14	76 39' 25"	11 33' 29"	Dry Thorn
dt9	27	76 39' 24"	11 37' 20"	Dry Thorn
dt10	15	76 39' 6"	11 37' 26"	Dry Thorn
md1	25	76 24' 8"	11 36' 7"	Moist Deciduous
md2	29	76 17' 2"	11 27' 17"	Moist Deciduous
md3	18	76 16' 42"	11 27' 56"	Moist Deciduous
md4	23	76 16' 42"	11 27' 55"	Moist Deciduous
md5	15	76 14' 30	11 27' 42	Moist Deciduous
md6	20	76 14' 55'	11 28' 2"	Moist Deciduous
md7	20	76 14' 47"	11 27' 55"	Moist Deciduous
md8	24	76 14' 20"	11 47' 49"	Moist Deciduous

Appendix 2: List of species found in the transects in the Nilgiri landscape

Code	Botanical Name	Distribution
ACAC	<i>Acacia chundra</i>	Dry Thorn
ACAF	<i>Acacia ferruginea</i>	Dry Thorn
ACAL	<i>Acacia leucophloea</i>	Dry Thorn
ACTM	<i>Actinodaphne malabarica</i>	Moist Deciduous
ALBA	<i>Albizia amara</i>	Dry Thorn
ANOL	<i>Anogeissus latifolia</i>	Dry Deciduous, Dry Thorn
APHP	<i>Aphanamixis polystachya</i>	Moist Deciduous
ATLW	<i>Atlantia wightii</i>	Dry Thorn
AZAI	<i>Azadirachta indica</i>	Dry Thorn
BAUR	<i>Bauhinia racemosa</i>	Dry Thorn
BAUS	<i>Bauhinia species</i>	Dry Deciduous
BISJ	<i>Bischofia javanica</i>	Dry Deciduous
BOMC	<i>Bombax ceiba</i>	Dry Deciduous, Dry Thorn
BOMM	<i>Bombax malabaricum</i>	Moist Deciduous, Dry Deciduous
BRIR	<i>Bridelia retusa</i>	Dry Deciduous, Moist Deciduous, Dry Thorn
BUTM	<i>Butea monosperma</i>	Dry Deciduous, Dry Thorn
CAND	<i>Canthium diccocum</i>	Dry Thorn
CANP	<i>Canthium parviflorum</i>	Dry Thorn
CAPZ	<i>Capparis zeylanica</i>	Dry Thorn
CARA	<i>Careya arborea</i>	Dry Deciduous
CASE	<i>Casearia esculenta</i>	Dry Deciduous
CASF	<i>Cassia fistula</i>	Dry Deciduous, Moist Deciduous, Dry Thorn
CASO	<i>Casearia ovoides</i>	Moist Deciduous
CHLS	<i>Chloroxylon switenia</i>	Dry Thorn
CINM	<i>Cinnamomum malabathrum</i>	Moist Deciduous
CLES	<i>Clerodendrum serratum</i>	Moist Deciduous
CORO	<i>Cordia obliqua</i>	Dry Deciduous, Moist Deciduous, Dry Thorn
CORW	<i>Cordia wallichii</i>	Dry Deciduous, Dry Thorn
DALL	<i>Dalbergia latifolia</i>	Dry Deciduous, Moist Deciduous, Dry Thorn
DALL	<i>Dalbergia lanceolaria</i>	Dry Thorn, Moist Deciduous
DICC	<i>Dichrostachys cineatea</i>	Dry Thorn
DILP	<i>Dillenia pentagyna</i>	Moist Deciduous
DIOM	<i>Diospyros Montana</i>	Dry Thorn, Dry Deciduous
EHEC	<i>Ehertia concanensis</i>	Dry Thorn
ELAG	<i>Elaeodendron glaucum</i>	Dry Thorn, Dry Deciduous
ELAT	<i>Elaeocarpus tuberculatus</i>	Dry Deciduous, Moist Deciduous
ERIQ	<i>Eriolaena quinquelocularis</i>	Dry Deciduous
ERYI	<i>Erythrina indica</i>	Dry Deciduous
ERYM	<i>Erythroxylon monogynum</i>	Dry Thorn
FICD	<i>Ficus drupacea</i>	Dry Deciduous, Moist Deciduous
FICH	<i>Ficus hispida</i>	Dry Thorn
FICT	<i>Ficus tsjahela</i>	Moist Deciduous
FICR	<i>Ficus religiosa</i>	Dry Thorn, Moist Deciduous
FLAI	<i>Flacourtia indica</i>	Dry Deciduous, Dry Thorn
GARP	<i>Garuga pinnata</i>	Dry Deciduous, Moist Deciduous
GART	<i>Gardenia turgida</i>	Dry Deciduous
GIVR	<i>Givotia rottleriformis</i>	Dry Thorn, Dry Deciduous
GLOM	<i>Glochidion malabaricum</i>	Dry Deciduous, Moist Deciduous
GMEA	<i>Gmelina arborea</i>	Dry Deciduous, Dry Thorn
GREO	<i>Grewia orbiculata</i>	Dry Deciduous, Dry Thorn
GRET	<i>Grewia tiliifolia</i>	Dry Deciduous, Moist Deciduous
HALC	<i>Haldenia cordifolia</i>	Moist Deciduous
HARB	<i>Hardwickia binata</i>	Dry Thorn
HOPP	<i>Hopea parviflora</i>	Moist Deciduous

HYMO	<i>Hymenodictyon orixense</i>	Dry Deciduous
IXON	<i>Ixora nigricans</i>	Moist Deciduous, Dry Thorn
KYDC	<i>Kydia calycina</i>	Dry Deciduous
LAGL	<i>Lagerstroemia lanceolata</i>	Dry Deciduous, Moist Deciduous
LAGP	<i>Lagerstroemia parvifolia</i>	Dry Deciduous, Moist Deciduous
MITP	<i>Mitragyna parvifolia</i>	Dry Thorn, Moist Deciduous
MACP	<i>Macaranga peltata</i>	Moist Deciduous
MADI	<i>Madhuca indica</i>	Dry Deciduous, Dry Thorn
MADL	<i>Madhuca longifolia</i>	Dry Deciduous, Dry Thorn
MANI	<i>Mangifera indica</i>	Moist Deciduous
MAYE	<i>Maytenus emarginatus</i>	Dry Deciduous, Dry Thorn
MELP	<i>Meliosma pinnata</i>	Moist Deciduous
MORO	<i>Moringa oleifera</i>	Dry Thorn
NARC	<i>Naringi crenulata</i>	Dry Thorn
OUGO	<i>Ougeinia oojeinensis</i>	Dry Deciduous
OLED	<i>Olea dioica</i>	Moist Deciduous
OLEG	<i>Olea glandulifera</i>	Moist Deciduous
PERM	<i>Persea macrantha</i>	Moist Deciduous
PHYE	<i>Phyllanthus emblica</i>	Dry Deciduous, Dry Thorn, Moist Deciduous
PRET	<i>Premna tomentosa</i>	Dry Deciduous, Dry Thorn
PTM	<i>Pterocarpus marsupium</i>	Dry Deciduous, Moist Deciduous, Dry Thorn
RADX	<i>Radermachera xylocarpa</i>	Dry Deciduous, Moist Deciduous
RAND	<i>Randia dumetorum</i>	Dry Deciduous, Moist Deciduous
SANA	<i>Santalum album</i>	Dry Thorn
SAPE	<i>Sapindus emarginatus</i>	Dry Thorn, Moist Deciduous
SCHO	<i>Schleichera oleosa</i>	Dry Deciduous, Moist Deciduous
SCHS	<i>Schrebera switenioides</i>	Moist Deciduous, Dry Deciduous
SCOC	<i>Sclopia crenata</i>	Moist Deciduous
SHOR	<i>Shorea roxburghii</i>	Dry Deciduous
STEA	<i>Stereospermum atrovirens</i>	Dry Deciduous, Dry Thorn
STEG	<i>Sterculia guttata</i>	Moist Deciduous
STEP	<i>Stereospermum personatum</i>	Dry Deciduous, Moist Deciduous
STEU	<i>Sterculia urens</i>	Dry Deciduous, Moist Deciduous
STRP	<i>Strychnos potatorum</i>	Dry Thorn, Moist Deciduous
SYZC	<i>Syzygium cumini</i>	Dry Deciduous, Moist Deciduous, Dry Thorn
TAMI	<i>Tamarindus indicus</i>	Dry Thorn
TECG	<i>Tectonia grandis</i>	Dry Deciduous, Dry Thorn, Dry Deciduous
TERB	<i>Terminalia bellerica</i>	Dry Deciduous, Moist Deciduous
TERC	<i>Terminalia crenulata</i>	Dry Deciduous, Moist Deciduous
TERCH	<i>Terminalia chebula</i>	Dry Deciduous, Moist Deciduous, Dry Thorn
TERP	<i>Terminalia paniculata</i>	Moist Deciduous
TERT	<i>Terminalia tomentosa</i>	Dry Deciduous, Moist Deciduous
TODA	<i>Todalia acculeata</i>	Moist Deciduous
UNI1	<i>Unidentified species</i>	Dry Thorn, Moist Deciduous
UNI2	<i>Unidentified species</i>	Dry Thorn
UNI3	<i>Unidentified species</i>	Moist Deciduous
VIBP	<i>Viurnum punctatum</i>	Dry Deciduous, Moist Deciduous
VITA	<i>Vitex altissima</i>	Moist Deciduous
WRIT	<i>Wrightia tinctoria</i>	Dry Thorn, Moist Deciduous
XERS	<i>Xeromphis spinosa</i>	Dry Deciduous, Moist Deciduous
ZIZM	<i>Ziziphus mauritiana</i>	Dry Thorn
ZIZR	<i>Ziziphus rugosa</i>	Dry Thorn, Dry Deciduous
ZIZX	<i>Ziziphus xylopyros</i>	Dry Deciduous, Moist Deciduous, Dry Thorn

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