ABUNDANCE AND DISTRIBUTION OF AN ALIEN PLANT PANAMA RUBBER (*Castilla elastica*) IN AMANI NATURE RESERVE, TANZANIA

BY

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ABSTRACT

Castilla elastica an exotic tree species to Tanzania was reported as a potential invasive plant in Amani Nature Reserve (ANR). In order to provide information toward efforts to combat alien invasive plants in ANR, the research was undertaken to study the abundance and distribution of *Castilla elastica* in the Reserve. To locate definite plots in the Amani Botanical Garden originally planted with *Castilla*, reports on the botanical survey in ANR by Greenway (1934) and Hones (1963) were used. The point-centred quarter method was used for the determination of the relative abundance of *Castilla* and other constituent species of the canopy in the study area. Furthermore, the population characteristics of *Castilla* were studied and its Weed Risk Assessment (WRA) was carried out. The distribution of *Castilla* in the reserve falls within the altitudinal range of 380 m to just over 600 m asl. There was significant difference ($F_{3\ 16} = 28.2$, $P < 0.001$) in density of *Castilla* between the four levels of forest disturbance in the study area. The density of *Castilla* in areas of high disturbance was approximately 192 stems/ha, 98 stems/ha in moderately disturbed, 63 stems/ha in plantation forest and 26 stems/ha in areas of low disturbance. The diameter distribution of *Castilla* suggests an expanding population with 55% seedlings, 30% saplings, 10% juveniles, 3% young trees, and 2% mature trees. The sex ratio of *Castilla* in ANR is female-biased and presence or absence of male individuals in a population does not significantly (t-test, $P < 0.05$) influence the number of regenerants produced. The total score obtained from the WRA of *Castilla* in ANR is 11 and this shows that *Castilla* has a risk of becoming invasive in the

reserve. Therefore, this calls for close monitoring and control of its population growth in ANR.

DECLARATION

I, **John Richard** do hereby declare to the Senate of Sokoine University of Agriculture that this dissertation is my own original work and has not been submitted to any other university for a degree award

02/11/2007

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The above declaration confirmed

7/11/2007

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DEDICATION

This work is dedicated to Almighty God (Allah), through whose powerful hands, this work was possible.

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ABBREVIATIONS

ABG	-Amani Botanical Garden
ANR	-Amani Nature Reserve
asl	-above sea level
CAD	-Canadian dollar
CSIRO	-Commonwealth Scientific and Industrial Research Organization
dbh	-diameter at breast height
SCOPE	-Scientific Committee on Problems of the Environment
TBA	-Tropical Biology Association
WRA	-Weed Risk Assessment

CHAPTER ONE

1.0 INTRODUCTION

1.1 Background Information

Alien trees and shrubs contribute significantly to the economies of many countries by providing a wide range of products and services. Hundreds of alien species have been widely and successfully planted for a variety of purposes including afforestation, desertification and erosion control, and for the supply of fuelwood and other products (Richardson, 1999; Meyer and Malet, 2006). These species are highly preferred because of their exceptional adaptability to a wide variety of sites, their rapid growth, and multiple uses of their products. However, introduction of alien species has led to serious problems into many forest ecosystems. In the absence of their natural predators, competitors, and pathogens, they prosper in their new environment and spread at the expense of native species, affecting entire ecosystems (FAO, 2005).

Intentional introduction of alien plants in Tanzania, particularly in East Usambara Mountains can be traced back to early 1890s, when a small botanical garden was started at Amani (Schulman *et al.*, 1998; Frontier, 2001). In 1902, more than three hundred hectares of land were set-aside at Amani to expand the botanical garden (Hamilton and Bensted-Smith, 1989), where about 1000 different indigenous and exotic species from similar ecological zones in the tropics were planted (Sandy *et al.*, 1997). *Castilla elastica* (Panama rubber), which was the early source of rubber, is among the commercial crops that were intentionally introduced to Amani. Engler, cited in Markus (2006) reported that *Castilla elastica* was among the 859 specimens of commercially important tropical plants that were provided by the Royal Botanic Garden of Berlin to Amani Botanical Garden (ABG). Some of these exotic plant species that were introduced through the botanical garden are reported to invade native habitats of Amani Nature Reserve (ANR). The invasion was mainly facilitated by logging activities, which had been carried out in the East Usambara for more than 50 years before it was banned in the year 1987 (Hamilton, 1989), therefore this opened large gaps that facilitated establishment of these plants. The ABG and associated facilities are situated within ANR in the East Usambara Mountains, a globally recognized centre of biodiversity and one of the most valuable conservation areas in Africa (Maunder, 1999). Whilst a botanical garden can play an important role in the conservation of biodiversity it should also be recognised that a collection of exotic species can represent a conservation liability.

1.2 Problem Statement and Justification

The biodiversity of ANR is threatened by the spreading of alien species (Hamilton and Bensted-Smith, 1989). Studies by Maunder (1999) at ANR identified 15 alien plant species that are locally important invasive species. *Castilla elastica* is among these plant species, and it is reported to spread from the botanical garden to other parts of the Reserve. In Samoa, French Polynesia, and Australia where *Castilla* was also introduced, it has proven to be very invasive and therefore threatens the biodiversity of these areas (Space and Flynn, 2002).

Although more than seven years have elapsed since Maunder (1999) alerted about the potential of *Castilla elastica* to invade Amani ecosystem, there is still very limited knowledge which has not even been documented on the abundance and distribution of this plant species in ANR. The present study will therefore provide baseline information on the abundance and distribution of *Castilla elastica* and consequently assist towards the efforts to combat alien invasive plants in Amani Nature Reserve.

1.3 Objectives

1.3.1 Overall objective

The overall objective of this study is to assess the abundance and distribution of an alien plant *Castilla elastica* in Amani Nature Reserve.

1.3.2 Specific objectives

i) To determine dominance of tree species in the study area.
ii) To investigate the relative abundance of *Castilla elastica* in various levels of disturbance.
iii) To assess the population structure of *Castilla elastica* in Amani Nature Reserve.
iv) To carry out risk assessment of *Castilla elastica* in Amani Nature Reserve.

1.3.3 Research questions

This study was therefore guided by the following questions:

i) Which species dominate the forest communities in the study area?

ii) Does the relative abundance of *Castilla elastica* differ in different disturbance levels in the study area?

iii) Is regeneration of *C. elastica* influenced by its sex ratio?

iv) Is the population of *C. elastica* stable in terms of reproductive ratio and sex ratio?

v) Does *C. elastica* pose risk of invasion in Amani Nature Reserve?

1.3.4 Hypotheses

Null hypothesis 1: *C. elastica* does not significantly dominate the forest communities it invades.

Null hypothesis 2: The rate of invasion of *C. elastica* is independent of the level of disturbance.

CHAPTER TWO

2.0 LITERATURE REVIEW

2.1 Alien Invasive Species: An Overview

In recent years alien invasive species have gained considerable notoriety as major threats to indigenous species and ecosystems. Alien invasive species can be defined as any species that are non-native to a particular ecosystem and whose introduction and spread cause, or are likely to cause, socio-cultural, economic or environmental harm or harm to human health (FAO, 2005). All over the world, invasive species are considered undesirable in conservation areas because they may modify natural and semi-natural habitats (Binggeli and Hamilton, 1993), this can be by replacing a diverse system with single species stands, introducing a new life form to the habitat, altering the water or fire regime, changing the nutrient status of the soil and humus, removing a food source or introducing a food source where none existed before, or altering sedimentation processes (Vitousek, 1987; Ramakrishnan, 1991). Such alteration may have profound effects on the composition of both flora and fauna and on the landscape as a whole.

The problem of biological invasions has been recognised by the Scientific Committee on Problems of the Environment (SCOPE) as a central problem in the conservation of biological communities. Invasion of habitats by alien organisms is considered a serious threat by conservationists due to the fact that it is a lasting and pervasive threat. It is a lasting threat because when exploitation or pollution stops, ecosystems often begin to recover. However, when the introduction of alien organisms stops the existing aliens do not disappear; in contrast they sometimes continue to spread and consolidate, and so may be causing a more pervasive threat (Cronk and Fuller, 1995).

2.2 Introduction and Spread of Alien Species

The increasing global movement of people and products, though beneficial to many people, is also facilitating the movement of alien species around the world. These species may be unintentionally introduced to new environments in shipments of food, household goods, wood and wood products, used tires, animal and plants products, containers, pallets, internal packaging material, and by human travel. However, species that are challenging to natural resource management are non-native species that have been intentionally introduced into an ecosystem to provide economic, environmental or social benefits (FAO, 2005). Intentional introduction began long ago, ancient human migrations and trade led to the early spread of some domesticated species such as cereals, dates, rice, cattle, and fowl, alongside with the inadvertent spread of their parasites. The number of species that have entered new ranges through human agency has increased by orders of magnitude in the past 500 years, and especially in the past 200 years because of significant increase in human movements.

In recent past, there were very few habitats on earth that were free of species introduced by humans, and these were those unique habitats that were considered immune, for example locations above 80^0 latitude (Mack *et al.*, 2000). Global factors, both primary and secondary, that support the introduction and spread of alien species include: land use changes, economic and trade, climate change and changes in atmospheric composition, tourism, conflict and reconstruction, regulatory regimes, biological control of pests, and public health (FAO, 2005). A highlight of mechanisms of spread and impacts to natural environment due to invasion by some alien plants namely; *Leucaena leucocephala*, *Lantana camara,* and *Clidemia hirta* is given below.

2.2.1 *Leucaena leucocephala*

This is a fast growing, nitrogen-fixing and drought-tolerant tree native to Mexico and Central America (Mathew, 2004; Mathew and Brand, 2004). It was introduced to the Pacific islands by the Spanish who over 400 years ago transported *L. leucocephala* feed and seeds to the Philippines. The shrub was also introduced to Hawaii in 1864 and to the Marquesas Islands prior to 1893. It is now common in the lowlands of many Pacific islands often forming monotypic stands (Binggeli, 1998). Subspecies *leucocephala* was introduced to Asia over two centuries ago and is now established around the world. Subspecies *glabrata* was widely introduced across the tropics in the 1970s and 1980s by agroforestry groups and organizations and is now widely cultivated throughout the tropics (Mathew, 2004). For the past three decades, *L. leucocephala* has been advocated all over Eastern Africa as an efficient agroforestry tree for soil amelioration through nitrogen fixation (Lulandala, 1985).

Leucaena leucocephala is spreading naturally and is considered a weed in more than 20 countries across all continents except Europe and Antarctica. It invades open, often coastal or riverine habitats, disturbed sites and agricultural lands. It forms dense thickets which in some areas are replacing native forests and threatening endemic species of conservation concern (FAO, 2005). Such dense thickets also render extensive areas of land unusable and inaccessible, and once established the species is very difficult to eradicate. From introduced areas in the tropics, *L. leucocephala* is spread by birds and pollinated by wind. Their fruits are small and therefore produce many seeds which make it possible for invasion. *Leucaena leucocephala* is considered a conflict tree because although it is invasive and destructive, it can also provide many positive benefits. Widely promoted as a miracle tree, *L. leucocephala* has been planted for the provision of wood and non-wood products and for the reforestation purposes (Lulandala, 1985; Mathew and Brand 2004).

2.2.2 *Lantana camara*

Lantana camara is an invasive shrub native to Central and South America. It was introduced during the late 19th century throughout the tropics and subtropics as an ornamental shrub which was often used for hedges and erosion control. In these regions, *L. camara* has established and become a major weed of pastures, roadsides, wastelands and plantations (Mathew, 2004). It thrives in dry and wet regions and often grows in valleys, mountain slopes, and coastal areas. The plant flowers all the year in many warm countries and has many varieties and colour forms. *Lantana camara* is somewhat shade-tolerant and, therefore, can become the dominant understorey in open forests or in tropical tree crops. In pastures it forms dense thickets which shade out and encroach upon desirable pasture plants (Swarbrick, 1997). Access problems are also often experienced which affects recreation and forest harvesting activities, cattle can be poisoned when *L. camara* is ingested, and exclusion of understorey species and changes in wildlife composition through the provision of perch sites and cover (Mathew and Brand, 2004). In Tanzania, *L. camara* can be considered as a serious health threat, as its thickets provide breeding grounds for tsetse flies infected with trypanosomes of domestic animals (Binggeli, 1998). On the other side, *L. camara* has many beneficial uses: it is grown as a hedge plant; its stalks are used for paper pulp; its bark is used as an astringent; and its leaves are used for many medicinal purposes. Humans are responsible for its introduction from native range to other parts of the world. Once introduced, *L. camara* is efficiently spread by a number of birds and sometimes sheep and goats disperse their seeds. Due to its high prolific seed production, the shrubs can establish in disturbed areas just in few years after its introduction (Weber, 2003).

2.2.3 *Clidemia hirta*

Clidemia hirta is a toxic weedy shrub from Central America that has been introduced to almost all tropical islands and Southeast Asia. This noxious weedy shrub grows up to two metres tall in pastures and forest and can shade out all other vegetation (Wester and Wood, 1977). *Clidemia hirta* flowers and fruits prolifically throughout the year, producing sweet, pulpy, dark blue berries filled with minute seeds. A mature plant can produce over 500 berries (6-9 mm long) per year each containing over 100 seeds (0.5-0.75 mm long). Seeds form a very large seed bank where they remain viable for up to four years (Binggeli, 1998). The seeds are principally dispersed by frugivorous birds, but any organism moving through the thickets will carry seeds away with it. Long distance dispersal is carried out by humans. On a good soil, this weed can produce an impenetrable stand. It thrives both in open grassland and in deep shade, and can result in greatly increased weeding costs in commercial plantations. *Clidemia hirta* is a pioneering species after disturbance; in forests it tends to displace native plants. It is probably not resistant to fire; an unlikely event in its habitat, but it rapidly colonizes burned areas (PIER, 2006).

2.3 Alien Tree Species in Tanzania

Numerous plant species have been introduced to Tanzania during the main four phases of the history, these are; early exploration and slave trade, early colonial period, colonial exploitation, and post colonial development. However, the main purpose of their introduction is poorly known. Reasons that are given for the introduction of exotic plants are agricultural, horticultural and forestry based (Binggeli, 1998). In Tanzania, non-indigenous plants were introduced to the experimental gardens of the Bagamoyo Mission shortly after its inception in 1869. From 1893, 273 species of mostly tropical plants, including many ornamental and forest trees, such as *Acacia* spp., *Cinchona* spp.. *Delonix regia*. *Eucalyptus* spp. and *Terminalia catappa*, were tested in an experimental nursery at Dar es Salaam which functioned as a seed production and distribution centre. Large quantities of seeds of various timber species were distributed to private planters and government stations. By 1903 a total of 118 indigenous and exotic timber species were under cultivation in the Lushoto District (Schabel, 1990). Following the closure of the Kwai Agricultural Station, founded in 1896, a station was officially established in the East Usambaras at Amani in 1902 as an experimental site for field testing of exotic species (Iversen, 1991a). Species that were introduced from other continents were either potential cash crops or known to be fast growing and therefore useful in reforestation of steep mountain sites. Amani became a major distribution centre for seeds and seedlings in Tanzania and the tropics (Schabel, 1990). During the 1980s it was realised that a number of introduced trees and shrubs such as *Maesopsis eminii*, *Cedrela odorata*, *Lantana camara*, *Clidemia hirta* and others, were spreading in the logged and natural forests of the East Usambaras thus threaten its biodiversity (Binggeli *et al.*, 1989).

2.3.1 *Maesopsis eminii*

Maesopsis eminii is a fast growing, gregarious pioneer tree species, which was initially introduced to the East Usambaras around 1913 (Binggeli, 1998). The origin of the seed is unknown, but the species occurs naturally in low and moderate elevations in East and equatorial Africa (Hall, 1995). Large-scale spread started in the 1970s, following industrial logging and subsequent planting of *Maesopsis* to restock the logged sites, and its use as a nurse tree for the main timber species, *Cephalosphaera usambarensis*. The two species were also planted in mixtures from the late 1960s (Mugasha, 1981). These plantations provided a massive seed source of *Maesopsis*, while logging and private pit-sawing created suitable sites for colonization. Commercial harvesting was banned in 1986, and pit-sawing has also been effectively regulated in the area. The establishment of *Maesopsis* plantations was discontinued in 1981 (Hall, 1995).

Conflicting reports about the threat of *M. eminii* to indigenous forests have been presented. For instance, according to Binggeli (1989), *M. eminii* would have the potential to invade up to 50% of the natural forest area in the East Usambaras within the next 200 years. On the other hand, Hall (1995) concluded that the spreading rate of *M. eminii* had significantly declined after the discontinuation of logging in the East Usambaras in 1986. Geddes (1998) suggested that the regeneration of *M. eminii* is more a reflection of past human intervention than a measure of its 'invasive' nature.

Different strategies have been proposed to manage *M. eminii* in the East Usambaras. Binggeli (1989) and Binggeli and Hamilton (1993) recommended the harvesting of the *M. eminii* plantations and the replacement of this species by native species. Hall (1995) proposed selective removal of mature trees on public and estate land as well as forest reserves. Geddes (1998), in contrast, associated the spreading of *M. eminii* with logging and other disturbance in the past, and did not foresee a need for intervention. Invasion by *Maesopsis* in East Usambaras is greatly facilitated by the presence of the huge bird *Ceratogymna brevis* (hornbill), which is capable of travelling over a long distance. Therefore, make it possible for *Maesopsis* to occur in large number of isolated forest fragments separated from the continuous forest block by tea plantation (Cordeiro *et al.*, 2004). Considering the present low level of man-made disturbance in the East Usambaras, large-scale elimination of *M. eminii* is not feasible. Removal of the species by clear-cutting would result in increased soil erosion and renewed spread (Hall, 1995). Dense populations of ephemeral seedlings combined with prolific seed production from adjacent stands would inevitably make it possible for the species to retain such sites. From a management perspective, it is very difficult to contain the spread of *M. eminii* in the non-protected forest areas where agriculture and forest clearing provide continuous sites for colonization. At present the most appropriate management strategy is to control disturbance in the conservation area.

2.3.2 *Cedrela odorata*

Cedrela odorata is a large tree up to 40 m tall and 2 m in diameter which produces a light-weight timber. Its natural distribution range is confined to the New World, extending from northern Mexico to Argentina, including the Caribbean. It is widely planted throughout the tropics and its timber is well known for its use in cigar boxes and a broad range of other products, including musical instruments. It is also occasionally planted for shade and used as an ornamental tree on roadsides and in parks. *Cedrela odorata* has great potential as a plantation species, due to its fast growing and timber producing characteristics. It is also used as an agroforestry species in cocoa and coffee plantations (Lemmens *et al.,* 1995). However *C. odorata* is considered as a potentially invasive alien plant in Amani Nature Reserve. About 40 years after its introduction in ABG, Greenway (1934) reported heavy regeneration of *Cedrela* seedlings close to parent trees which hindered regeneration of native trees. In addition. *C. odorata* has been classified as an invasive plant in some other parts of the world (Lemmens, 1995). Apart from seeds dispersed from the original parent trees in the ABG. there are other scattered sources of seeds from *Cedrela* trees that were planted in the reserve boundaries to make the demarcation noticeable. Similar situations have been reported in some other parts of Tanzania, including the Kimboza Forest Reserve in Morogoro Region (Madoffe, 1993), where *Cedrela* is also invading and displacing native tree species.

2.4 Impacts of Biological Invasions

Biological invasions by non-indigenous species are widely recognised as a significant component of human-caused global environmental change, often resulting in a significant loss in the economic value, biological diversity and function of invaded ecosystems (Hulme, 2003). Species invasions also affect economics and public health of many nations as a result of change in global ecosystem processes. These socio-economic effects of invaders stem from both intentionally and unintentionally introduced species (FAO, 2005). Although only a small fraction of introduced species establish in their new environments, each one that does has some ecological impact, and a small proportion have dramatic impacts that must be addressed in conservation, economic, aesthetic, and ethic realms (Groom *et al.*, 2006). It should also be known that, effects of a given invasive species differ in different locations and through time, as a result of differences in invader densities and behaviour, differences in the invaded community, and the interaction of both. In Tanzania, conifer trees such as *Pinus* are not as a serious threat to biodiversity as in other part of the world for example Malawi (Chilima, 2005). Many plantations that are established in highlands of Tanzania are of *Pinus* species and their spread to the adjacent sites is obstructed by either dense natural vegetation or farmlands which are continuously under cultivation. Therefore, *Pinus* species are not considered as a threat to these areas. The *Pinacea* has the highest proportion of invasive species than any other angiosperm family comprising mainly trees and shrubs (Richardson and Rejma'nek, 2004). The invasion of *Pinus* species in other areas of the world have caused many ecosystem level changes by altering factors such as biomass distribution, plant density and vegetation height, litterfall and decomposition rates (FAO, 2005).

2.4.1 Ecological impacts of alien invasive plants

In many cases, invasive species form very dense populations that affect the population dynamics and persistence probability of native species, which may reduce the local diversity of native species. Most obvious, perhaps, are impacts on the abundance of particular plant populations (Mack *et al.*, 2000). Looking more closely at populations, we may also detect changes in the behaviour or genetic makeup of individuals; stepping back, we may see changes in community-scale diversity and species interactions, and in ecosystem characteristics such as habitat structure and nutrient cycling. Introduced species are implicated in the endangerment and extinction of species around the world (Cronk and Fuller, 1995). This population-scale impact is relatively easy to document, and may extend to major changes in community structure, measured as changes in trophic dynamics or interaction strengths between species. Also ecological processes may change as invading species have established and spread. The changes may be minimal and the plant invader may simply increase species richness (Mack *et al.*, 2000; Groom *et al.*, 2006). In contrast, where ecological processes are sufficiently disrupted, native species can be displaced, increasing plant community vulnerability to further invasion and regeneration of the invasive plant. When perturbation of ecosystem exceeds ecological threshold, ecosystem change can be so profound that controlling the invader may not restore the ecosystem to a desired condition (Masters and Sheley, 2003).

2.4.2 Social-economical impacts of alien invasive species

Biological invasions cause two main categories of economic impacts. First is the loss of potential economic output; that is losses in forest and crop production. Second is the direct cost of combating invasions, including all forms of quarantine, control, and eradication (FAO, 2005). Therefore, the threats biological invasions pose to biodiversity and ecosystem-level processes are translated directly into economic consequences (losses in crops, forestry and grazing capacity). Economical impacts also result from changes in different ecological processes, which bring about alteration in ecological services that are provided by an invaded ecosystem. However, the more direct economical impacts of invasion are those such as diseases, loss of harvested volume in forest and loss of arable land for agriculture due to invasion (Groom *et al.*, 2006). Although there are ample anecdotal examples of local and even regional costs of invaders, we consistently lack clear, comprehensive information on these costs at national and especially global levels. In the U.S., cost associated with impacts and control of the Eurasian zebra mussel (*Dreissena polymorpha*) are estimated at US$1 billion over the first 10 years of its invasion, the European gypsy moth (*Lymantria dispar*) at US$11 million per year, European ship worms (*Teredo navalis*) at US$200 million per year, and purple loosestrife (*Lythrum salicaria*) at US$45 million per year and these are only a few of the hundreds of unwanted species introduced worldwide (Pimentel *et al.*, 2000).

The other most direct economic impact of alien invasive species on the forest sector is related to the loss or reduced efficiency of production. Approximately US$4.2 billion in forest products are lost each year to alien insect pests and pathogens in the

United States (Pimentel *et al.*, 2000). In Canada, the damage resulting from past introduction of harmful invasive plant pest on agricultural crops and forestry has been estimated at CAD$7.5 billion annually and in the Canadian province of Manitoba alone, economic losses due to Dutch elm disease (*Ophiostoma ulmi sensu lato*) have been estimated at approximately CAD$30 million. The detection of Asia longhorned beetle (*Anpplophora glabripennis*) in Canada poses a significant threats to both the hardwood products industry and the maple syrup industry, whose products were valued in 1997 at CAD$480 million and CAD$130 million respectively (Environment Canada, 2004 in FAO, 2005). Such infestation of alien invasive species directly affect the quantities of forest product demanded or supplied thereby impacting global prices and markets (FAO, 2001a). Although quantitative estimates of the economic impacts are not readily available for other countries, alien invasive species no doubt significantly impact productivity.

In addition, many public health threats are caused by introduced diseases. Smallpox, avian and human malaria, and the red tides caused by toxic phytoplankton can outbreak in new areas and have devastating effects, particularly on immunologically naive populations (FAO, 2005). The global pandemic of HIV/AIDS has spread around the world and has claimed approximately 18 million victims to date, with 5 million new infections annually. The seventh global cholera pandemic, which began in 1961 in Indonesia and which now persists in Latin America, Africa, and Asia, claims an annual global death toll of tens of thousands of victims (Groom *et al.*, 2006). With today's rapid global transport of humans, crops, and livestock, the potential for disease spread is increasing. This is an immediate conservation concern in terms of species losses, and a large-scale concern in terms of the human health and economic stability that are essential ingredients for effective conservation.

2.5 Disturbance: A Factor that Increases Invasibility of Tropical Ecosystems

Tropical forests are dynamic systems, which to a large extent are regulated by disturbance. Disturbance can be caused by various external agents such as hurricanes, wind, fire and landslides, varying in intensity from region to region. In most cases disturbance is caused by treefalls, either due to uprooting or stem breakage, and as a consequence gaps are produced in the forest canopy (Hamilton, 1989). Present population structures and patterns of genetic diversity in forest trees have been influenced by past anthropogenic activities. Very little 'natural' forest (defined as areas that have not experienced a break in forest continuity because of cultural activities since conditions became suitable for tree growth) remains in the tropics (Lonsdale, 1999).

Invasion by alien plants had rarely been considered as a significant threat to the diversity of tropical forests and while invasions have been the subject of intensive ecological research during the last two decades, this research has largely ignored tropical forests (Drake *et al.*, 1989; Williamson, 1996). One reason for this neglect of tropical forests is the perception that their high species diversity makes them naturally resistant to invasion. However, anthropogenic disturbance that created openings in tropical forest habitats made them susceptible to invasion by exotic species (Davis *et al.*, 2003). For example, in tropical forest-reserves, trails were kept open by removing bordering vegetation to allow eco-tourists and others to experience tropical forest habitats in a convenient way, which resulted in the formation of gaps of contrasting sizes along those trails. Exotic populations established in these gaps served as starting points and from where seeds dispersed into pristine forest interiors or naturally disturbed patches within the forest (Davis *et al.*, 2003).

2.6 Disturbance: A Factor that has Changed Amani Ecosystem

Forest disturbance is recognized to have played a great role in facilitating invasions in forest ecosystem of East Usambara Mountains. There is evidence of human disturbance in the East Usambara forest dating back to the early Iron Age (Hamilton, 1989). Consequently it is believed that few areas of forest have escaped disturbance by man at one time or another during the past 2000 years. In recent past, since 1954, forest utilisation and clearance has led to a 70% decline in forest cover (Iversen, 1991a). What was formerly a continuous forest block is now a fragmented network of forest reserves covering an area of 33 500 ha (Johannson *et al.*, 1997). The major threats to the forest ecosystem include agricultural encroachment, cash crop cultivation (tea, coffee, sisal and cardamom), pit-sawing and logging (Kilahama, 1998).

Disturbance has facilitated invasions by eliminating or reducing the cover or vigour of competitors or by increasing resource levels such as light, nutrients and space (Binggeli and Hamilton, 1993). In such cases, the increase in invasibility following disturbance can be explained by the theory of fluctuating resource availability (Lonsdale, 1999). Whether the disturbance introduces additional resources into the community or whether there is a decline in resource uptake by the resident vegetation due to mortality or debilitation of the resident species, resource availability will increase, and thus, according to the theory, invasibility should increase (Lonsdale, 1999). Disturbances need not be community-wide to increase invasibility. Frequent small-scale disturbances, e.g. by burrowing animals, tree falls, can create localized patches of unexploited resources, and thereby may facilitate invasions.

The six forest reserves (Amani Sigi, Amani East, Amani West, Kwamsambia, Kwamkoro and Mnyuzi) that form the Amani Nature Reserve had been under constant disturbances for more than 50 years before logging was banned in 1987 (Frontier, 2001). Two forest reserves out of these are said to have experienced mild disturbance during that time due to their inaccessibility and terrain, which could not be easily logged, these are Amani Sigi and Kwamsambia. During the Sikh Saw Mill operation in 1980s, the average logged area was one hectare of forest per day. Therefore by the time logging was banned, many of the areas in Amani had already been highly disturbed (Hamilton, 1989). In spite logging being banned in ANR, small scale illegal harvesting is still a problem. This has also created localized gaps, which facilitate invasion by alien plants inside the forest (Killenga, 2007). With the presence of the botanical garden in Amani Nature Reserve, any disturbance may pose threats by opening gaps for invasive plants to establish and increase in their population. Invasive tree species that are commonly found inside the forest due to gap creation through logging activities includes *Maesopsis eminii*, *Lantana camara*, and *Clidemia hirta* (Fowler and Nyambo, 1996).

2.7 Study *Species - Castilla elastica*

2.7.1 Biology, ecology, distribution and spread

Castilla elastica Cerv., commonly known as the Mexican or Panama rubber tree belongs to the family Moraceae. *Castilla* is a deciduous androdioecious plant; i.e. there are cosexes and male plants within a population. A cosex plant will be referred to as "a mother tree" throughout this document. *Castilla elastica* grows to a height of 10-30 m tall depending on soils, climatic conditions and genetic of the tree (Sakai. 2001). It has spreading or drooping branches, which do not develop second-order branches, the young ones being woolly-hairy. The leaves are coarse, densely hairy, short-stemmed, arranged in two rows, the blade oblong, broadest in the upper half, with about 18 pairs of prominent veins. Inconspicuous female flowers in short-stemmed heads at leaf axils develop into fruit about 4 cm in diameter (Neal, 1965).

The natural distribution of the species ranges from Mexico through Panama and the coastal region of western Colombia and western Ecuador. *Castilla* is reported to be very invasive in Samoa, where it has claimed large area of secondary forests and has become established in rainforests thus posing threats to intact native forests and their biodiversity. It has also naturalised on the Western end of Tutuila, American Samoa. near Maloata (Space and Flynn, 2002) and was introduced in Tanzania for rubber production during 1890s by Germans (Engler, cited in Markus, 2006).

Castilla elastica is an early successional tree in the rainforests; it prefers areas of moist forests at low elevation (Hammel, 1986). In its native range, *C. elastica* is an abundant tree species in secondary forests and open areas such as clearings and forest edges. In these areas, *Castilla* has been used for hanging beehives, arts and crafts, industrial, medicinal and ornamental purposes. It was widely grown as an income-generating tree among the early subsistence farmers in Costa Rica where it is native. Despite being indigenous to Costa Rica, it tends to be invasive, whenever there is a gap in the canopy; it is one of the first plants to establish (Swarbrick, 1997; Wayne *et al.*, 2004).

2.7.2 Population structure and status in plant communities

Population status of a species can be determined based on assessment of the population structure in terms of size class distribution of the species. Knowledge on the structure of the population is of considerable importance in the management of any species or forest in general. A stable, self-maintaining population is characterized where there is a smooth decrease in the number of individuals from the smaller to large size classes, with the intermediate class being well represented (Crawley, 1986). A common characteristic of size distribution of most tropical trees in a population is the pronounced absence of saplings and juveniles. This type of size class distribution results when regeneration of a species is severely limited for several reasons, with most seedlings dying before becoming established (Whitmore, 1975). For example, large *Ocotea usambarensis* trees which are not regenerating have been noted in the East Usambara Mountains (Hamilton & Macfadyen, 1989), and also the species has been reported to have an unbalanced population structure at Mazumbai in the West Usambara (Hall, 1990). This may be due to the fact that *Ocotea usambarensis* does not produce well by seeds, and therefore its reproduction in the area is mainly through suckers (Willan, 1965). Natural regeneration is important in understanding the forest dynamics. It is the process that shows the dynamic of the forest and determines the fate of any particular species in the future.

Forest trees ensure their perpetuity through regeneration. The regeneration may be through seedlings or through vegetative propagation by coppicing or root suckers. Therefore all types of regeneration depend on the presence of the mother tree at that particular time or previous years (Busing, 1994; Ganzhorn, 1995). In dioecious and some androdioecious (co-sexes and males in a single population) plants, the situation is even more complex, because there should be both male and female plants in a population to ensure their perpetuity. However, this is not the case with the androdioecious plant *Castilla elastica*, because the cosexual (mother tree) of *Castilla* is capable of producing viable seeds in absence of male tree. The stamens in both the co-sex and male tree are fertile; therefore the co-sex tree is capable of self pollination and self fertilization (Sakai, 2001). Seed production through self-pollination must be essential when *Castilla* invades a new habitat without conspecific trees.

Plant performance can be measured by various traits ranging from competitive ability to fecundity. The higher the quantity of seeds produced, the higher is the chance of regeneration and thus its competitive ability. PIER (2006) reports *Castilla elastica* as a heavy seeder plants, which has an average of 2035 seeds per kg with a total germination of 86% (Sautu *et al.*, 2006). In any case, whether or not alien invasive plants that substantially outperform natives, have important consequences for conservation. An invader that outperforms co-occurring natives is expected to increase in relative abundance over time through natural regeneration, and as a result poses significant impacts on native populations (Daehler and Carino, 1999).

2.8 Management of Alien Invasive Species

Two sets of complementary strategies for dealing with alien invasive species are: prevention and early detection; and response, which include eradication (rarely achieved), containment, control and mitigation (FAO, 2003). Prevention is the first line of defence against biological invasion and is also the most cost effective since once an alien invasive species becomes established, it is extremely difficult and hence costly to eradicate. An important first step in prevention is identification of the species capable of becoming invasive, the possible susceptible sites and more importantly, the pathways in which they can be introduced (Wittenberg and Cock, 2001). Early detection of alien species should be based on a system of regular surveys. This can be general, site-specific or species-specific, in order to identify newly established species. Although not all alien species become invasive, the costs of those that do become invasive suggest that a precautionary approach to the issues is best. The longer species go undetected the fewer the option for its control or eradication and more expensive any intervention will become (FAO, 2005).

When the preventative and early detection measures have failed to stop the introduction of alien invasive species, eradication is the preferred next method of action. Eradication involves the elimination of the entire population of an alien species, including resting stages, in managed areas (Parker, 2004). As a rapid response to early detection of an alien species, eradication is often the key to a successful and cost-effective solution. Careful analysis of the costs involved and the likelihood of success must be made before any eradication attempts are made so as to ensure the ecosystems will be restored after eradication (Groom *et al.*, 2006). Containment is a special form of control aimed at restricting the spread of an alien invasive species and to contain the population in defined geographical range. In cases where eradication is not possible, containment of the alien invasive species into a defined area can be very effective in saving other regions of a country (Wittenberg and Cock, 2001). The long-term reduction in density and abundance of an alien invasive species to below a pre-set acceptable threshold is the aim of control programmes. Suppression of invasive population below such threshold can favour native species. Control methods that have been used successfully in controlling alien invasive plants include; mechanical control, chemical control, biological control, habitat management, and a combination of these.

Biological control of weeds through the importation of natural enemies from the area of origin has a long history and a good success rate (McFadyen, 1998). Some attempts have been made against invasive forestry trees, mostly in Australia, Southeast Asia and South Africa. A long-running programme of biological control of *Mimosa pigra* in Australia and Southeast Asia has been undertaken by CSIRO (Commonwealth Scientific and Industrial Research Organization) along with other Australian partners and the government of Thailand (Lonsdale *et al.*, 1995). Similarly, efforts are being made in South Africa to control *Acacia saligna* through the use of a gall-forming rust fungus, *Uromycladium tepperianum*, which kills saplings and some older trees and reduces seed production in others (FAO, 2003).

Host-specific bruchids have now been successfully established for the control of *Acacia melanoxylon* and various *Prosopis* species, and the same approach is being used for an ongoing programme to control black wattle (*Acacia mearnsii*) in South Africa and *Prosopis juliflora* in Ascension Island (Adair, 2002). Another bruchid species has also been introduced into South Africa for the control of *Leucaena*. This species was also accidentally released in Australia, where its impact is currently being assessed (Hughes, 1998).

CHAPTER THREE

3.0 MATERIALS AND METHODS

3.1 Study Area

3.1.1 Location

Amani Nature Reserve is situated in the southern part of the East Usambara Mountains approximately 55 km by road from Tanga town. The Reserve lies between 5°14'10'' - 5°04'30'' S and 38°30'34'' - 38°40'06'' E., with an altitudinal range of 190 to 1130 metres above sea level (asl) (Frontier, 2001). This study was conducted at ABG and Amani Sigi Forest Reserve, both of which are within the ANR (Fig. 1). The two areas are separated from each other by the Sigi River and lie in the northeast corner of ANR. The study site has an area of approximately 125 ha, and the altitude ranges from 380 m up to just over 600 m asl. The area was chosen due to the presence of original plots of *Castilla elastica* trees that were planted during the establishment of the botanical garden (Greenway, 1934) and the suitable climatic conditions for invasion.

3.1.2 Climate, soil and topography

The climate of the East Usambaras is monsoonal, with two rain seasons namely the long rains from March to May and the short rains from October to December. The mean annual precipitation is highest in the southern parts of the plateau, reaching 2200 mm, and decreasing gradually to the north and the lowlands. The mean annual rainfall at ANR is 1900 mm (Binggeli *et al.*, 1989). Mean monthly temperatures in the East

Usambaras vary considerably during the year with a difference of 5^0 C between the hottest and the coldest months, i.e. March and July, respectively. The mean annual temperatures are 20.4^0 C and 20.6^0 C at Kwamkoro and Amani stations, respectively. The soils are deep, reddish clay-loams, with nutrients concentrated in topsoil less than 20 cm deep. Forest topsoils above 850-900 m are markedly acidic (pH 3.5-5.5) and exceptionally strongly leached compared with lowland soils (pH 6-7), which are less leached and more fertile (Hamilton and Mwasha, 1989b).

3.1.3 Vegetation

The two main forest types in the East Usambaras are the lowland rain forest and the submontane rain forest, which occur below and above 880 m asl, respectively. Due to variation in rainfall, the lowland forest is semi-deciduous, whereas the submontane forest on the plateau of ANR is evergreen. However, despite the distinction between the two forest types, floristic variation is continuous with altitude (Hamilton and Mwasha, 1989). As in the case for other parts of East Usambara Mountains, the ANR contains a large number of endemic or near-endemic tree species (Frontier, 2001). Most frequent indigenous tree species in the study area were *Synsepalum msolo*, *Trilepsium madagascariense*, *Leptonychia usambarensis*, *Funtumia africana*, *Antiaris toxicaria*, *Ficus* spp., *Milicia excelsa*, *Ricinodendron heudelotii*, *Tabernaemontana* spp., and *Macaranga capensis* (Binggeli and Hamilton, 1989). In addition, more than 600 tropical and sub-tropical alien plant species were introduced to the study area as part of the ABG. The botanical garden covers an area of approximately 300 ha, of which about a third remains under the original forest, while a large part of the remainder is occupied by permanent plantations (Schulman *et al.*, 1998).

3.2 Study Procedures

3.2.1 Reconnaissance survey

A reconnaissance visit was conducted to provide a general picture of the distribution of *Castilla elastica* in ANR and to locate definite plots in the ABG that were originally planted with the study species. Maps from Greenway (1934) and Honess (1963) were used to locate the original plots of *C. elastica*. The reconnaissance survey revealed that the distribution of *C. elastica* in the Reserve falls within the altitudinal range of 380 m to just over 600 m asl. It was also found that, out of 8 plots that were originally planted with *C. elastica* and had a total of about 198 trees in 1930s, only 3 plots were found with *Castilla* trees. Though very few trees remain in these original plots, some of these old trees are undoubtedly the original trees that were planted in early 1900s.

The visit also revealed that *C. elastica* has escaped from the botanical garden and has naturalised in the lower part of Amani Sigi Forest Reserve, adjacent to and separated from the botanical garden by the Sigi River. This area was therefore purposefully selected for studying the influence of forest edge (due to presence of roads and Sigi River) on the relative abundance of *C. elastica*. Also, the decision was made to categorise the study area into four disturbance levels. Areas that were formerly used as a school farm and timber yard by the Sikh Saw Mills were categorised as forest of high disturbance because they almost resemble secondary forest with only few mature indigenous trees remaining.

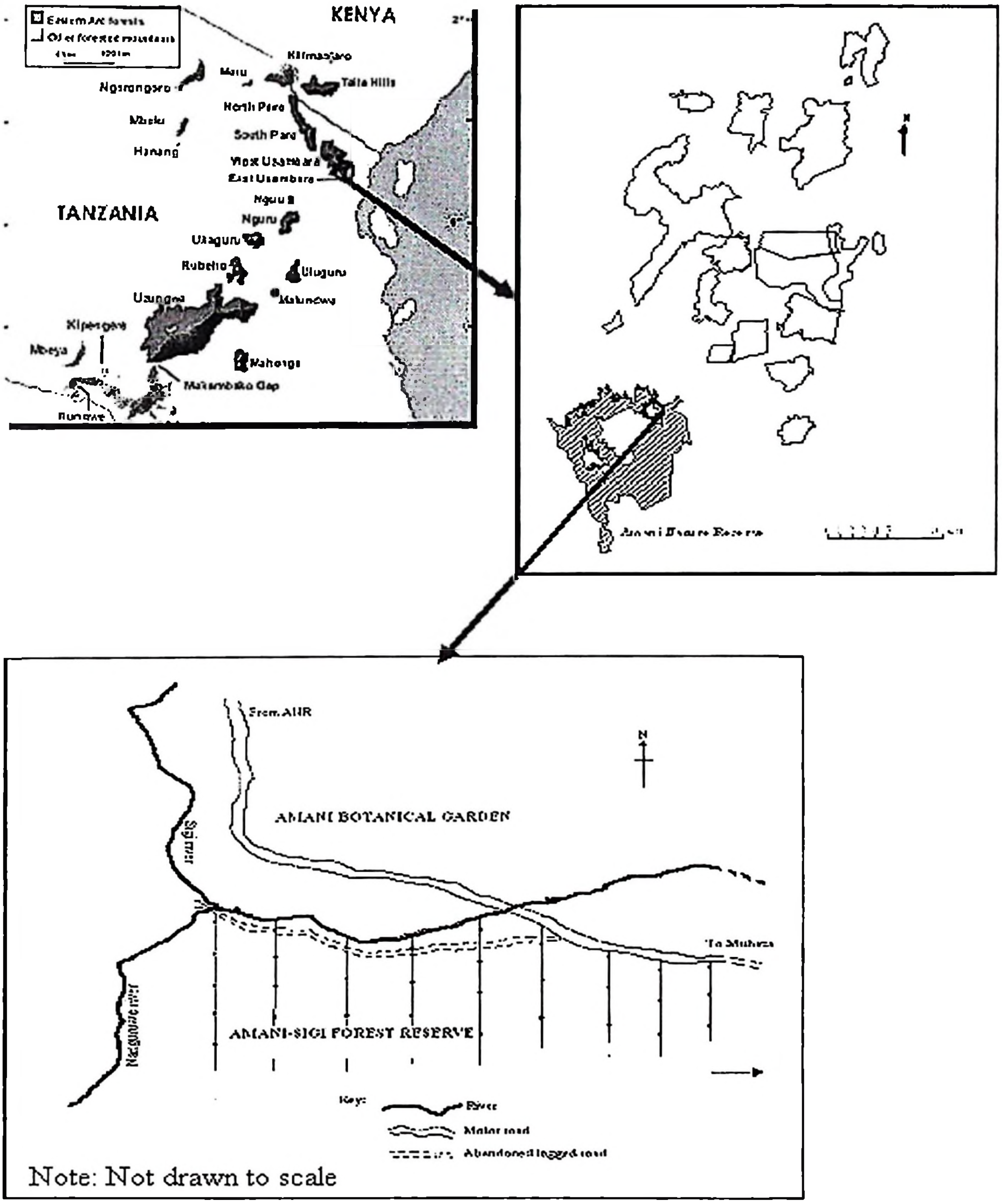


Figure 1: A sketch map of the study area zoomed from the East Usambara Mountains, Tanzania

Areas that were selectively logged by the Sikh Saw Mills were categorised as forest of medium disturbance whereas the other remaining part of the study area where disturbance was minimal was categorised as forest of low disturbance. The area in the botanical garden was categorised as plantation forest, due to the presence of plantation plots of different tree species. In addition, plant species such as *Ficus exasperata* that are indicators of disturbance were also used to categorise the disturbance levels. During this survey, *Castilla* mother trees were selected for population structure studies and the total area naturalised with *C. elastica* was also estimated for determination of the number of sampling units.

3.2.2 Methods used for data collection

3.2.2.1 Point centred quarter method

The point-centred quarter method was used for the determination of the relative abundance of the constituent species of the canopy in the study area. The point-centred quarter technique is a plotless sampling method designed for quantitative description of the structure and composition of forest canopies (Cottam *et al.*, 1953). Therefore. the method provides estimates both of total tree density and of the total basal area per unit area of ground, as well as frequency, density and basal area for each of the constituent species of the canopy (Mitchell, 2001; Hulme, 2006). The ground around each point was "quartered', using the four cardinal compass directions, giving four segments each of 90^0. Within each segment (quarter) the nearest tree to the sampling point or centre was located and its identity (I). distance (d) from the centre in meter and girth (G) at the breast height (1.3m from the ground) in centimetres were recorded (Fig. 2). Trees that were measured in these sampling points had a minimum diameter at breast height (dbh) of 10 cm.

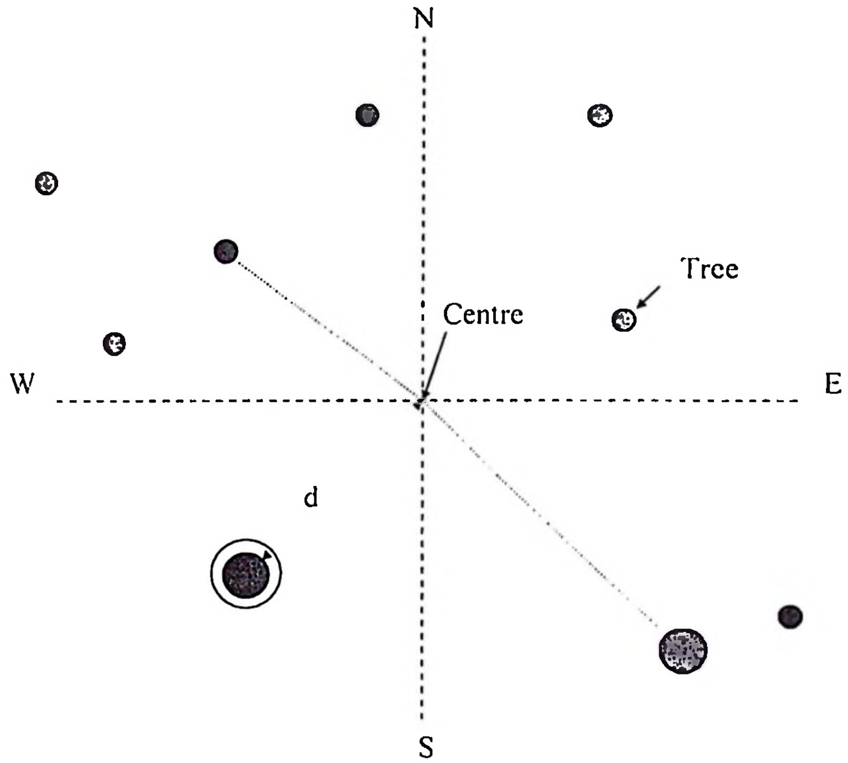


Figure 2: Illustration of field method with the point centred quarter

The point centred quarter sampling points can be laid out randomly or systematically. For the determination of the relative density of tree species in the invaded area, sampling points were laid out systematically at intervals of 50 m along transects. Transects were initiated at the junction of Nanguruwe River and Sigi River, running in North-South direction with intervals of 100 m between them to cover the area that is naturalised by

Castilla. A total of 16 transects were laid out in this area, and each transect had 6 sampling points. The first of each sampling point was laid at 20m from the forest edge to ensure that trees could be sampled in all quarters. In addition, from each of the sampling point, another set of data of the nearest *Castilla* tree in each quadrant was taken for ascertaining the size distribution based on their diameter at breast height.

To determine the absolute density of *C. elastica* in the four areas of forest with different disturbance levels, five sampling points were selected randomly from each level. In this case, only the nearest *Castilla* tree to the sampling point with minimum dbh of 10 cm was considered for measurements when using the point centred quarter method. The principal consideration for the random selection of the sampling points was that they should be far enough from each other to ensure that no *Castilla* tree is counted twice by the sampling procedure.

3.2.2.2 Population structure

Sampling methods for the assessment of population structure of *C. elastica* closely followed those of Sawe (1997), when assessing the regeneration status of indigenous tree species in the East Usambara Mountains. In this study, five *Castilla* mother trees were randomly but objectively selected from each of the four forest categories. The minimum dbh of the selected tree was 40 cm and it was at least 50 m away from other *Castilla* mother trees of the same size. The aim of this separation was to minimise the regeneration recorded from seeds of individuals other than the focal mother tree. From each selected mother tree four transects of 25 m were made in the cardinal directions.

Plots of 2 m × 2 m were established at 4 m intervals along transects (Fig. 3), such that each transect had six plots, with a total of 24 plots per mother tree. For each disturbance level, 120 plots were established making a total of 480 plots for all four-disturbance levels. The assumption here was that, other *Castilla* trees within the 25 m radius were from regenerants of the selected mother tree. However, other mature trees that were capable of producing fruits within the 25 m radius were counted and recorded for the determination of the ratio of mother trees to regenerants.

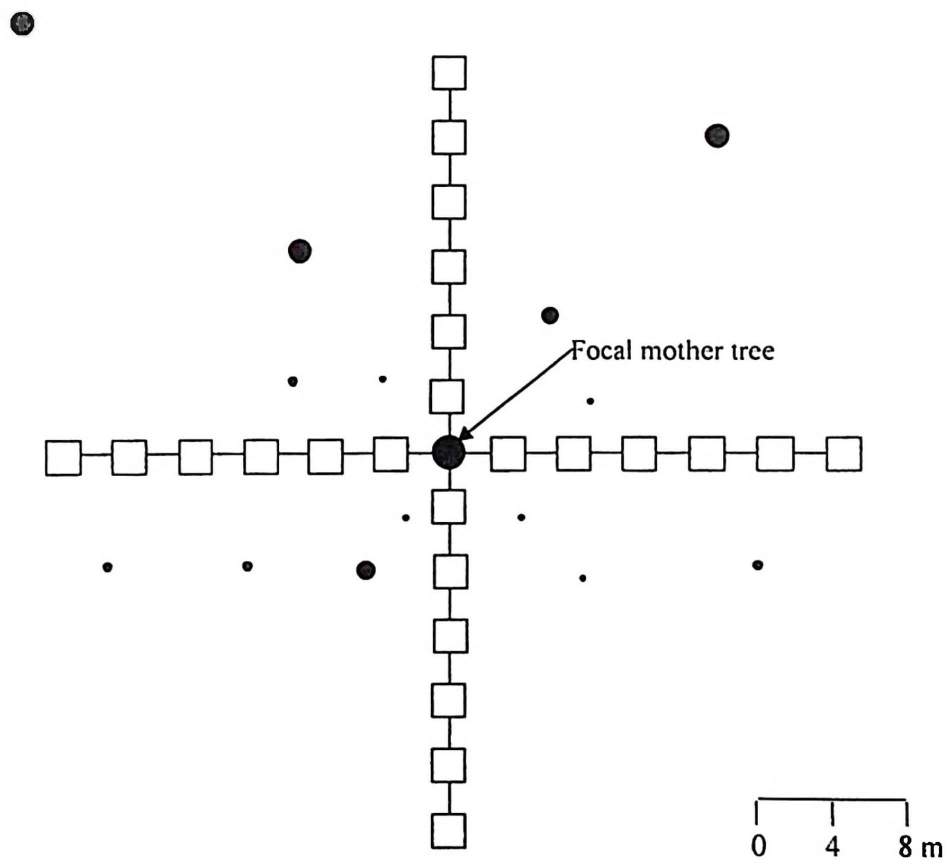


Figure 3: Field layout of population structure study; 6 plots of 2 × 2 m radiating from the *Castilla* mother tree at the centre

In each plot of 2m × 2m. the regenerants of *Castilla* of different age classes were counted and their dbh measured and recorded as follows: -

Seedlings: dbh < 1cm

Saplings: dbh ≥1cm but with < 5 cm

Juveniles: dbh ≥ 5cm but with < 10 cm

Young trees: dbh ≥ 10 cm but with < 20cm

Mature trees: dbh ≥ 20cm

In addition, mature males and cosexual trees within the 25 m radius of the mother tree were counted and recorded, by critically observing the appearance of flowers and fruits produced using binoculars (plate 1 and 2: male and females fruits).

3.2.2.3 Canopy measurement

Canopy cover was measured using a spherical densiometer (Lemmon, 1957). Four measurements were taken in the cardinal directions from every point centred quarter plot and from each of the selected mother trees, and the average canopy cover of the four measures was calculated. Prior to analysis, canopy cover was arcsine root transformed to acquire a normal distribution of the data. A clinometer was used to measure gradient of slope, with the average of three measurements taken at each point.

Plate 1: Female inflorescences (roundish) and complimental male inflorescence of *Castilla*

Plate 2: Primary male inflorescence of *Castilla*

3.2.2.4 Indices of Density Dominance (IDD)

This reflects dominance because it weights the proportion of a particular species in the total density without excluding the influence of the density of that particular species in the overall density. In this case, the IDD was used to determine the density of different tree species that were found in the study area. This index is calculated as follows:

Index of Density Dominance (IDD) of a species = (ni / N-ni)

Where; ni is the density of an individual tree species.

N is the total density of all other tree species

3.2.2.5 Weed Risk Assessment of *Castilla elastica* in Amani

The Weed Risk Assessment (WRA) system that was employed in this study resembles the Hawaiian WRA with only minor modifications for the use in Eastern Arc Mountains (Hulme, 2007). The WRA is calculated from answers to a maximum of 50 questions (Appendix 1 and 4); each relating in a logical or scientific (statistical) way to the risk of a plant becoming a pest. Therefore the system minimizes the role of personal opinion during the assessment process. A comprehensive search of literature was carried out in order to obtain information on the ecology, biology, distribution and control methods of *Castilla elastica* before filling in the questionnaires. Available databases from the internet such as (http://www.hear.org/pier) were also searched. Information obtained from these sources was used to answer the questionnaires.

3.3 Data Analysis

Data obtained from this study were organised and analysed using the Microsoft excel (Ms excel) and the Statistical Package for Social Sciences (SPSS) Programme, Version 12.0. Prior to regression analysis, tests for homogeneity of variances were conducted, using the Laverne statistics. If Laverne statistics revealed that there was no homogeneity in variances, then mean separation was conducted using the Least and Tamhane significant difference tests, when equal and unequal variance were assumed respectively. Pearson correlation coefficients were calculated between all variables for abundance and population studies.

3.3.1 Relative abundance and dominance of tree species

Distance from the forest edge was considered as the explanatory variable, and the relative density of each recorded tree species as the dependent variable. Linear regression analysis for each tree species was used to investigate the variation in relative density of each tree species as distance increases from the forest edge. In addition the IDD for each tree species was calculated for the determination of density dominance in the study area.

3.3.2 Frequency distribution of dbh size classes

The frequency distribution of dbh size classes was ascertained from the 280 *Castilla* trees that were recorded and measured in the 16 transects which had a total of about 96 point-centred quarters with 384 quadrants. The dbh were sorted out and arranged in the five diameter classes based on their sizes. The dbh size classes were; 10 - <20 cm; 20 - <30 cm; 30 - <40 cm; 40- <50 cm and 50 - <60 cm. Finally, a graph showing the frequency distribution of individuals according to their dbh size classes was drawn.

3.3.3 Density of *Castilla* in different disturbance levels

The explanatory variable in this case was the level of disturbance and there were five replicates from each disturbance level. One-way analysis of variance (ANOVA) was used to test for the difference between densities of *Castilla elastica* in the disturbance levels. Based on the analysis, the Tamhane significant difference test was used to separate means.

3.3.4 Population structure

To investigate the difference in regenerants density between areas with male trees and areas without male trees, *t*-test was employed. Also, linear regression analysis was used to investigate the effect of distance from mother trees on the density of different regenerants life stages.

3.3.5 Weed Risk Assessment analysis

Appendix 1 is the screening systems questionnaire/score sheet. Unless otherwise noted on the questionnaire, all questions were to be answered "Yes" or "No". The first column in Appendix 1 indicates the serial number of each questionnaire and the WRA questionnaires are shown in the second column. The third column indicates how each answer affects the score and in the fourth column is where scores for each questionnaire were filled in. The far right-hand column shows relevant references for answers against each questionnaire. All information available was used to fill out the questionnaires as completely as possible. Nevertheless. information needed to answer some questions could not be obtained from literature. In WRA it is recommended to answer at least one-third of the questions overall (http:www.hear.org/pier). Therefore species score obtained from this study could still be used to make an assessment for *Castilla elastica*, provided that several questions were answered in each general area. The total species score was obtained by summing the scores from each individual question, with the exception of questions 2.01, 2.02 and 2.05. The answers to these three questions were used to determine the appropriate scores for questions 3.01 to 3.05 by looking up a "multiplier" (ranging from 0.5 to 2) in the Appendix 2.

Based on the WRA scores obtained, species in question could be placed into the following categories (Pheloung, 2001):

- Accept (not likely to be a pest; WRA score < 1),
- Reject (likely to be a pest; WRA score > 6),
- Or evaluate (requires further evaluation; WRA score = 1-6)

CHAPTER FOUR

4.0 RESULTS AND DISCUSSION

4.1 Abundance and Dominance of Tree Species

4.1.1 Abundance with respect to distance from the forest edge

Figure 4 shows the association between distance from the forest age and relative densities of *Castilla elastica* and of two other exotic species. *Cedrela odorata* and *Maesopsis eminii*, which are the most abundant exotic trees in the study area. Subsequent to the first use of the specific epithet in the descriptions that follow, each species is referred to by its genus: *Castilla*, *Cedrela*, or *Maesopsis*; use of the generic name implies the particular species that was studied. Relative densities of *Castilla* and *Cedrela* were significantly negatively correlated with distance from the forest edge (r = -0.89, $P< 0.017$ and r = -0.83, $P< 0.036$ respectively). The relative density of *Maesopsis* had insignificant negative correlation with distance from the forest edge (r = -0.24, $P< 0.64$). At the edge of the forest, relative densities of both *Castilla* (96 stems/ha) and *Cedrela* (96 stems/ha) were approximately five times higher the density of *Maesopsis* (21 stems/ha). At 70 m from the edge of the forest, the relative density of *Castilla* was two times higher the density of *Cedrela*, which was the second most abundant.

In the interior at approximately 300 m from the forest edge, the density of *Maesopsis* (19 stems/ha) was as twice as much the density of *Castilla* (9 stems/ha), and *Cedrela* was not recorded at this distance from the edge (Appendix 5). The high relative densities of *Castilla* and *Cedrela* in the invaded area close to the forest edge could be attributed to two reasons. The first one being the proximity of the area to the original plots (sources of seeds) of the two species, *Castilla* and *Cedrela* and the second one is the altitude which favours the performance of the two species more than the *Maesopsis*.

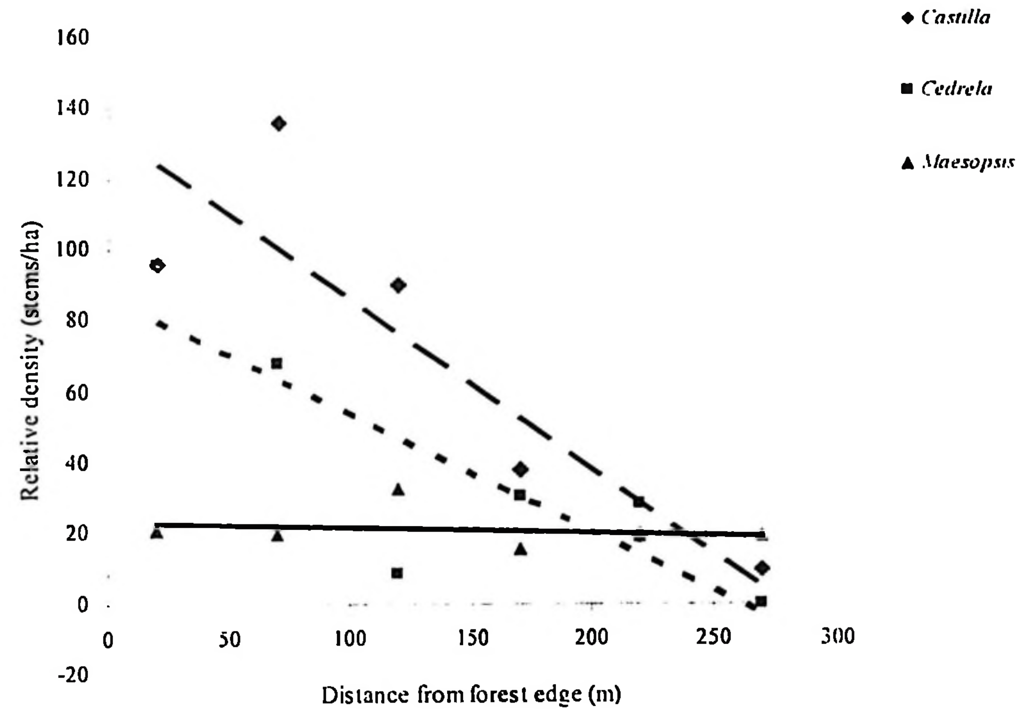


Figure 4: Relationship between relative densities of the three exotic tree species and distance from the forest edge

Linear regression coefficient of *Castilla elastica* = -0.477, $P < 0.017$, $R^2 = 0.794$; *Cedrela odorata* = -0.329, $P < 0.036$, $R^2 = 0.70$ and *Maesopsis eminii* = -0.015, $P < 0.645$, $R^2 = 0.05$.

Original plots of both *Castilla* and *Cedrela* are relatively closer to the invaded area than the original plots of *Maesopsis* (Greenway, 1934). Human-induced propagule pressure in particular is a crucial factor in plant invasion (Lonsdale, 1999; Leung *et al.*, 2004;

Chytrý *et al.*, 2005). That is why other areas with same climatical conditions do not experience invasions from known invasive species, possibly because there are no sources of propagule nearby (Lepš *et al.*, 2002) and long range dispersal is ineffective. Although *Maesopsis* seeds could be dispersed by Hornbill to this area and dominates the forest as it does in some other part of ABG, the elevation of the area (which ranges from 380 to 650 m asl) might have not been as favourable for *Maesopsis* as it is for *Castilla* and *Cedrela*. *Maesopsis* grows best at altitudes from 600 to 900 m asl; it is also found distributed in lowland tropical rainforest to savanna, extending into submontane forest up to 1800 m altitude (Binggeli, 1998). *Castilla* and *Cedrela* are low land tree species, but the later can be found distributed up to 1500 m asl (Woodson and Schery, cited in PIER, 2006), while the former is distributed up to 800 m asl. During the survey by Frontier 2001, the most commonly recorded plant species across the ANR was *Maesopsis*, however this is among the areas that were found to have relatively few stems of *Maesopsis* per hectare (in this area they recorded 5 to 24 stems/ha of dbh greater than 10 cm). The high *Maesopsis* density areas e.g. near the Kwamkoro Nature Trail (where the density > 100 stems/ha), are the areas of forest close to where the species was originally planted.

The concept of propagule pressure can also explain the reason for successive invasion of *Maesopsis* in the whole of ANR, because more than 580 ha of *Maesopsis* were planted in the Reserve near Kwamkoro (Binggeli and Hamilton, 1993), which account for a reliable and a very large source of seeds of *Maesopsis* in the reserve. The prolific seedling habit of *Maesopsis*, its efficient dispersal by Hornbills, its very high germination rate and its fast growth rates give *Maesopsis* the higher chance of invading the interior of the forest than *Castilla* and *Cedrela*. On the other side, the 96 stems/ha recorded for the study species *Castilla,* which represents more 21% of trees (both exotic and native) at the forest edge, is large proportion that poses threat to native trees which formerly composed the original vegetation of this area. *Castilla* was found to be an early successional tree in the rainforest of Costa Rica (Wayne *et al.*, 2004). Despite its being indigenous to this area, it also tends to be invasive, and whenever there is a gap in the canopy, it is one of the first plants to establish (Wayne *et al.,* 2004; PIER, 2006). With this concern of invasion from exotic plants, Binggeli and Hamilton (1993) commented that there is a need to keep such exotics out of the East Usambaras so that such invasions as of *Maesopsis* may not arise.

The distribution of the five most abundant native trees *Cephalosphaera usambarensis*, *Funtumia africana*, *Leptonychia usambarensis*, *Antiaris toxicaria* and *Milicia excelsa* in relation to distance from the forest edge is shown in Fig. 5. The relative densities of *Cephalosphaera usambarensis* and *Leptonychia usambarensis* were significantly positively correlated with distance from the forest edge ($r = 0.91$, $P< 0.01$ and $r = 0.96$, $P< 0.002$ respectively). The relative densities of *Funtumia africana*, *Antiaris toxicaria* and *Milicia excelsa* had a weak positive correlation with distance ($r = 0.4$, $P< 0.42$, $r = 0.26$, $P< 0.6$ and $r = 0.35$, $P< 0.48$ respectively). Therefore among the native tree species, *Cephalosphaera* and *Leptonychia* were considered as interior species but

Funtumia, *Antiaris* and *Milicia* were not, because they were found almost both at the edge and in the interior.

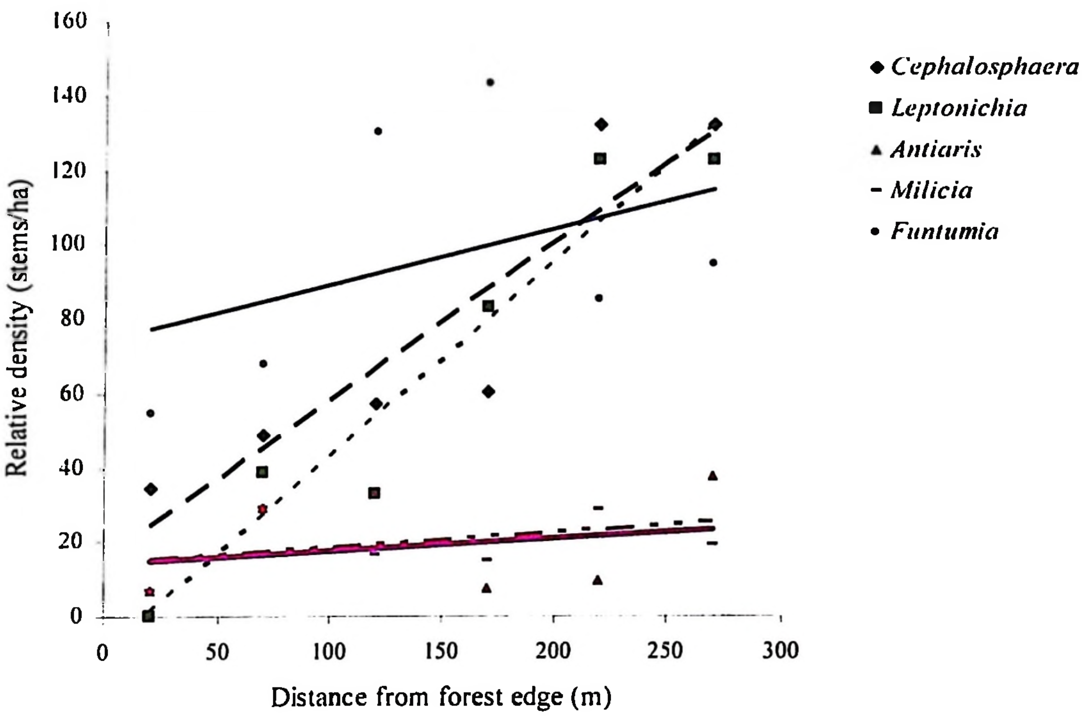


Figure 5: Relationship between relative densities of the five native tree species and distance from the forest edge

Linear regression coefficient of *Cephalosphaera usambarensis* = 0.43, $P< 0.01$, $R^2 = 0.84$; *Funtumia africana* = 0.15, $P< 0.42$, $R^2 = 0.16$; *Leptonychia usambarensis* = 0.52, $P< 0.002$; $R^2 = 0.92$, *Antiaris toxicaria* = 0.04, $P< 0.6$, $R^2 = 0.07$ and *Milicia excelsa* = 0.03, $P< 0.48$, $R^2 = 0.12$.

Antiaris is actually a light-demanding species and its seedlings and young trees are normally not found under shade in the forest (Hamilton, 1989), therefore it was not expected to be an interior species. *Cephalosphaera* is a pioneer species; it was expected to be found at the edge than in the interior. However, the extraction of this tree species may have contributed to its low abundance at the edge due to the fact that, Sikh Saw Mills extracted almost all the mature *Cephalosphaera* in this area, even the disturbance that is seen in this area is due to the extraction of *Cephalosphaera* by the Company (Hamilton, 1989). *Leptonychia* is a climax as well as a low land native tree species; therefore it was not expected to be found at the edge. This result matches with Hall (1995) results which showed high abundance of *Leptonychia* in the interior of the forest of East Usambaras.

The overall relationship between distance from the forest edge and mean densities of all exotic tree species and all native tree species in the study area is shown in Figure 6. There was a significant negative correlation between mean density of exotics and mean density of natives ($r = -0.836$, $P < 0.05$). At 20 m from the edge, the mean density of exotic tree species was almost six times greater (34 stems/ha) than the mean density of native species (6 stems/ha). As the distance from the edge increases, the difference in mean density between alien and exotic species decreases until 180 m, after which native species have on average a greater density than exotic species.

The regeneration of indigenous species in the study area after disturbance was relatively low compared to that of exotic tree species such as *Cedrela*, *Maesopsis* and *Castilla*. The colonizing characteristics of exotic species have on many occasions enabled them to compete successfully against native vegetation for resources (Sawyer, 1993). In open environments, alien species may out-compete native species by rapid aboveground growth, in particular, height growth. Rapid height growth is a well-known trait of other successful gap invaders, such as *Maesopsis* (Frontier, 2001). Many of the most successful plant invaders are species that occur in extensive disturbed habitats, especially in high-fertility environments. Consequently, many authors have argued that the most successful plant invaders are those species possessing traits that enhance success in disturbed habitats (Rejma'nek and Richardson 1996; Williamson and Fitter 1996; Leps' *et al.*, 2002). Such traits include rapid growth rate, early maturity, high fecundity and rapid dispersal (Rejma'nek, 1996). The three exotics tree species, *Castilla*, *Cedrela* and *Maesopsis* possess these traits and these make them aggressive in disturbed environments and forest gaps.

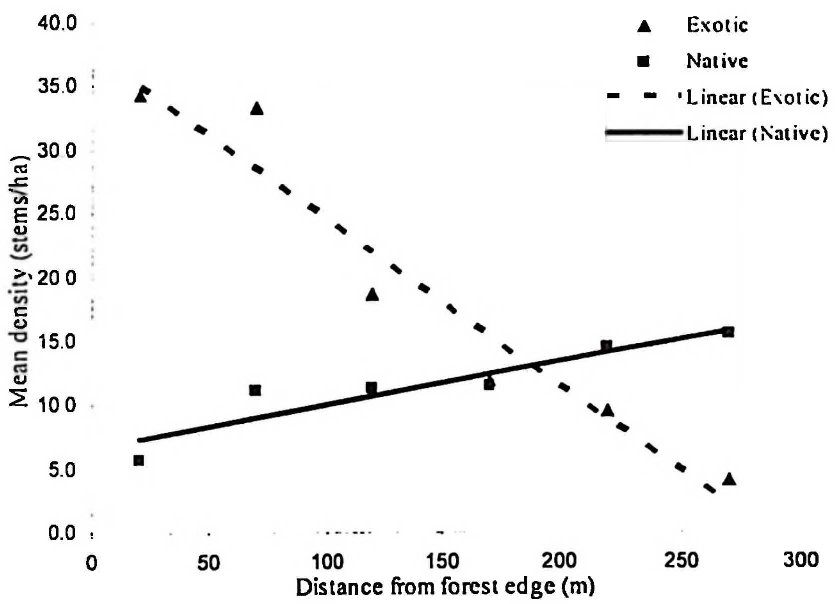


Figure 6: The overall relationship between the mean density and distance from the forest edge of both native and exotic tree species

Linear regression coefficient of exotics = -0.13, *P< 0.0015*, R^2 *0.937* and of natives = 0.033, *P< 0.008*, R^2 *0.854*.

4.1.2 Index of Density Dominance

The indices of density dominance of tree species in the study area are presented in Appendix 6. The IDD of *Castilla* at 20 m from the edge is 0.28, which is the same as that of *Cedrela* (0.28). This value is twice as much as that of *Funtumia africana* (0.14) the most abundant native tree at this distance from the edge. Again, *Castilla* dominates the area at 70 m from the forest edge by the IDD value of 0.28. The area at 120 m and 170 m from the forest edge is dominated by the native tree *Funtumia africana* (IDD values 0.33 and 0.42 respectively) while the area at 220 m and 270 m is dominated by native tree *Cephalosphaera usambarensis* with IDD values of 0.3 and 0.3 respectively. Disturbance has probably facilitated the invasion by *Castilla* and *Cedrela* which have occupied the area that was initially dominated by native trees mainly *Cephalosphaera* and *Funtumia* (Hamilton, 1989; Sawe, 1997). Therefore among the exotic species, *Castilla* and *Cedrela* are the ones that exhibit density dominance in the area close to the forest edge, while *Maesopsis* shows dominance in the interior.

4.2 Frequency Distribution of *Castilla elastica* by dbh Size Classes

The frequency distribution of *Castilla elastica* by dbh size classes in the PCQ method survey is shown in Fig. 7. The frequency distribution is negatively skewed, with nearly half (49%) of individuals having a diameter between 10 and 20 cm, compared to just 6% of individuals with a dbh of 50 to 60 cm. In addition to this, frequency distribution from the population structure study which considered regenerants and young trees with dbh less than 10 cm was also negatively skewed with an average of more than 56 % of individuals being seedlings (Appendix 8). The assessment of the size-class distribution of a plant population provides knowledge about the population structure and function of a given population (Tsingalia, 1982). From the results of this study, heavy recruitment of regenerants into the population was obvious (Plate 4). However, the survival rate of the lower size classes into the larger size classes was relatively low, leading to a decline in tree densities with increased size. High densities of seedlings in this area highlight the importance of propagule density in predicting successful invasion of *Castilla*.

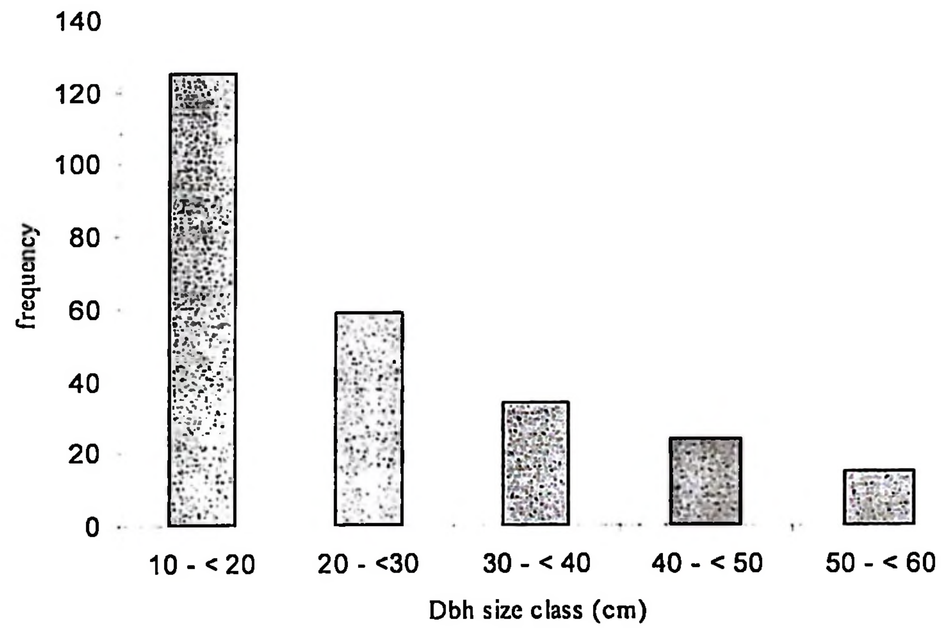


Figure 7: Frequency distribution of *Castilla* by dbh size classes in Amani Nature Reserve

The diameter distribution of *Castilla* in this area followed the reverse J shaped form. Therefore this suggests an expanding population which is characterized by many young individuals than mature individuals in a population. This is a pattern that is expected for many invasive tree species and therefore shows the potential for population increase because the number of the young individuals in a population exceeds the number of mature individuals thus assures replacement and perpetuity. Forest trees ensure their perpetuity through regeneration; this may be through seedlings or through vegetative propagation by coppicing or root suckers (Schmidt, 2000). For tree species such as *Castilla* which regenerates through seedlings (Swarbrick, 1997) have difficult time to perpetuate if their seeds are severely damaged or consumed. After the seed germination the major problem that arises is the growth and development through seedling, sapling, juvenile stage and finally a full mature tree. Through these stages plants encounter a number of problems which either retard their growth or kill them outright.

The frequency distribution of dbh size exhibited by *Castilla* in this part of ANR shows that *Castilla* do not encounter severe herbivory and therefore *Castilla* may form a successful invader in the lowlands of ANR if disturbance continues. Fruits of *Castilla* are attractive to numerous frugivores such as; large vertebrates, birds, and insects. These use the tree as a food source and apparently the trees are important centres of their daily activity. In Costa Rica, Collared Peccary and Coati, are reported to be the prime dispersers of *Castilla*, both of which swallow whole fruits and defecate the seeds (Wayne *et al.*, 2004). Seeds of *Castilla* are very hard to be consumed by many of frugivores; therefore if swallowed, they will be defecated. In ANR, numerous seedlings of *Castilla* would be seen on the course of seasonal streams (personal observation) and if given the light opportunity, they have a fair chance of growing into mature trees.

Plate 3: **Huge number of seedlings of *Castilla* in Samoa. Source: PIER (2006)**

Plate 4: **Huge number of saplings of *Castilla* in ANR. *Picture by:* John Richard (2006)**

4.3 Abundance of *Castilla elastica* in the Four Levels of Forest Disturbance

Highly disturbed forest had a higher density of *Castilla* (192 stems/ha) than all the other three disturbance levels (Fig. 8). The lowest density of *Castilla*, with a mean of 26 stems per hectare was found in the mildly disturbed forest. There was a significant difference ($F_{3,16} = 28.2$, $P<0.001$) in number of stems per hectare between the four disturbance levels. Disturbance commonly enhances the abundance and distribution of exotic plants (D'Antonio *et al.*, 1999). As expected, tree density of *Castilla* was higher in the forest that experienced high disturbance than the lower disturbed forests. In its native range, *Castilla* is an abundant tree species in secondary forests, disturbed forests and open areas such as clearings and forest edges (Sakai, 2001). The positive role of disturbance in some plant invasions appears to help many newcomers to establish nearly monospecific stands (Hierro *et al.*, 2006). The tree density of 192 stems/ha exhibited by Castilla in highly disturbed forests, could pose a threat to the existing native population.

Increase in disturbance particularly removal of trees is known to increase the amount of light, space and amount of other unused resources, which may be readily available for any particular invasive or pioneer plant species (Cronk and Fuller, 1995). Nevertheless, disturbance might not be the only factor that enhanced the abundance of *Castilla*, due to the fact that successful invasion depends on a number of factors, including the attributes of the invaders, the characteristics of the community and propagule availability (Lonsdale, 1999) as discussed earlier. However, many studies (Kotanen *et al.*, 1998; Fine, 2002; Ross *et al.*, 2002; Knops *et al.*, 2003; Leishman and Thomson, 2005) have shown that invasion success of exotic species is positively related to physical disturbance such as construction of tracks, roads and buildings. Hubbell (1980) commented that species recruitment in a particular site of a forest community depends on several factors. Such factors include the availability of the environment that can suit its regeneration at that particular time, developmental stage of that particular forest community and the type of the species in question. The pioneer species will have high recruitment in young forest communities (Binggeli and Hamilton, 1993) that is why *C. elastica* a pioneer species, successful invades secondary forests and disturbed forests.

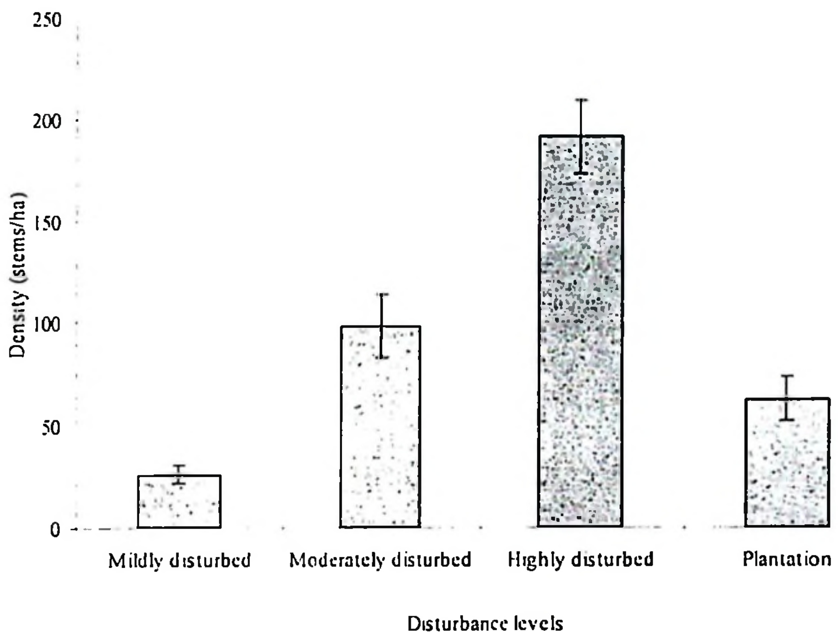


Figure 8: Mean tree densities of *Castilla elastica* in the four different levels of forest disturbance in Amani Nature Reserve
Error bars are ± 1SE.

4.4 Effect of Canopy Cover on the Density of *Castilla*

Result from this study shows that only 19% of the variations in density of *Castilla* could be explained by the change in canopy overstorey. There was a poor negative correlation ($r = -0.43$, $n = 18$, $P<0.059$) between the canopy overstorey (Appendix 9) and the density of *Castilla*. This suggests that, light availability may not be the only factor that facilitates increase in population abundance of *Castilla*. However, the duration of data collection might have influenced the results of this study, due to the fact that during the time of data collection *Castilla* trees had restored their leaves which before that they were almost all shed.

4.5 Population Structure

Population structure of an invasive plant provides explanations on the expected compositional change of an invaded area. Usually invasive plant species are characterised by high ratio of mature to small individuals. The average number of regenerants per mature cosexual tree of *Castilla* is not static, it changes from year to year even within the given year, as the amount of resources fluctuates, however the average number recorded in this study was 118 regenerants per cosexual tree.

4.5.1 Regeneration of *Castilla elastica*

The frequency of seedlings (1979), saplings (2187) and juveniles (1666) were almost equal in the area with low levels of disturbance (Appendix 8). In areas with high, medium disturbance levels and plantation, seedling frequencies were relatively high but the frequencies of the later life stages declined sharply. Across all life stages, frequencies of individuals were highest in the highly disturbed area. There is a significant association ($\chi 2$ = *21*, df = 12, *P >0.001*), between the level of disturbance and the frequency of individuals at different life stages of growth (Fig. 9). This result shows that. *Castilla* is capable of recruiting many seedlings in relatively more disturbed areas during the initial stages of which a small proportion make to large trees. *Castilla* is a heavy seeder plant species (PIER, 2006), which has an average of 2035 seeds per kg with a total germination of 86% (Sautu *et al.*, 2006). During a study on fruit dispersal of *Castilla* at La Selva Biological Station in Costa Rica, Wayne *et al.*, (2004) recorded a seed density of 371 seeds per square meter within the seed shadow in the forest.

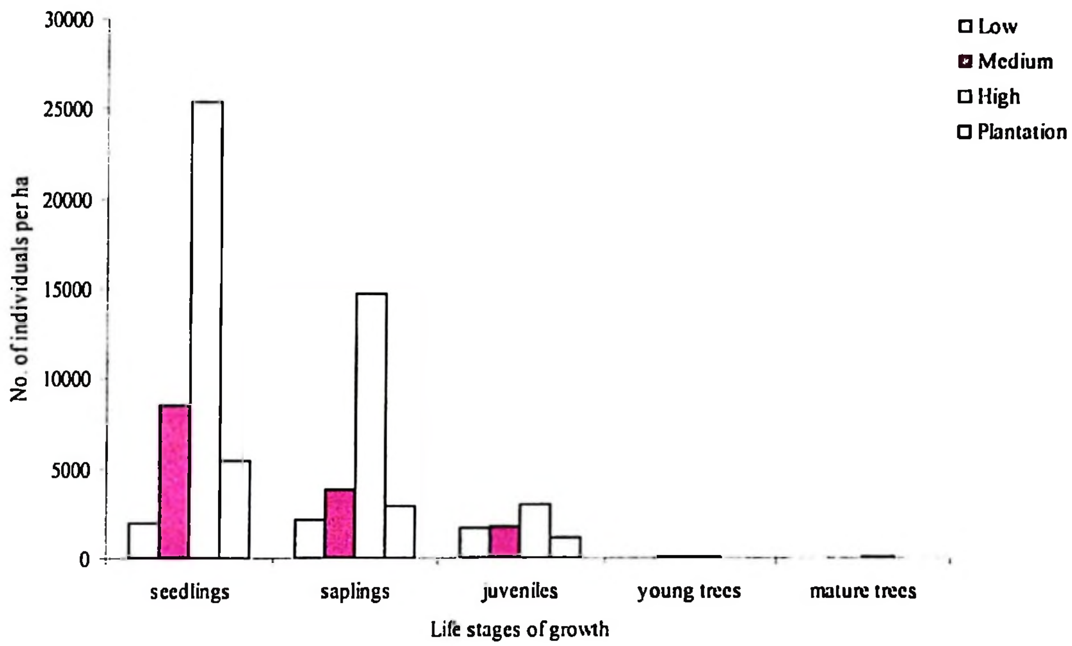


Figure 9: The relationship between level of disturbance and the frequency distribution according to the five life stages of growth

Despite the positive advantages *Castilla* has toward invasiveness, *Castilla* seeds lose their viability just after a month (Sautu *et al.,* 2006). Otherwise, the regeneration from seed could recover fully after disturbance as a result of germination of seeds buried in the soil when they are exposed to the surface. Studies in various habitats have indicated that species with long-lived seed banks regenerate successfully from buried seeds usually after disturbance (Baskin and Baskin, 1978; Baker, 1989). According to Whitmore (1983), short-lived pioneer species which are also relatively shade-intolerant appear shortly after disturbance. However, if there are no viable seeds in the seed bank, regeneration may take place either through recently dispersed seeds, or in the case of undisturbed vegetation, through slow-growing seedlings and saplings, accumulated below the forest vegetation.

4.5.2 Influence of male trees on the number of regenerants

Results revealed that there was no significant difference (*t*-test, $P > 0.05$) in the number of regenerants between the areas with and without male trees (Fig. 10 and Appendix 10). Presence or absence of male trees might have not influenced the number of offsprings hence regenerants in a population. *Castilla elastica* is an androdioecious plant, whereby in a population there are trees that bear only male inflorescences, while others produce pistillate inflorescences flanked by a few staminate inflorescences, which are smaller than primary ones (Plate 1 and 2) and the stamens on both types of trees are fertile (Pittier, cited in Sakai, 2001). Therefore the cosexual *Castilla* plant is capable of producing viable seeds through self pollination. The androdioecy in *C. elastica* might have evolved from dioecy. Among three *Castilla* species, closely related *C. elastica* and *C. ulei* are androdioecious, but *C. tunu* is strictly dioecious without complemental inflorescences (Sakai, 2001). The ability of *Castilla* to produce seeds through self-pollination must be essential when they invade a new habitat without conspecific trees.

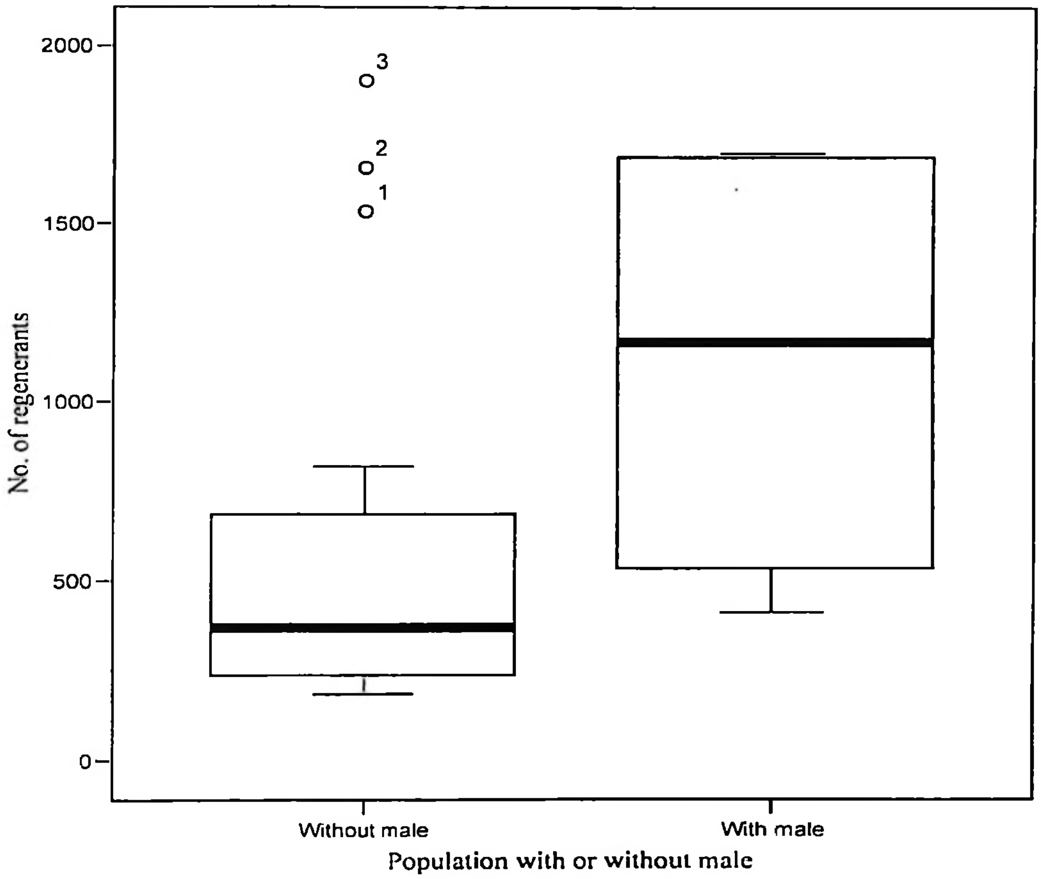


Figure 10: Number of regenerants in areas with male and without male trees of *Castilla* in Amani Nature Reserve

Out of 134 mature *Castilla* trees that were observed in the 20 plots surrounding mother trees. only 6 trees (4.5 %) were male and 128 trees (95.5 %) were cosexual. These observations imply that the sex ratio of *Castilla* in ANR is female-biased. Female-biased sex ratio is of great importance for invasion and colonization success in a new habitat. Male-biased sex ratio is a common phenomenon in stressful habitats since the female reproductive investment per plant is sometimes larger than the male investment

(Tetsuto *et al.*, 2002). Reproductive investment may results into higher mortality of female individuals than male, thus form a population with male-biased sex ratio. These results from ANR suggest that the original sex ratio of *C. elastica* was probably female-biased and therefore equality in sex ratio (if any) in other areas where the plant also naturalises, might be brought about by environmental stresses. In some of its native ranges, *C. elastica* grows to 5 to 10 m high (Woodson and Schery, 1960), while in ANR *Castilla* grows to a height of 15 to 30 m (personal observations). This growth performance shows that, *Castilla* in ANR are not stressed and therefore exhibit their original sex ratio (female-biased) which may also favour its invasive ability.

4.5.3 Regeneration with respect to distance from the cosexual plant

There is a strong negative correlation between density of seedlings and distance from the mother tree ($r = -0.97$, $P < 0.001$) and between density of saplings and distance from the mother tree ($r = -0.85$, $P < 0.01$) (Fig. 11). The relationships between densities of juveniles and young trees and distance from the mother tree are not strong but they are also negative suggesting that as the distance increases from the mother tree, both densities of juveniles and young trees of *C. elastica* decreases too. The reverse is the case for mature trees, because there is a positive relationship, showing that as the distance increases from the mother tree, density of trees with dbh > 20 cm increases too. This is a common phenomenon for those tree species which do not exhibit (intraspecific competition) competition between individuals of the same species. Bakari (1995) reported the same trend when studied the regeneration trend of tropical lowland rain forest tree species in Rau Catchment Forest Reserve in Tanzania, out of eight tree species studied, five of them showed negative relationship between distance from mother tree and seedling density.

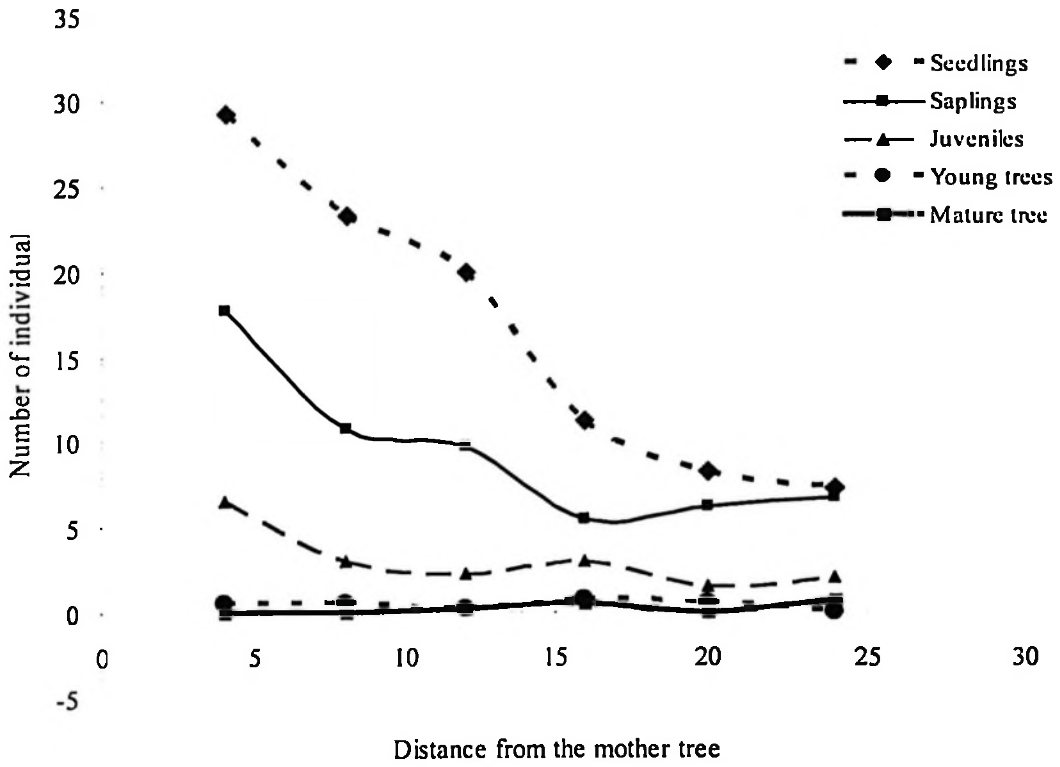


Figure 11: Relationship between the five classes of *Castilla elastica* and the distance from the mother tree in Amani Nature Reserve

Linear regression coefficient of seedlings is -0.97, P< 0.001; saplings -0.85346, P< 0.01; Juveniles -0.7831, P< 0.052; Young trees -0.41, P< 0.32 and Mature trees 0.67, P< 0.52.

Seed dispersal by frugivores confers a variety of benefits on plants; perhaps the most widely cited advantage is the escape of seeds away from the vicinity of parent plants (Howe and Smallwood, 1982). Janzen (1970) described the seed density distribution as a curve declining exponentially with distance from the parent tree, and hypothesized that host-specific seed and seedling predators prey upon conspecifics near the parent, thus preventing recruitment. In this way, distance and density were inversely linked, and vertebrate dispersers assumed an essential role by moving seeds away from the parent, thereby increasing the probability of seed survival with "escape" from density-dependent and distant-dependent mortality. Results from this study do not agree with Janzen (1970) hypothesis of "escape". However, it is obvious that trees which have large stature like *Castilla* may not afford to clump together even if they do not exhibit intraspecific competition.

4.6 Weed Risk Assessment of *Castilla elastica* in Amani Nature Reserve

The total score obtained from this WRA in ANR (which was based on the ecology, biology, abundance and distribution of *Castilla elastica*) was 11 (Appendix 1). This score falls in the "reject" category (Ref. 3.3.5) and shows that *Castilla* has a risk of becoming invasive in the reserve. Therefore *Castilla* needs a close monitoring and also, efforts should be geared to control its population growth in ANR. Results from ANR are in agreement with the results obtained on the WRA carried out in Pacific Islands as reported in http://www.hear.org/pier/species/castilla_elastica, where *Castilla* elastica had a risk assessment score of seven. In these areas, *Castilla* is considered as one of the noxious invasive plants (Space and Flynn, 2002).

CHAPTER FIVE

5.0 CONCLUSION AND RECOMMENDATIONS

Invasion by non-native species is a process usually occurring over a period of years or decades, however, the process can be shortened as a result of habitat destruction by man. Amani Nature Reserve was formerly a continuous block of forests extending west and east of Usambaras, but threats to the forest ecosystems such as cash crop cultivation (tea, coffee, sisal and cardamom), logging, pit-sawing and agricultural encroachment have led to its fragmentation. Among the best documented impacts of fragmentation are edge effects. Edge zones are susceptible to invasion by alien pioneer plants, because these zones are usually less shady, and warmer than the forest interiors. Therefore, this favours species that are shade-intolerant which is the characteristic of most alien invasive plants.

Results from this study have shown that *Castilla elastica* poses threats to native plant communities in the lowland forests of Amani Nature Reserve. At the edge of the forest in the study area, approximately 21% of trees with dbh greater than 10 cm were *Castilla* trees. The species is also distributed to the interior of forest up to 300 m from the forest edge. Threats posed by *Castilla* to native species are a result of its significant population increase due to heavy recruitment of seedlings in both the forest gaps and highly disturbed forests. Similarly, results from this study show that disturbance has a positive role in facilitating the invasion of *Castilla*. Therefore, if disturbance continues in the lowlands of ANR, recruited seedlings of *Castilla* will have a fair chance to grow and form big populations of large trees that may out-compete the slow growing native species. The score (11) on Weed Risk Assessment of *Castilla* in ANR has shown that the species has a potential of becoming invasive in the reserve.

In terms of population characteristics, *Castilla elastica* has an expanding population characterized by many young individuals than mature individuals in a population. The sex ratio of *Castilla* in ANR is female-biased and presence or absence of male individuals in a population does not significantly influence the number of regenerants produced. Therefore the cosexual *Castilla* plant is capable of producing many viable seeds through self pollination. The density of seedlings and saplings decreased with distance from the mother tree. With regard to Weed Risk Assessment and distribution of *C. elastica*, results indicated that the main limitation on the distribution of this plant species in ANR is altitude.

Fruits of *Castilla* are attractive to many frugivores; however the prime dispersers observed in the study area were colobus monkeys, blue monkeys, birds and squirrels. These animals chew the pulp and expel the large seeds. Evidences of regeneration in gaps and open areas in the forest where mature trees were absent, shows that the few seeds that happen to be swallowed are not destroyed during passage through the alimentary canal and when defecated are able to germinate. Also, many seedlings were observed on courses of seasonal streams suggesting that *Castilla* seeds are also dispersed from mother trees by water down stream during rain seasons.

Finally, further research is needed on the management of the growing population of *Castilla*, which will take into account the social-economic effects as a result of the management method that may be applied. However, there is an urgent need to prioritize management of alien invasive plants in ANR, so that the eradication/control of one invasive plant may not result into providing room for more serious and destructive invasive plants.

REFERENCES

Adair, R. (2002). Black Wattle: South Africa manages conflict of interest. *Biocontrol News and Information* 23(1): 5 – 7.

Bakari, R. J. (1995). Natural regeneration of twelve trees species in Rau Forest Reserve Tanzania: Influence of the distance from mother tree and orientation. Dissertation for Award of MSc Degree at Wageningen Agricultural University, The Netherlands, 45pp.

Baker, H. G. (1989). Some aspects of the natural history of seed bank. In: *Ecology of soil seed bank. (Edited by Leck, M. A., Parker, V. T. and Simpson, R. L.)*, Academic Press, San Diego, California. pp. 9 – 21.

Baskin, J. M. and Baskin, C. C. (1978). The seed bank in population of an endemic plant species and its ecological significance. *Biological Conservation* 14: 125 - 130.

Binggeli, P. (1989). The ecology of *Maesopsis* invasion and dynamics of the evergreen forest of the East Usambara Mountains and their implications for forest conservation and forestry practices. In: *Forest conservation in the East Usambara Mountains Tanzania. (Edited by Hamilton, A. C. and Bensted-Smith, R.)*, IUCN, Gland. pp. 256-269.

Binggeli, P. (1998). *An overview of invasive woody plants in the tropics*. School of Agricultural and Forest Sciences, Publication Number 13, University of Wales, Bangor. 83pp.

Binggeli, P. and Hamilton, A. C. (1993). Biological invasion by *Maesopsis eminii* in the East Usambara forests. Tanzania. *Opera Botanica* 121: 229 - 235.

Binggeli, P., Ruffo, C. K., Taylor, D. and Hamilton, A. C. (1989). Seed banks in forest soils. In: *Forest conservation in the East Usambara Mountains Tanzania. (Edited by Hamilton, A. C. and Bensted-Smith, R.)*, IUCN, Gland. pp. 302 - 307.

Busing, R. T. (1994). Canopy cover and tree regeneration in old-growth cove forests of the Appalachian Mountains. *Vegetatio* 115: 19 – 27.

Chilima, C. G. (2005). An update of forest invasive species in Malawi. Forest invasive species network for Africa. [http://www.fao.org/forestry/site/26951] site visited on 25/09/2006.

Cordeiro, N. J., Patric, D. A. G., Munisi, B. and Gupta, V. (2004). Role of dispersal in the invasion of an exotic tree in an East African submontane forest. *Journal of Ecology* 20: 449 - 457.

Cottam, G., Curtis, J. T. and Hale, B. W. (1953). Some sampling characteristics of a population of randomly dispersed individuals. *Ecology* 34: 741 - 757.

Crawley, M. J. (1986). What makes a community invasible? In: *Colonisation, succession and stability. (Edited by Gray, A. J. Crawley, M. J. and Edwards, P. J.)*, Blackwell Scientific, Oxford, UK. pp. 34 - 48.

Cronk, Q. B and Fuller, J. L. (1995). *Plant invaders: The threat to natural ecosystems.* Chapman and Hall, London, UK. 127pp.

Chytrý, M., Pyšek, P., Tichý, L., Knollová, I. and Danihelka, J. (2005). Invasions by alien plants in the Czech Republic: A quantitative assessment across habitats. *Preslia* 77: 339 – 354.

D'Antonio, C. M., Dudley, T. M. and Mack, M. (1999). Disturbance and biological invasions: Direct effects and feedbacks. In: *Ecosystems of Disturbed Ground (Edited by Walker, L. R.)*, Elsevier, Amsterdam. pp. 413 – 452.

Daehler, C. C. and Carino, D. A. (1999). Threats of invasive plants to the conservation of biodiversity. In: *Biodiversity and Allelopathy: From Organisms to Ecosystems in the Pacific. (Edited by Chou, C. H., Waller, G. R. and Reinhardt, C.)*, Academic Press, Taipei. pp. 21 – 27.

Davis. M. A., Grime, J. P. and Thompson, K. (2003). Fluctuating resources in plant communities: a general theory of invasibility. *Journal of Ecology* 98: 528 – 534.

Drake. J. A., Mooney, H. A., di Castri, F., Groves, R. H., Kruger, F. J., Rejmánek, M. and Williamson, M. (Eds.) (1989). *Biological Invasions: A global Perspective.* Wiley, Chichester, UK. 523pp.

FAO. (2001a). The state of Food and Agriculture 2001. Rome, Italy. [http://www.fao.org/docrep/003/x9800e/x9800e14.htm] site visited on 25/6/2006

FAO. (2003). *The status of invasiveness of forest tree species outside their natural habitat: A global review and discussion paper.* Biosecury working paper. Rome, Italy. 83pp.

FAO. (2005). *Alien Invasive Species: Impact on Forests and forestry.* Forest Health and Biosecurity working papers. Rome. Italy. 60pp.

Fine, P. V. A. (2002). The invasibility of tropical forests by exotic plants. *Journal of Tropical Ecology* 18: 687 – 705.

Fowler, S. and Nyambo, B. (1996). *Survey of invasive weeds affecting biodiversity in the forest of the East Usambara Mountains, Tanzania.* East Usambara Catchment Forest Project, Technical Paper No. 20. Forest and Beekeeping Division, Dar es Salaam, Tanzania. 15pp.

Frontier Tanzania. (2001). *Amani Nature Reserve: A biodiversity Survey.* East Usambara Conservation Area Management Programme. Technical Paper No 52. Dar es Salaam, Tanzania. 126 pp.

Ganzhorn, J. (1995). Causes for lemur population decline in degraded and secondary forests of the Morondava Region. In: *Environmental Change in Madagascar.* (*Edited by Goodman, S. M., Patterson, B. D. and Sedlock, J. L.*), The Field Museum, Chicago. pp. 39 - 41.

Geddes, N. R. (1998). Maesopsis invasion of the tropical forest in the East Usambara Mountains, Tanzania. Thesis for Award of M.Phil Degree at University College of North Wales, Bangor. 177 pp.

Greenway, P. J. (1934). Report of a botanical survey of the indigenous and exotic plants in cultivation at the East Africa Agricultural Research Station, Amani, Tanganyika Territory. 511pp.

Groom, M. J., Meffe, G. K. and Carrol, C. R. (2006). *Principle of Conservation Biology.* Third Edition. Sinauer Associates Publishers, Sunderland. Massachusetts, USA. 779pp.

Hall, J. B. (1990). Succession in a natural forest at Mazumbai. In: *Research for conservation of Tanzanian catchment forests.* (*Edited by Hedberg. I. and Persson. E.*), 13-17 March 1989. Morogoro, Tanzania. pp. 24 - 38.

Hall, J. B. (1995). *Maesopsis eminii and its status in the East Usambara Mountains.* East Usambara Catchment Forest Project, Technical paper No. 13. Forest and Beekeeping Division, Dar es Salaam, Tanzania. 41pp.

Hamilton, A. C. (1989). History of resource utilization and management: The pre-colonial period. In: *Forest conservation in the East Usambara Mountains, Tanzania.* (*Edited by Hamilton, A. C. and Bensted-Smith, R.*), IUCN, Gland. pp. 35 - 44.

Hamilton, A. C. and Bensted-Smith, R. (1989). The place and the problem: In: *Forest Conservation in the East Usambara Mountains, Tanzania. (Edited by Hamilton, A. C. and Bensted-Smith, R.)*, IUCN, Gland. pp. 22 - 32.

Hamilton, A. C. and Macfadyen, A. (1989). Climatic change on the East Usambara. In: *Forest conservation in the East Usambara Mountains, Tanzania. (Edited by Hamilton, A. C. and Bensted-Smith, R.)*, IUCN, Gland. pp. 103 - 108.

Hammel, B. E. (1986). The vascular flora of La Selva Biological Station. Costa Rica, Moraceae. *Selbyana* 9: 243 - 259.

Hierro, J. L., Villarreal, D., Eren, O., Graham, J. M. and Ragan M. C. (2006). Disturbance Facilitates Invasion: The Effects Are Stronger Abroad than at Home. *American naturalist* 168 (2): 144 - 156.

Honess, B. L. (1963). *The Amani Plantations.* A detailed report on a survey made in 1961 – 1962. Royal Botanic Gardens, Kew. 85pp.

Howe, F. H. and Smallwood, J. (1982). Ecology of seed dispersal. *Annual Review of Ecology and Systematics* 13: 200 – 228.

Hubbell, S. P. (1980). Seed predation and the coexistence of tree species in tropical forest. *Oikos* 35: 214 – 229.

Hughes, C. E. (1994). Risks of species introductions in tropical forestry. *Commonwealth Forestry Review* 73: 243 – 252.

Hulme, P. (2003). Biological invasions: winning the science battle but losing the conservation war? *Oryx* 37(2): 178 – 193.

Hulme, P. (2006). *Invasive plants in tropical forest ecosystems*: An introduction to mapping and monitoring biological invasion. 12 -16 June 2006. Amani Nature Reserve, Tanga, Tanzania. 29pp.

Hulme, P. (2007). *Invasion Risk Assessment in tropical ecosystems*: An introduction to assessing the risk of plant invasion and prioritizing management. 14-18 May 2007, Amani Nature Reserve, Tanga, Tanzania. 70pp.

Iversen, S. T. (1991a). *The Usambara Mountains, NE Tanzania: History, Vegetation and conservation.* Uppsala University, Uppsala. 86pp.

Janzen, D. H. (1970). Herbivores and the number of tree species in tropical rain forests. *American Naturalists* 104: 501 – 528.

Johannson, S. G., Katigula. M. I. L., Mashauri, S. and Mndolwa. A. (1997). *Conservation of biodiversity in the East Usambaras*. Approaches and experience from the East Usambara Catchment Forest Project. EUCFP. Tanzania. 56pp.

Kilahama, F. B. (1998). *Threats to the East Usambara biodiversity*. East Usambara Catchment Forest Project, Tanzania. 58pp.

Killenga, R. R. (2007). Effects of human disturbances on endemic and threatened plant species in Amani Nature Reserve, Tanzania. Dissertation for Award of MSc Degree at Sokoine University of Agriculture. Morogoro. Tanzania. 157pp.

Knops, J. M. H., Griffin, J. R. and Royalty, A. C. (2003). Introduced and native plants of the Hastings Reservation, central coastal California – a comparison. *Biological Conservation* 91: 115 - 123.

Kotanen, P. M., Bergelson. J. and Hazlett, D. L. (1998). Habitats of native and exotic plants in Colorado shortgrass steppe – a comparative approach. *Canadian Journal of Botany* 76: 664 – 672.

Laps, R., Pizo, M. A., Silva, W. R. and Galetti, M. (2002). Frugivory in cotingas of the Atlantic Forest of southeast Brazil. *Ararajuba* 10: 177 - 185.

Leishman, M. R. and Thomson, V. P. (2005). Experimental evidence for the effects of additional water, nutrients and physical disturbance on invasive plants in low fertility Hawkesbury Sandstone soils, Sydney, Australia. *Journal of Ecology* 93: 38 – 49.

Lemmens, R. H. M. J., Chung, R. C. K., Boer, E. and Noshiro, S. (1995). *Cedrela* (Meliaceae). In: *Timber trees: Minor commercial timber*. (*Edited by Lemmens. R. H. M. J., Soerianegara. I. and Wong. W. C.*), Blackhuy Publisher, Leiden. 122 - 127pp.

Lemon, E. P. (1957). A new instrument for measuring forest overstory density. *Journal of Forestry* 55(9): 667 - 668.

Leung, B., Drake, J. M. and Lodge, D. M. (2004). Predicting invasion: Propagule pressure and the gravity of Allee effect. *Ecology* 85: 1651 – 1660.

Lonsdale, W. M. (1999). Global patterns of plant invasions and the concept of invasibility. *Journal of Ecology* 80(5): 1522 - 1536.

Lonsdale, W. M., Miller, I. L. and Forno, I. W. (1995). *Mimosa pigra*. In: *The biology of Australian weeds*. (*Edited by Groves, R. H., Shepherd, R. C. H. and Richardson, R. G.*), Melbourne, Australia. pp. 69 – 188.

Lulandala, L. L. L. (1985). Intercropping *Leucaena leucocephala* with maize and beans. Thesis for Award of PhD Degree at Sokoine University of Agriculture, Morogoro, Tanzania. 254pp.

Mack, R. N., Simberloff, D., Lonsdale, W. M., Evans, H., Clout, M. and Bazzar, F. A. (2000). Biotic invasions: Causes, epidemiology, global consequence, and control. *Ecology Applications* 10: 689 - 710.

Madoffe, S. S. (1993). Seasonal abundance and host selection of bark and wood boring beetles in freshly felled *Cedrela mexicana*, *Khaya nyasica* and *Milicia excelsa* in lowland forest in Tanzania. Thesis for Award of PhD Degree at Agricultural University of Norway. 163pp.

Markus, R. (2006). Management plan for the Botanical Research Institute at Amani. [http://www.rbgkew/conservation/cpdu/amani.htm] site visited on 2/5/2006.

Masters, R. A. and Sheley, R. L. (2003). Principles and practices for managing rangeland invasive plants. *Journal of Range Management* 74: 502 - 517.

Matthews, S. (2004). Tropical Asia invaded: The growing danger of invasive alien species. The Global Invasive Species Programme (GISP) Secretariat, Cape Town, South Africa. [http://www.gisp.org/downloadpubs/gispAsia.pdf] site visited on 6/2/2006.

Matthews, S. and Brand, K. (2004). Africa invaded: The growing danger of invasive alien species. The Global Invasive Species Programme (GISP) Secretariat, Cape Town, South Africa. [http://www.gisp.org/downloadpubs/gisp%20africa%202.pdf] site visited on 6/2/2006.

Maunder, M. (1999). *Strategic Development Plan for the Amani Botanic Garden*. Conservation Projects Developemt Unit. Royal Botanic Gardens, Kew. 39pp.

McFadyen, R. E. C. (1998). Biological control of weeds. *Annual Review of Entomol* 43: 369 – 393.

Meyer, J. and Malet, J. (2006). Agroforestry alien trees as invasive plants in the Pacific Islands. [http://www.fao.org/docrep/006/ad672e/ad672e03htmP2918_11256P2918_112565] site visited on 29/5/2006.

Mitchell, K. (2001). *Quantitative analysis by the point-centered quarter method.* Department of Mathematics and Computer Science, Hobart and William Smith Colleges, Geneva. 67pp.

Mugasha, A. G. (1981). The Silviculture of Tanzanian indigenous tree species - *Maesopsis eminii.* Tanzania Silviculture Technical Note Number 52. Dar es Salaam, Tanzania. 64 pp.

Neal, M. C. (1965). *In Gardens of Hawaii.* Museum Special Publication 50, Bishop Museum Press. 924 pp.

Pacific Island Ecosystems at Risk (PIER). (2006). [http://www.hear.org/pier/species/castilla_elastica.htm] site visited on 9/6/2006

Parker, I. M. (2004). *Mating pattern and rates of biological invasion.* Proceeding of the National Academy of Science 101(38): 13695 - 13696

Pheloung, P. C. (2001). Weed risk assessment for plant introductions to Australia. In: *Weed risk assessment.* (*Edited by Groves, R. H., Panetta, F. D. and Virtue, J. G.*), CSIRO Publishing. Collingwood, Australia. pp. 83 – 92.

Pimentel, D., Lach. L., Zuniga, R. and Morrison, D. (2000). Environmental and economic costs of nonindigenous in the United States. *Bioscience* 50(1): 53 - 65.

Ramakrishnan, P. S. (1991). Biological invasion in the tropics: An overview, in ecology of biological invasion in the tropics. International Scientific Publication, New Delhi. pp. 1 - 20.

Rejmánek, M. (1996). Species richness and resistance to invasions. In: *Diversity and processes in tropical forest ecosystems.* (*Edited by Orians, R. D., Dirzo, R. and Cushman, J. H.*), Springer, New York. pp. 153 - 172.

Rejmánek, M. and Richardson, D. M. (1996). What attributes make some plants species more invasive? *Ecology* 77: 1655 - 1661.

Richardson, D.M. (1999). Invasive plants species and invasible ecosystems. In: *Invasive species and biodiversity management. Population and Community Biology series.* (*Edited by Sandlund, O. T., Schei, P. J. and Viken, A.*), Kluwer Academic Publishers, Dordrecht, The Netherlands. pp. 79 - 102.

Richardson, D. M and Rejmánek, M. (2004). Conifers as invasive aliens: a global survey and predictive framework. *Diversity and Distributions* 10: 321 – 331.

Ross, K. A., Fox, B. J. and Fox, M. D. (2002). Changes to plant species richness in forest fragments: Fragment age, disturbance and fire history may be as important as area. *Journal of Biogeography* 29: 749 – 765.

Sakai, S. (2001). Thrips pollination of androdioecious *Castilla elastica* (Moraceae) in a seasonal tropical forest. *American Journal of Botany* 88(9): 1527 - 1534.

Sandy, R. F., Boniface, G. and Rajab, I. (1997). *A survey and inventory of the Amani Botanical Garden.* East Usambara Catchments Forest Project, Technical Paper No. 38. Forestry and Beekeeping Division, Dar es Salaam, Tanzania. 51pp.

Sautu, A., Jerry, M. B., Carol, C. B. and Richard. C. (2006). Studies on the seed biology of 100 native species of trees in a seasonal moist tropical forest, Panama, Central America. *Forest Ecology and Management* 234: 245 – 263.

Sawe, T. C. (1997). The regeneration status of indigenous tree species in the forests of East Usambara Mountains, Tanzania. Dissertation for Award of MSc Degree at Sokoine University of Agriculture, Morogoro, Tanzania. 108 pp.

Sawyer, J. (1993). *Plantations in the Tropics.* Environmental Concern Programme. IUCN/UNEP/WWF. Gland, Switzerland. 78pp.

Schabel. H. (1990). Tanganyika forestry under German colonial administration, 1891–1919. *Forestry and conservation history* 34: 130 - 141.

Schmidt, L. (2000). Guide to Handling of Tropical and Subtropical Forest Seed. Danida Forest Seed Centre. CSIRO Forestry and Forest Products, Australian Tree Seed Centre. 89pp.

Schmitz, D. C., Simberloff, D., Hofstetter, R. H., Haller, W. and Sutton. D. (1997). The ecological impact of nonindigenous plants. In: *Strangers in paradise: impact and management of nonindigenous species in Florida. (Edited by Simberloff, D., Schmitz, D. C. and Brown, T. C.)*, Island Press. Washington, D.C., USA. pp 36 -61.

Schulman, L., Junikka, L., Mndolwa, A. and Rajabu, I. (1998). *Trees of Amani Nature Reserve.* Helsinki University Printing House. Finland. 142pp.

Sheil, D. (1994). Naturalised and invasive plant species in the evergreen forests of the East Usambara Mountains, Tanzania. *African Journal of Ecology* 32:66 - 71.

Space, J. C. and Flynn, T. (2002). *Report to the Government of Samoa on invasive plant species of environmental concern.* USDA Forest Service, Honolulu. 83pp.

Swarbrick, J. T. (1997). *Environmental weeds and exotic plants on Christmas Island, Indian Ocean*: A report to Parks Australia. Weed Science Consultancy, Australia. 24pp.

Tetsuto, A., Yamashita, N., Naboyuki, T. and Hiromichi, K. (2002). Sex Ratio Variation of *Bischofia javanica* (Euphorbiaceae) Between Native Habitat, Okinawa (Ryukyu Islands), and Invaded Habitat, Ogasawara (Bonin Islands). *International Journal of Plant Science* 163(6): 1011 – 1016.

Tsingalia, M. H. (1982). Animals and regeneration of an African rainforest tree. Dissertation for Award of MSc Degree at University of California, Berkeley, USA. 82pp.

Vitousek, P. M. (1987). Biological invasion by *Myrica faya* alters ecosystem development in Hawaii. *Science* 238: 802 - 804.

Wayne, C., Paye, G. D., Roberts, C. and Birsall, D. (2004). Fruit dispersal of *Castilla elastica* in secondary forest and a developed area of the La Selva Biological Preserve, Costa Rica. *Journal of Ecology* 20: 124 – 129.

Weber, E. (2003). *Invasive Plant Species of the World: A Reference Guide to Environmental Weeds*. CAB Publishing, Wallingford, UK. 46pp.

Wester, L. L. and Wood, H. B. (1977). Koster's curse (*Clidemia hirta*): A weed pest in Hawaiian forest. *Environmental conservation* 4(1): 35 - 41.

Willan, R. L. (1965). Natural regeneration of high forest in Tanganyika. *East African Agricultural and Forestry Journal* 31: 43 - 53.

Williamson, M. (1996). *Biological invasion*. Chapman and Hall, London, UK. 143pp.

Williamson, M. and Fitter, A. (1996). The varying success of invaders. *Ecology* 77: 1661 - 1666.

Wittenberg, R. and Cock, M. J. W. (2001). *Invasive alien species: A toolkit of best prevention and management practices*. CAB International, Wallingford, Oxon, UK. 97pp.

APPENDICES

Appendix 1: Alien plant screening system from Pheloung (2001) with minor modificati9ons for use in Eastern Arc Mountains

S/N	Questionnaires	Answers	Score	References
1	**Domestication/cultivation**			
1.01	Is the species highly domesticated?	y=-3, n=0	0	http://www.hear.org/pier/species/castilla_elastica
1.02	Has the species become naturalized where grown?	y=1, n=-1	1	Sakai, 2001; Wayne *et al.*, 2004
1.03	Does the species have weedy races?	y=1, n=-1	-1	http://www.hear.org/pier/species/castilla_elastica
2	**Climate and Distribution**			
2.01	Species suited to tropical or subtropical climate(s) (0-low; 1-intermediate; 2-high)	Append 2	2	Sakai, 2001; Wayne *et al.*, 2004; Kitajima et al., 2005;Sautu *et al.*, 2006
2.02	Quality of climate match data (0-low; 1-intermediate; 2-high) see appendix 2		2	
2.03	Broad climate suitability (environmental versatility)	y=1, n=0	0	http://www.hear.org/pier/species/castilla_elastica
2.04	Native or naturalized in regions with tropical or subtropical climates	y=1, n=0	1	Sakai, 2001; Wayne *et al.*, 2004; Sautu *et al.*, 2006
2.05	Does the species have a history of repeated introductions outside its natural range?	?=-1, n=0 y=-2	-2	Sakai, 2001; Wayne *et al.*, 2004
3	**Weed Elsewhere (depends on 2.01 and 2.02)**			
3.01	Naturalized beyond native range y = 1*multiplier (see Append 2), n= question 2.05		2	
3.02	Garden/amenity/disturbance weed y = 1*multiplier (see Append 2)	n=0	2	http://www.hear.org/pier/species/castilla_elastica
3.03	Agricultural/forestry/horticultural weed y = 2*multiplier (see Append 2)	n=0	4	Wayne *et al.*, 2004
3.04	Environmental weed y = 2*multiplier (see Append 2)	n=0	4	
3.05	Congeneric weed y = 1*multiplier (see Append 2)	n=0	0	

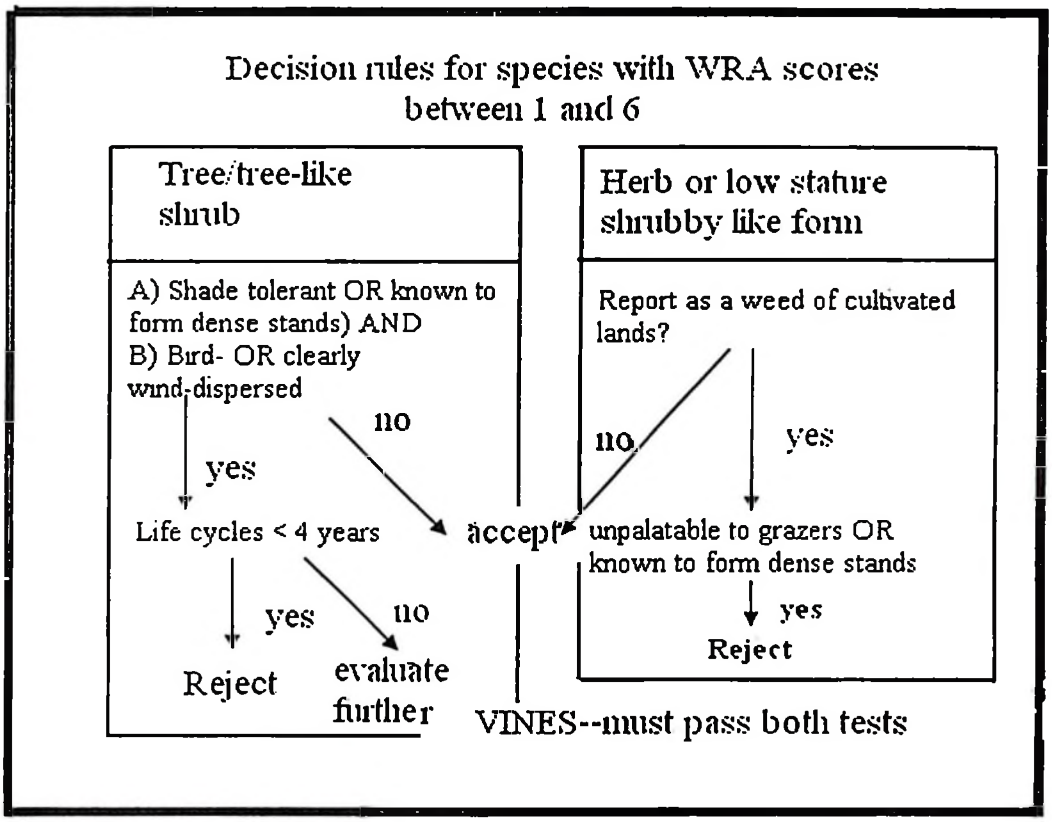
4	**Undesirable traits**			
4.01	Produces spines, thorns or burrs	y=1, n=0	0	http://www.hear.org/pier/species/castilla_elastica
4.02	Allelopathic	y=1, n=0	0	Wayne *et al.*, 2004
4.03	Parasitic	y=1, n=0	0	http://www.hear.org/pier/species/castilla_elastica
4.04	Unpalatable to grazing animals	y=1, n= -1	1	
4.05	Toxic to animals	y=1, n=0		
4.06	Host for recognized pests and pathogens	y=1, n=0		
4.07	Causes allergies or is otherwise toxic to humans	y=1, n=0	0	http://www.hear.org/pier/species/castilla_elastica
4.08	Creates a fire hazard in natural ecosystems	y=1, n=0	0	
4.09	Is a shade tolerant plant at some stage of its life cycle	y=1, n=0	0	Sakai, 2001; Wayne *et al.*, 2004
4.10	Tolerates a wide range of soil conditions	y=1, n=0		
4.11	Climbing or smothering growth habit	y=1, n=0	0	Personal observation
4.12	Forms dense thickets	y=1, n=0	0	Personal observation
5	**Plant type**			
5.01	Aquatic	y=5, n=0	0	http://www.hear.org/pier/species/castilla_elastica
5.02	Grass	y=1, n=0	0	http://www.hear.org/pier/species/castilla_elastica
5.03	Nitrogen fixing woody plant	y=1, n=0	0	http://www.hear.org/pier/species/castilla_elastica
5.04	Geophyte (herbaceous with underground storage organs)	y=1, n=0	0	http://www.hear.org/pier/species/castilla_elastica
6	**Reproduction**			
6.01	Evidence of substantial reproductive failure in native habitat	y=1, n=0	0	Sakai, 2001; Wayne *et al.*, 2004; Sautu *et al.*, 2006
6.02	Produces viable seed.	y=1, n=-1	1	Wayne *et al.*, 2004; Sautu *et al.*, 2006
6.03	Hybridizes naturally	y=1, n=-1	-1	
6.04	Self-compatible or apomictic	y=1, n=-1	-1	
6.05	Requires specialist pollinators	y=-1, n=0	-1	Sakai, 2001
6.06	Reproduction by vegetative fragmentation	y=1, n=-1	-1	
6.07	Minimum generative time (years) 1 year = 1, 2 or 3 years = 0, 4+ years = -1			
7	**Dispersal mechanisms**			
7.01	Propagules likely to be dispersed unintentionally	y=1, n=-1	-1	http://www.hear.org/pier/species/castilla_elastica

	(plants growing in heavily trafficked areas)			
7.02	Propagules dispersed intentionally by people	y=1, n=-1	1	http://www.hear.org/pier/species/castilla elastica
7.03	Propagules likely to disperse as a produce contaminant	y=1, n=-1	-1	
7.04	Propagules adapted to wind dispersal	y=1, n=-1	-1	http://www.hear.org/pier/species/castilla elastica
7.05	Propagules water dispersed	y=1, n=-1	1	Personal observation
7.06	Propagules bird dispersed	y=1, n=-1	1	Wayne *et al.*, 2004; Personal observation
7.07	Propagules dispersed by other animals (externally)	y=1, n=-1	1	
7.08	Propagules survive passage through the gut	y=1, n=-1	1	Wayne *et al.*, 2004; Personal observation
8	**Persistance attributes**			
8.01	Prolific seed production (>1000/m2)	y=1, n=-1	-1	Wayne *et al.*, 2004; Sautu *et al.*, 2006
8.02	Evidence that a persistent propagule bank is formed (>1 yr)	y=1, n=-1	-1	Sautu *et al.*, 2006
8.03	Well controlled by herbicides	y=1, n=1		
8.04	Tolerates, or benefits from, mutilation, cultivation, or fire	y=1, n=-1	-1	
8.05	Effective natural enemies present locally (e.g. introduced biocontrol agents)	y=-1, n=1	1	
		Total score	**11**	
		Outcome	**Reject**	

Appendix 2: A multiplier table for climate match questions

Quality(question 2.02)	Climate match (question 2.01) 0	1	2
0	**2**	**2**	**2**
1	**1**	**1.5**	**2**
2	**0.5**	**1**	**2**

Appendix 3: Decision rules for species with Weed Risk Assessment (WRA) scores between one and six



Appendix 4: Criteria for alien plant screening system with minor modifications for use in Eastern Arc Mountains

S/N	Questions	Definitions and Criteria
1.01	Is the species highly domesticated?	Domesticated = selected and bred by humans so that the plant now differs substantially from its wild relatives. Typically, this means at least 20 generations of human selection. If it is a cultivar selected by humans for growth in a cultivated environment, and this selection has resulted in a "handicap" that is expected to limit the species' survival in the wild, then answer may be "yes" even if human selection has been <20 generations.
1.02	Has the species become naturalized where grown?	Naturalized = species has become well established (widespread/and OR common, and self-maintaining) in the wild flora of a new area/region. The following descriptions are NOT sufficient evident of naturalization: "occasional escape from cultivation", "casual", "apparently becoming naturalized around town X", "sparingly naturalized around town Y", "waif" etc. Almost all introduced species can occasionally be found in the wild but this does not indicate naturalization.
1.03	Does the species have weedy races?	There may be substantial variation within a species, designated as races, sub-species, varieties, or cultivars; Are any of these races known to be weedy?
2.01	Species suited to tropical or subtropical climate(s) (0-low; 1-intermediate; 2-high) – If island is primarily wet habitat, then substitute "wet tropical" for "tropical or subtropical"	2 = high =native to tropical and subtropical regions. 1 = intermediate = not native to tropical or subtropical regions but is grown successfully in these regions. 0 = low = not native to tropical and subtropical habitats and not know to be grown in these regions. NOTE: For the special case of a temperate species whose seeds have been reported to require cold-stratification for germination, the answer to this question is 0 (low) and the answer to question 2.02 is 1 (intermediate) regardless of knowledge of the species' native range.
2.02	Quality of climate match data (0-low; 1-intermediate; 2-high) see appendix 2	High = 2 = native range is well known. If there are regions outside the native range where the plant is currently grown, these regions are well known. Intermediate = 1 = boundaries of native range are not well known. Or, the plant is grown outside native range but the region where it is grown is not known, or the plant has a native range that only marginally reaches sub-tropical climates. Low = 0 = the native

		range is highly questionable.
2.03	Broad climate suitability (environmental versatility)	Wide latitudinal range (growing in 2 or more biomes or 2 Holdridge life zones) OR commonly found across an altitudinal range >>1000 m.
2.04	Native or naturalized in regions with tropical or subtropical climates	
2.05	Does the species have a history of repeated introductions outside its natural range? y=-2	"Introduction" does not necessarily mean naturalized. Has the species been repeatedly grown (e.g. cultivated) or accidentally introduced to several regions that are clearly outside its native range? "Repeated introductions" means records from at least 3 separate places. If introduction to a place was limited to a botanical garden (ie the species was not distributed to the public) it is not counted. For species that are widely cultivated or widely grown in horticulture, the answer is usually "yes"
3.01	Naturalized beyond native range y = 1*multiplier (see Append 2), n= question 2.05	Must be clearly outside of native range. Criteria for "naturalization" are given under 1.02
3.02	Garden/amenity/disturbance weedy = 1*multiplier (see Append 2)	Yes = found in areas highly modified by humans but not covered in questions below (e.g. roadsides, gardens). Must be subject to control in these areas, or has been clearly reported as a nuisance. Occasionally reports of occurrence along roadsides is not sufficient evidence to answer "yes"
3.03	Agricultural/forestry/horticultural weed y =2*multiplier (see Append 2)	Yes = Must be subject to control in agriculture. horticulture or forestry, at least occasionally. or evidence that the weed reduces yields. or the species is listed in weed control manuals.
6.03	Hybridizes naturally	Yes - only if direct reference refers to hybridization UNDER NATURAL CONDITIONS, other wise answer "Don't know" (unless there is good reason to believe that hybrids are NOT formed.
6.04	Self-compatible or apomictic	Leave blank if no information. Yes or No only on the basis of direct reference.
6.05	Requires specialist pollinators	Yes only if pollinator is specialized Eg: bats. humming bird. General insect pollinators like bees = No. If no studies on pollinators are available, consider the shape of the flowers /corolla tube.
6.06	Reproduction by vegetative fragmentation	Yes only if it spreads without seeds -- includes rhizomes, stolons, root fragments, suckers, and these vegetative parts commonly survive without human cultivation.
6.07	Minimum generative time (years) 1 year	If plant can reproduce vegetatively, it is the

	= 1, 2 or 3 years = 0, 4+ years = -1	time needed to do this (may be shorter than time to seed production). If no direct reference available then contact people in the nursery business or in forestry and cite as personal communication.
7.01	Propagules likely to be dispersed unintentionally (plants growing in heavily trafficked areas)	Consider the dispersal structure (or small seeds in muddy habitats) AND location where plants grow (heavily trafficked areas such as trail sides, road sides, farms). Does it have hooks or is it sticky? - any method by which it could get attached and hence be dispersed unintentionally by people.
7.02	Propagules dispersed intentionally by people	Does it have any useful properties which might be a good reason for people to move it around? Horticultural, edible fruit, etc.
7.03	Propagules likely to disperse as a produce contaminant	Produce' = any economic product (agriculture, forestry, horticulture)
7.04	Propagules adapted to wind dispersal	If no direct reference then answer based on the morphology of the seed/fruit. For example presence of pappus. Includes tumbling plants.
7.05	Propagules water dispersed	Any reproductive structure is known to be buoyant (e.g. pods), AND the structure is reasonably likely to come in contact with moving water.
7.06	Propagules bird dispersed	If no direct reference then consider the type of fruit. For example small berries then probably - Yes
7.07	Propagules dispersed by other animals (externally)	Seeds attach to animals, or carried but not swallowed
7.08	Propagules survive passage through the gut	If fruits have adaptations for bird dispersal, then assumed to be yes.
8.01	Prolific seed production (>1000/m2)	Greater than 1000/m2 viable seeds produced during one event/ season of reproduction. If total per plant is given, estimate by dividing by the crown area of an average plant. Estimates can also be made based on seed size and type or fruit/number of seeds per fruit.
8.02	Evidence that a persistent propagule bank is formed (>1 yr)	Look for evidence of survival > 1 year IN THE FIELD (not lab storage). Usually hard seeded legume seeds remain viable in the soil for >> 1 year. More than 1% of seeds must consistently remain viable > 1 year under field conditions. Seeds with low viability after 1 year of lab storage are unlikely to meet this criterion.
8.03	Well controlled by herbicides	Is there documented evidence that herbicides have been successfully used to control the plant?
8.04	Tolerates, or benefits from, mutilation, cultivation, or fire	Mutilation includes logging/ lopping, vigorous regrowth from cut stumps; cultivation = ploughing the soil to cultivate the soil.

8.05	Effective natural enemies present locally (e.g. introduced biocontrol agents)	Answer "yes" if an effective enemy (e.g. one known from another country) is present. If the plant is already present and is successfully forming thickets with little evidence of natural enemies, answer "no". Otherwise (for most cases), the answer is "don't know"

Appendix 5: Relative densities (stems/ha) of the tree species in the study area

S/N	Species	Family	Status	Distance from the forest edge (m)					
				20	70	120	170	220	270
1	*Castilla elastica* Cerv.	Moraceae	Exotic	95.9	135.9	89.7	37.7	18.9	9.5
2	*Cedrela odorata* L.	Meliaceae	Exotic	95.9	67.9	8.2	30.2	28.4	0
3	*Funtumia Africana* Stapf.	Apocynaceae	Native	54.8	67.9	130.5	143.4	85.1	94.6
4	*Cephalosphaera usambarensis* Warb.	Myristicaceae	Native	34.3	48.5	57.1	60.4	132.4	132.5
5	*Artocarpus heterophylla* Lam.	Moraceae	Native	34.3	9.7	0	0	0	0
6	*Maesopsis eminii* Engl.	Rhamnaceae	Exotic	20.6	19.4	32.6	15.1	18.9	18.9
7	*Albizia gummifera* C.A. Sm.	Mimosaceae	Native	13.7	0	0	0	0	0
8	*Albizia chinensis* Durazz.	Mimosaceae	Exotic	13.7	0	0	0	0	0
9	*Antiaris toxicaria* Lesch.	Moraceae	Native	6.9	29.1	32.6	7.5	9.5	37.9
10	*Milicia excelsa* welw.	Moraceae	Native	6.9	29.1	16.3	15.1	28.4	18.9
11	*Celtis mildbraedii* Engl.	Ulmaceae	Native	6.9	19.4	8.2	7.5	47.3	18.9
12	*Synsepalum ceraciferum* L.	Sapotaceae	Native	6.9	19.4	8.2	0	9.5	9.5
13	*Celtis gomphophylla* Bak.	Ulmaceae	Native	6.9	19.4	0	0	0	9.5
14	*Quassia undulata* D. Dietr.	Simaroubaceae	Exotic	6.9	0	0	0	0	0
15	*Ficus sycomorus* L.	Moraceae	Native	6.9	0	0	0	0	0
16	*Ficus sur* Forssk.	Moraceae	Native	6.9	0	0	0	0	0
17	*Tabernaemontana sp.* L	Apocynaceae	Native	6.9	0	0	0	0	0
18	*Spondias lutea* L.	Anacardiaceae	Exotic	6.9	0	0	0	0	0
19	*Blighia unijugata* Koen.	Sapindaceae	Native	6.9	0	0	0	0	9.5
20	*Leptonychia usambarensis* Turcz.	Sterculiaceae	Native	0	38.8	32.6	83.0	123.0	123.0
21	*Croton sylvaticus* Hochst.	Euphorbiaceae	Native	0	19.4	16.3	0	0	9.5

22	*Sorindeia madagascariensis* DC.	Anacardiaceae	Native	0	19.4	8.2	7.5	0	0
23	*Bombax rhodognaphalon* K. Schum.	Bombacaceae	Native	0	19.4	0	0	0	0
24	*Rothmannia manganjae* Keay.	Rubiaceae	Native	0	9.7	24.5	0	0	0
25	*Trilepisium madagascariense* DC.	Moraceae	Native	0	9.7	8.2	7.5	18.9	9.5
26	*Ficus exasperate* Vahl.	Moraceae	Native	0	9.7	0	0	0	0
27	*Cola scheffleri* K. Schum.	Sterculiaceae	Native	0	9.7	0	0	0	0
28	*Pouteria alinifolia* Aulb.	Sapotaceae	Native	0	9.7	0	0	0	9.5
29	*Mimusops balata* L.	Euphorbiaceae	Exotic	0	9.7	0	0	0	0
30	*Ricinodendron heudelotii* Heckel	Euphorbiaceae	Native	0	0	16.3	0	0	0
31	*Synsepalum msolo* L.	Sapotaceae	Native	0	0	16.3	15.1	9.5	47.3
32	*Sapium ellipticum* Pax.	Euphorbiaceae	Native	0	0	8.2	15.1	0	0
33	*Khaya anthotheca* welw.	Meliaceae	Native	0	0	8.2	0	0	0
34	*Trichilia emetica* Vahl.	Meliaceae	Native	0	0	0	7.5	0	0
35	*Lanea welwitschii* Engl.	Anacardiaceae	Native	0	0	0	7.5	0	0
36	*Grewia goetzeana* K. Schum.	Tiliaceae	Native	0	0	0	7.5	0	0
37	*Electrophleum suavelonce* Benth.	Mimosaceae	Native	0	0	0	15.1	0	0
38	*Macaranga capensis* Sim.	Euphorbiaceae	Native	0	0	0	0	9.5	0
39	*Albizia glaberrima* Benth.	Mimosaceae	Native	0	0	0	0	9.5	0
40	*Antidesma membranaceum* Muell. Arg.	Euphorbiaceae	Native	0	0	0	0	9.5	0
41	*Anglocalix brownie* L.	Sterculiaceae	Native	0	0	0	0	9.5	0
42	*Anikia kummariae* K. Schum	Annonaceae	Native	0	0	0	0	0	9.5

Appendix 6: Index of Density Dominance (IDD) of tree species in the study area

				Distance from the forest edge (m)					
S/N	Species	Family	Status	20	70	120	170	220	270
1	*Castilla elastica* Cerv.	Moraceae	Exotic	0.28	0.28	0.21	0.08	0.03	0.02
2	*Cedrela odorata* L.	Meliaceae	Exotic	0.28	0.12	0.02	0.07	0.05	0
3	*Funtumia Africana* Stapf.	Apocynaceae	Native	0.14	0.12	0.33	0.42	0.18	0.20
4	*Cephalosphaera usambarensis* Warb.	Myristicaceae	Native	0.08	0.08	0.12	0.14	0.30	0.30
5	*Artocarpus heterophylla* Lam.	Moraceae	Native	0.08	0.02	0	0	0	0
6	*Maesopsis eminii* Engl.	Rhamnaceae	Exotic	0.05	0.03	0.07	0.03	0.03	0.03
7	*Albizia gummifera* C.A. Sm.	Mimosaceae	Native	0.03	0	0	0	0	0
8	*Albizia chinensis* Durazz.	Mimosaceae	Exotic	0.03	0	0	0	0	0
9	*Antiaris toxicaria* Lesch.	Moraceae	Native	0.02	0.05	0.07	0.02	0.02	0.07
10	*Milicia excelsa* welw.	Moraceae	Native	0.02	0.05	0.03	0.03	0.05	0.03
11	*Celtis mildbraedii* Engl.	Ulmaceae	Native	0.02	0.03	0.02	0.02	0.09	0.03
12	*Synsepalum ceraciferum*	Sapotaceae	Native	0.02	0.03	0.02	0	0.02	0.02
13	*Celtis gomphophylla* Bak.	Ulmaceae	Native	0.02	0.03	0	0	0	0.02
14	*Quassia undulata* D. Dietr.	Simaroubaceae	Exotic	0.02	0	0	0	0	0
15	*Ficus sycomorus* L.	Moraceae	Native	0.02	0	0	0	0	0
16	*Ficus sur* Forssk.	Moraceae	Native	0.02	0	0	0	0	0
17	*Tabernaemontana sp* L.	Apocynaceae	Native	0.02	0	0	0	0	0
18	*Spondias lutea* L.	Anacardiaceae	Exotic	0.02	0	0	0	0	0
19	*Blighia unijugata* Koen.	Sapindaceae	Native	0.02	0	0	0	0	0.02
20	*Leptonychia usambarensis* Turcz.	Sterculiaceae	Native	0	0.07	0.07	0.21	0.28	0.28
21	*Croton sylvaticus* Hochst.	Euphorbiaceae	Native	0	0.03	0.03	0	0	0.02

22	*Sorindeia madagascariensis* DC.	Anacardiaceae	Native	0	0.03	0.02	0.02	0	0
23	*Bombax rhodognaphalon* K. Schum.	Bombacaceae	Native	0	0.03	0	0	0	0
24	*Rothmannia manganjae* Keay.	Rubiaceae	Native	0	0.02	0.05	0	0	0
25	*Trilepisium madagascariense* DC.	Moraceae	Native	0	0.02	0.02	0.02	0.03	0.02
26	*Ficus exasperate* Vahl.	Moraceae	Native	0	0.02	0	0	0	0
27	*Cola scheffleri* K. Schum.	Sterculiaceae	Native	0	0.02	0	0	0	0
28	*Pouteria alinifolia* Aulb.	Sapotaceae	Native	0	0.02	0	0	0	0.02
29	*Mimusops balata* L.	Euphorbiaceae	Exotic	0	0.02	0	0	0	0
30	*Ricinodendron heudelotii* Heckel	Euphorbiaceae	Native	0	0	0.03	0	0	0
31	*Synsepalum msolo* L.	Sapotaceae	Native	0	0	0.03	0.03	0.02	0.09
32	*Sapium ellipticum* Pax.	Euphorbiaceae	Native	0	0	0.02	0.03	0	0
33	*Khaya anthotheca* welw.	Meliaceae	Native	0	0	0.02	0	0	0
34	*Trichilia emetica* Vahl.	Meliaceae	Native	0	0	0	0.02	0	0
35	*Lanea welwitschii* Engl.	Anacardiaceae	Native	0	0	0	0.02	0	0
36	*Grewia goetzeana* K. Schum.	Tiliaceae	Native	0	0	0	0.02	0	0
37	*Electrophleum suaveionce* Benth.	Mimosaceae	Native	0	0	0	0.03	0	0
38	*Macaranga capensis* Sim.	Euphorbiaceae	Native	0	0	0	0	0.02	0
39	*Albizia glaberrima* Benth.	Mimosaceae	Native	0	0	0	0	0.02	0
40	*Antidesma membranaceum* Muell. Arg.	Euphorbiaceae	Native	0	0	0	0	0.02	0
41	*Anglocalix brownie* L.	Sterculiaceae	Native	0	0	0	0	0.02	0
42	*Anikia kummariae* K. Schum.	Annonaceae	Native	0	0	0	0	0	0.02

Appendix 7: Relative basal areas (m^2/ha) of the tree species in the study area

				Distance from the forest edge (m)					
S/N	Species	Family	Status	20	70	120	170	220	270
1	*Cedrela odorata* L.	Meliaceae	Exotic	8.7	5.1	0.3	3.0	1.3	0
2	*Cephalosphaera usambarensis* Warb.	Myristicaceae	Native	5.7	3.4	3.5	5.3	6.9	7.7
3	*Ficus sycomorus* L.	Moraceae	Native	5.6	0	0	0	0	0
4	*Albizia gummifera* C.A. Sm.	Mimosaceae	Native	4.9	0	0	0	0	0
5	*Castilla elastica* Cerv.	Moraceae	Exotic	3.7	3.9	4.7	1.5	0.2	1.2
6	*Funtumia Africana* Stapf.	Apocynaceae	Native	3.6	3.3	8.7	9.6	6.4	5.4
7	*Artocarpus heterophylla* Lam.	Moraceae	Native	3.0	0.1	0	0	0	0
8	*Antiaris toxicaria* Lesch.	Moraceae	Native	2.8	12.1	2.3	6.0	1.1	3.2
9	*Quassia undulata* D. Dietr.	Simaroubaceae	Exotic	2.8	0	0	0	0	0
10	*Milicia excelsa* welw.	Moraceae	Native	2.3	3.9	1.2	1.0	3.5	1.9
11	*Spondias lutea* L.	Anacardiaceae	Exotic	0.7	0	0	0	0	0
12	*Ficus sur* Forssk.	Moraceae	Native	0.4	0	0	0	0	0
13	*Celtis gomphophylla* Bak.	Ulmaceae	Native	0.3	0.4	0	0	0	0.1
14	*Maesopsis eminii* Engl.	Rhamnaceae	Exotic	0.3	4.7	3.5	0.8	0.4	1.5
15	*Albizia chinensis* Durazz.	Mimosaceae	Exotic	0.3	0	0	0	0	0
16	*Blighia unijugata* Koen.	Sapindaceae	Native	0.2	0	0	0	0	0.3
17	*Synsepalum ceraciferum* L.	Sapotaceae	Native	0.1	0.7	0.5	0	4.2	2.0
18	*Celtis mildbraedii* Engl.	Ulmaceae	Native	0.1	0.5	0.3	1.6	4.0	1.0
19	*Tabernaemontana sp.* L	Apocynaceae	Native	0.1	0	0	0	0	0
20	*Bombax rhodognaphalon* K. Schum.	Bombacaceae	Native	0	8.8	0	0	0	0
21	*Croton sylvaticus* Hochst.	Euphorbiaceae	Native	0	5.1	2.8	0	0	3.5

22	*Sorindeia madagascariensis* DC.	Anacardiaceae	Native	0	0.9	0.2	0.1	0	0
23	*Leptonychia usambarensis* Turcz	Sterculiaceae	Native	0	0.6	0.5	1.1	2.3	3.2
24	*Trilepisium madagascariense* DC.	Moraceae	Native	0	0.3	0.3	0.1	0.5	0.5
25	*Mimusops balata* L.	Euphorbiaceae	Exotic	0	0.3	0	0	0	0
26	*Ficus exasperate* Vahl.	Moraceae	Native	0	0.3	0	0	0	0
27	*Pouteria alinifolia* Aulb.	Sapotaceae	Native	0	0.1	0	0	0	0.2
28	*Cola scheffleri* K. Schum.	Sterculiaceae	Native	0	0.1	0	0	0	0
29	*Rothmannia manganjae* Keay.	Rubiaceae	Native	0	0.1	0.5	0	0	0
30	*Ricinodendron heudelotii* Heckel	Euphorbiaceae	Native	0	0	6.4	0	0	0
31	*Synsepalum msolo* L.	Sapotaceae	Native	0	0	4.0	1.9	0.8	4.5
32	*Sapium ellipticum* Pax.	Euphorbiaceae	Native	0	0	1.7	2.8	0	0
33	*Khaya anthotheca* welw.	Meliaceae	Native	0	0	0.1	0	0	0
34	*Trichilia emetica* Vahl.	Meliaceae	Native	0	0	0	4.8	0	0
35	*Lanea welwitschii* Engl.	Anacardiaceae	Native	0	0	0	0.1	0	0
36	*Grewia goetzeana* K. Schum.	Tiliaceae	Native	0	0	0	0.4	0	0
37	*Electrophleum suaveionce* Benth.	Mimosaceae	Native	0	0	0	3.0	0	0
38	*Macaranga capensis* Sim.	Euphorbiaceae	Native	0	0	0	0	2.5	0
39	*Albizia glaberrima* Benth.	Mimosaceae	Native	0	0	0	0	6.6	0
40	*Antidesma membranaceum* Muell. Arg.	Euphorbiaceae	Native	0	0	0	0	0.4	0
41	*Angloculix brownii* L.	Sterculiaceae	Native	0	0	0	0	0.1	0
42	*Anikia kummariae* K. Schum.	Annonaceae	Native	0	0	0	0	0	1.3

Appendix 8: The Chi-square table for the association between levels of disturbance and frequency distribution of individuals at different growth stages.

Growth stages	Low disturbance	Medium disturbance	High disturbance	Plantation	**Total**
Seedlings	1979	8541	25416	5416	41352
Saplings	2187	3854	14687	2916	23644
Juveniles	1666	1770	3020	1145	7601
Young trees	18	57	96	32	203
Mature trees	6	28	72	23	129
Total number	5856	14250	43291	9532	72929

$\chi 2$ = 21.02, degree of freedom = 12, *P*- value = 0.001

Appendix 9: Regression analysis for overstorey canopy

Regression table

	Df	SS	MS	F	Significance F
Regression	1	16145.97	16145.97	4.06	0.0599
Residual	17	67601.93	3976.58		
Total	18	83747.89			

Appendix 10: Number of male trees and number of regenerants as observed in the study area.

No of plots	Number of male trees	Number of regenerants
1	0	1534
2	0	1657
3	0	1902
4	2	1698
5	1	1677
6	0	552
7	0	327
8	0	818
9	1	654
10	0	430
11	0	327
12	0	184
13	0	245
14	0	205
15	0	184
16	0	409
17	0	532
18	0	225
19	0	286
20	1	409