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Fragmented forests in South-West Ethiopia : impacts of land-use change on plant species composition and priorities for future conservation

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FRAGMENTED FORESTS IN SOUTH-WEST ETHIOPIA: IMPACTS OF LAND-USE CHANGE ON PLANT SPECIES COMPOSITION AND PRIORITIES FOR FUTURE CONSERVATION

By

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A thesis submitted in candidature for the degree of Philosophiae Doctor at Bangor University, Bangor, UK

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ABSTRACT

There is major concern internationally, and specifically in the Ethiopian afromontane biodiversity hotspot, about the impact of forest habitat fragmentation on biodiversity conservation. This study assessed the effect of land-use change on land-cover pattern, forest patch spatial structure and consequent effects on plant species richness and composition in two areas of southwest Ethiopia: Illubabor and the Gamo highlands of Gamo Gofa. Land-use change analysis was conducted on three sites, I (1500-2000 m altitude) and II and III (1878-2422 m altitude), each of 47,648 ha, in Illubabor and one area of 66,765 ha in the Gamo highlands. Two Landsat images from the years 1986 and 2000 and one Spot image from 2007 covering Illubabor and two Landsat images from 1995 and 2010 covering most of the Gamo highlands study area were analysed. The change in area, number, shape and edge density of patches of each land-cover class were quantified between the years. Field sampling of woody plants was conducted in plots within 30 forest patches in Illubabor stratified between the three sites, and ten in the Gamo highlands stratified between sacred groves and non-sacred forests. Information on local knowledge, cultural association, institutions, practices and threats of the sacred groves was obtained by interviewing 24 of their traditional custodians. The landscape pattern in both Illubabor Zone and Gamo highlands has changed rapidly over 20 years. In highland areas there has been a rapid conversion of forest to farm, settlement and grazing land. At lower altitude forest cover has been retained but it has been degraded by its use for understorey coffee cultivation. The area and number of forest patches has decreased while patch shape and edge density has increased. In Illubabor forests' woody plant species richness was higher in the lower altitude site I (70.8±9.2) than in sites II (50.9±6.3) and III (54.3±4.9), with little difference in the shape of their species accumulation curves. Species composition also differed between site I and sites II and III; altitude and disturbance were more strongly associated with this difference than were patch size, shape and edge density. There was little difference in tree density or basal area amongst the sites.

Within forest patches, tree basal area was higher in the patch interior (96.8±9.4 m^2 ha^{-1}) than in the edge zone (77.2±15.3 m^2 ha^{-1}), however total tree density did not differ significantly. The interior forest had twice the density of trees taller than 22 m and a higher density of small trees (5-14.9 cm DBH) than the edge. Tree species richness did not differ significantly between the two habitats, however including shrubs and vines total woody species richness was higher in the edge (69.3±5.9) than interior (52±3.5) forest. While the upper canopy of interior forest was dominated by species with a wide habitat distribution range, it also had a higher abundance of forest-habitat specialist species than the edge. Species with a distribution associated with forest-margins were, as expected, more abundant in the edge habitat. No association was found between tree density, basal area or height in both habitat types and any fragmentation variables (patch size, shape or edge density) or environmental variables (rainfall, altitude or cumulative disturbance). However, edge habitat basal area was negatively associated with disturbance. Shrub, vine and geophytic angiosperm herb species composition differed between forest edge and interior habitats. Species richness of vascular epiphytes was higher in interior (28.9±1.8) than edge (13.6±1.4) habitat, as was their individual density (114.5±6.5 and 42.7±3.7 respectively), which was associated with the density of large DBH trees. Epiphyte density was not associated with any fragmentation variables, rainfall or altitude, however in the edge habitat it was negatively associated with disturbance. Species composition varied between the two habitats, with forest-habitat distribution species, which tended to have herbaceous stems and leaves, being more abundant in the interior habitat. However, the same three epiphyte species were dominant in both habitats, and species with succulent stems and leaves or woody stems had similar abundance in both. Geophytic fern species richness was higher in interior (29.4±1.8) than edge (22.1±1.4) habitat, as was individual density (104±22.3 and 59.8±13.7 respectively). Species with forest-habitat distribution, those with creeping or erect rhizomes and those with tufted fronds were more abundant in the interior, while generalist distribution species and those with spaced fronds showed no difference.

In the Gamo highlands, sampled woody plant species richness and diversity were higher in sacred groves than in non-sacred forests, however, the species accumulation curve showed no difference between the two categories of forest. Their species composition differed and the sacred groves had a higher proportion of species endemic to Ethiopia (12.5%) than the non-sacred forests (9.2%). Two national conservation priority species, *Cordia africana* and *Hagenia abyssinica*, were only recorded in the sacred groves, and one IUCN red list species, *Prunus africana*, was more abundant there. Mean basal area was significantly higher in the sacred groves (1.55±0.45 m^2 ha^{-1}) than the non-sacred forests (1.28±0.41) as was tree seedling density (1111.7±393.2 ha^{-1} and 476.8±87.3 ha^{-1} respectively) but not sapling density.

In conclusion, forests in southwest Ethiopia are undergoing a high rate of fragmentation and degradation. The resulting loss of forest interior core habitat is associated with a loss of biodiversity, especially of vulnerable forest-specialist species of woody plant, vascular epiphyte and geophytic fern. Nonetheless, the remaining small forest patches do still have high biodiversity value and they should be made a high conservation priority. In the Gamo highlands these small patches are generally sacred sites with high cultural value, however they have recently come under high levels of threat which risks loss of biocultural diversity. Their conservation through strengthening of traditional community institutions is a high priority.

DEDICATION

This thesis is dedicated to my wife Melkam Hailu, my sons Robera Desalegn, Mati Desalegn, my daughters Latera Desalegn and Yadinet Desalegn who shared the feelings of loneliness while I was away for studies.

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CHAPTER 1

GENERAL INTRODUCTION

1.1 Background

A large number of studies have estimated that most of our planet's land surface has been transformed by human development. Expansion of the land area under productive use by the increasing human population has caused increased natural resource exploitation and alteration of landscape pattern (Turner et al., 2001). Such changes in land-use are occurring at a high rate in many developing countries (Laurance et al., 1999) and are recognized as major drivers of environmental change (Feddema et al. 2005). They often result in the severe habitat change of deforestation, often associated with fragmentation, or else degradation of the remaining forest habitat (Turner et al., 2001; Lindenmayer and Fischer, 2006). These land-use changes have created highly fragmented landscape patterns that affect the continuing capacity of ecosystems to maintain ecological processes and functions (FAO, 2003; Hersperger et al., 2003). The details of the forest fragmentation that often accompanies deforestation include the reduction in forest area (area effect), arrangement and connectivity, which can have substantial effects on the capacity of forest ecosystems to maintain their original biodiversity (Herzog and Lausch, 2002; FAO, 2003; Bierwagen, 2007). Populations of forest dependent species are more likely to become extinct in fragmented landscapes due to effects associated with the smaller size, greater isolation and increased amount of edge of forest habitat patches, leading to smaller and more isolated populations (Lindenmayer and Fischer, 2006). The sum of these changes have a negative impact on a variety of ecological process such as tree seed dispersal and cause shifts in the species composition of habitat patches (Turner, 1989; Turner et al., 2001). As a consequence these changes of forest habitats jeopardize the supply of ecological goods and services from forest lands (Forman, 1995; Laurance et al., 1997).

Ethiopia is a typical country in this respect. Natural forest habitats have been reduced to 2-3% of the original forest cover (Teketay, 2000) and the remaining habitats have become degraded and fragmented (with reduced capacity for regeneration) as a result of encroachment by farming, grazing and vegetation trampling as the most important causes (Teketaye, 2000). However, an assessment of the threat status and future management effectiveness of these habitats requires quantitative and objective assessment of land-use and land-cover change impacts on landscape structure, function and biodiversity (Bleher et al., 2006). A possible approach to fulfilling these criteria can be based on the quantification of the properties of land-cover change as a result of land-use expansion (Gustafson, 1998) and investigation of the effect of this change on species composition, diversity and population structure of the remaining fragments.

A brief overview of existing knowledge and methodology for research into land-use and land-cover change, change in landscape spatial pattern, fragmentation and their impacts on natural habitats are therefore given in the following section.

1.1.1 Land-use and land-cover change

Land-use changes have become a global concern because of the negative impact on land cover. Land use and land cover are two related land surface characteristics where land use is the way in which humans exploit the land cover (Riebsame et al., 1994). Land use summarises the human activities occurring on a given area of land, e.g. forestry, agriculture (which can be divided between pasture and cultivated land) and built up areas (Sivrikaya et al, 2007). The changes in these land uses have great influence on land cover. Land cover, on the other hand, refers to the surface cover of the earth. It is the actual physical characteristics of the landscape and the vegetation on it, and maps of this type often distinguish several different types of natural habitat (Turner et al., 1993; Forman, 1995).

Land-use changes are diverse and may be explained in terms of temporal and spatial aspects (William, 2003). They may involve the complete transformation of habitat into other land-use types. For example, transformation may be in the form of replacement of forest by agriculture, grazing or settlement land (Lindenmayer and Fischer, 2006). Land-use changes may also occur in the form of expansion or reduction in which case one or more land uses expand at the expense of other land-use/cover types. Expansion of crop agriculture and settlement, for example, can be at the expense of grazing land or forest which is area gain for agriculture and settlement, but area loss (or reduction) for grazing or forest land.

The fragmentation of remaining of natural forests into patches that then shrink in size and become perforated has long been recognized (Forman, 1995). In addition, however, because of land-use activities around and inside these patches they are under further anthropogenic pressure which leads to greater isolation between patches (as distances increase, and matrix permeability decreases, as well as disturbance within patches leading to degradation (Hanski, 1994; Lindenmayer and Fischer, 2006)). A more detailed account of the concept of forest fragmentation is given in section 1.1.3 below. This fragmentation of large forest habitats into smaller parcels as a result of land-use activities may disrupt the integrity of ecosystem process and function (Hobbs 1993), producing isolated segments of forest habitat and ecosystem surrounded by a matrix of intensively utilized areas which: modifies the ecological interrelationships between forest fragments (Forman, 1995); reduces the space available for some species (Haila, 2002; Fahring, 2003); influences a variety of ecological phenomena, like population and community structure (Turner et al., 1989; Pulliam et al., 1988); changes landscape pattern by altering the relative abundances of natural habitats (Turner et al., 2001; Sivrikaya et al., 2007); and affects soil condition and water quality and water supply (Meyer, 1995).

Land-use and land-cover changes are also a globally important source of emissions of greenhouse gases, whose accumulation in the atmosphere brings about global climate change. The increase in atmospheric CO_2 by 35% in the last 100 years may be attributed to land-use change, principally through deforestation (Meyer, 1995). Since then deforestation and forest degradation have been estimated to contribute up to 17% (in 2004) of global CO_2 emissions (IPCC, 2007). The major source of increased CH_4 and

N_2O emissions are all related to land-use including biomass burning, pastoral agriculture, fertilizer application and landfills (Meyer, 1995). Therefore, research to understand both spatial and temporal changes of land-use and land-cover and their effect on landscape pattern should be given a priority to provide the evidence base to recommend possible solutions to mitigate land-use impacts on land-cover and plan for plausible forms of sustainable management.

1.1.2 Landscape spatial pattern change

The dynamics of species and the decline they face are determined by their population characteristics and distribution, the quality of their habitat, and the changes in these factors in response to human-caused disturbance and various natural processes (Akcakaya et al., 2003). Natural processes, e.g. volcanic eruption and flooding, can produce new fragmented habitats (Lindenmayer and Fischer, 2006). However, habitat fragmentation due to anthropogenic disturbances is by far the most important manifestation of habitat loss (Forman, 1995; Fischer and Lindenmayer, 2002; Wade et al., 2003; Lindenmayer and Fischer, 2006; Farina, 2007). Ecologist use landscape models as a main approach to investigate these dynamic processes. Landscape models may predict changes in landscape spatial pattern (Akcakaya et al., 2003). Landscape spatial pattern can be characterized by both the composition of the habitat types and structure of those habitats (Turner, 1989; McGarigal and Marks, 1994; Turner et al., 2001; Drinnan, 2005; Farina, 2007). These are the two essential features that are required to describe any landscape (Gustafson and Parker, 1992; John et al., 1992).

Landscape composition refers to features associated with the presence and amount of each patch type without regard to how they are spatially arranged (Forman et al., 1986; McGarigal and Marks, 1995; Turner et al., 2001; McGarigal et al., 2004). This is important for many ecological processes, for example many plant and animal species require specific habitat types, and the amount and quality of this habitat therefore influences the occurrence and abundance of these species (Hersperger et al., 2003). For instance, a study on the effect of habitat amount on biodiversity in fragmented traditional orchards by Bailey et al. (2010) indicates that abundance of the weevil

(*Phyllobius oblongus*) increased with landscape-scale habitat amount. Landscape structure refers to features associated with the patch structure, i.e. arrangement of patches, patch shape, size, edge density, fractal dimension and connectivity (Turner et al., 2001). The patchiness of a landscape structure is the result of the interaction of past disturbance and the heterogeneity of the abiotic environment (Gergel and Turner, 2002). The patch structure of a landscape may have a significant effect in determining the structure of a particular community (e.g. Turner, 1989; Laurance et al., 2002; Hill and Curran, 2003; Goparaju and Jha, 2010). Consequently, monitoring change in patch-based measures of spatial pattern is an important way to assess landscape change (Gergel and Turner, 2002).

1.1.3 Fragmentation and its impact on natural habitat

Fragmentation is often defined as a process by which a natural landscape is broken up into small patches, isolated from one another in a matrix of lands dominated by human activities (Forman, 1995; Griffiths et al., 2000; Malcolm et al., 2007). This definition is adopted in the present study. Habitat fragmentation has been identified as one of the most important causes of loss of biodiversity worldwide (Saunders et al., 1991; Fahrig, 2003; Farina, 2007). Recent studies have estimated that over half of the temperate broadleaf and mixed forest biome and one quarter of the tropical rain forest biome have been fragmented or removed by human use (Wade et al., 2003, Farina, 2007).

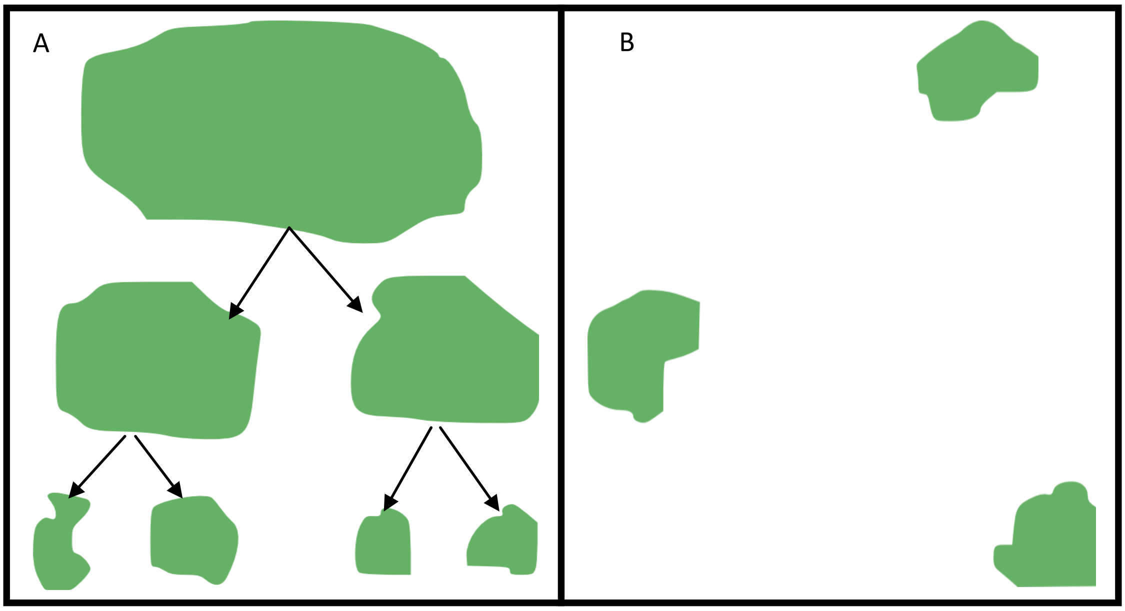


Figure 1.1 The spatial pattern process in forest fragmentation (A) and isolation (B).

The process of habitat fragmentation involves three main factors, each of which have an important effect on plant and animal species:

1) Fragmentation leads to the breaking up of large patches into numerous smaller patches (e.g. Figure 1.1A), resulting in a net habitat loss and change in spatial structure (Forman, 1995; Turner et al., 2001; Gehring and Swihart, 2002, Echeverria etal., 2006). This results in a decrease in the amount of resources and area for shelter available to plant and animal species dependent on this habitat (Haig et al 2000) and therefore leads to a general reduction in the number of individuals that can be hosted (Lindenmayer and Fischer, 2006). For example, Doherty et al. (2002) investigated bird communities living in forest fragments of different size in agricultural landscapes in USA and found that their survival rate was negatively affected by fragment size. They showed that their survival was higher in larger patches. Hillers et al. (2008) studied influence of forest fragmentation on leaf-litter frogs in tropical rainforests of West Africa. They found that species richness and diversity were lower, and species composition was altered, in forest fragments compared with continuous forest. The change in spatial structure and the loss of habitat due to fragmentation could also alter the population structure and size of individual plant species and therefore community diversity, richness and composition

(Saunders et al., 1992; Holt, 1992; Connor et al., 2001; Hill and Curran, 2001). For example, Bastine and Thomas (1999) found that the patch occupancy of 22 plant species increased with habitat age, habitat area and number of habitat. Other findings also indicate that habitat diversity in human-modified landscapes is important in explaining the presence of species groups of high conservation value (Honnay et al., 1998). Hill and Curran (2003) found that forest patch area, shape and isolation accounted for a high proportion of the variability in tree species diversity per patch in Ghana. They conclude that forest area is the most important consideration when planning tropical forest reserves. On the other hand, Honnay et al., (1998) concluded that small forest fragments can also be very important for maintaining regional plant species diversity, at least if they are of high habitat quality and if the forest management is appropriate. Other studies have also shown that a large number of small fragments in modified landscapes can be very important in providing “stepping stones” between larger patches (Lindenmayer and Fischer, 2006) and the presence of a large number of smaller habitat patches can increases connectivity between patches at a landscape scale (Farina, 2007).

Fragmentation also has an impact on forest shape (McGarigal and Marks, 1994). Changes of forest shape due to fragmentation can play a major role in patch species composition. Irregular and rectangular patch shapes are believed to have less core area which reduces the abundance of species dependent on forest interior habitat compared with circular shapes (McGarigal and Marks, 1994). Empirical studies indicate that community composition can also be altered as a result of change in forest shape, e.g. Hill and Curran (2005) who found that irregular-shaped fragments had a higher proportion of regenerating, light-demanding pioneers and animal-dispersed tree species than regular-shaped forest patches.

2) By opening core areas to the effects of exterior habitats, fragmentation of continuous habitat patches may lead to a dramatic increase in edge effect (Sih et al. 2000; Turner et al. 2001; McGarigal et al., 2002; Harper et al., 2005). In highly modified landscapes, many edges can be created by humans. Examples are the boundaries between recently clear-cut forest or other land uses and natural forest stands (Lindenmayer and Fischer, 2006). Edge effect refers to changes in biological and physical conditions that occur at

boundaries and within adjacent areas of affected forest (Laurance et al., 2002; Lindenmayer and Fischer, 2006). According to Harper et al. (2005) edge effect is classified into primary responses that arise directly from edge creation or secondary responses that arise indirectly. Primary responses include structural damage to the vegetation, disruption of the forest floor and soil layer, altered nutrient cycling and decomposition, changed evaporation, and altered pollen and seed dispersal (Harper et al., 2005). Secondary, or indirect responses, include patterns of plant growth, regeneration, reproduction, and mortality and are manifested as altered patterns of vegetation structure and species composition (Hobbs and Yate 2003; Harper et al., 2005). Thus edge effects can influence the distribution and composition of species that inhabit vegetation remnants (Lindenmayer and Fischer, 2006). For example, Lovei et al. (2006) have found that edge-preferring species increased with the edge: area ratio, and they concluded that the much greater abundance of generalist and edge-preferring species at the edge distorts the species-area relationship.

3) Habitat fragmentation results in the geographic isolation of patches among a matrix of, e.g., agricultural crop and grazing land (Bender et al., 2003) - see Figure 1.1B. This might greatly restrict the mobility of plant propagules and less mobile animals (Andreassen et al. 1996), and thereby isolate some populations, especially of species with specialist habitat requirements that are obligately dependent on the habitat type restricted to the isolated patch islands. Small isolated populations can be threatened by inbreeding, which represents a serious problem for their survival (Schmitt and Seitz, 2002). Moreover, small populations are more sensitive to stochastic events, such as epidemic outbreaks, that could drive local populations to extinction (Sutherland, 2000). As the isolation of habitat patches increases, the probability of species recolonizing individual patches, and forming a viable metapopulation across the landscape may decrease (Parker and Nally, 2002).

The equilibrium theory of island biogeography (MacArthur and Wilson, 1967) has subsequently been applied to describe the biodiversity pattern of isolated patches of mainland ecosystems. According to this theory, the species richness of a habitat island is positively correlated with the area of the “island” and negatively affected by the distance from the “mainland”, or another habitat patch which acts as a source of immigrant organisms, e.g. seeds for plants. These are the main factors influencing the immigration of species to patches, the rate of local extinction and recolonization, and the rate of species turnover (Gilpin and Hanski, 1991). However, island biogeography theory alone cannot be used to explain the pattern of biodiversity in landscapes because the heterogeneity of habitat within each patch and human influences surrounding the patch are not considered in this concept (Wagner and Edwards 2001; Laurance, 2008). Island biogeography theory also provides few predictions about how community composition in fragmented patches changes over time and which species are more affected by fragmentation (Laurance, 2008). However, island biogeography theory remains one of the main conceptual frameworks for the development of metapopulation theory (Hanski 1998) which is now very influential in landscape-scale conservation biology. A metapopulation consists of several spatially separated sub-populations that are related through the dispersal of individuals between them (Forman, 1995; Hanski 1998; Farina, 2007). For example, in a matrix around a given islet, many species can exist as sub-populations in scattered habitat patches that are interrelated due to dispersal of individuals (e.g. as seeds or pollen). Consequently the survival of a whole population depends upon the relationship between extinction and recolonization rate among these habitats patches (Hanski 1998). For instance, Mennechez et al. (2003) investigated the effect of habitat loss and fragmentation on population functioning of butterflies by comparing demography (daily and total population size) and dispersal (dispersal rate and dispersal kernel) and their results suggest that habitat loss and fragmentation affect dispersal more than demography.

The source – sink model is one version of the metapopulation model. In this model, one viable sub-population has a positive growth rate and therefore acts as a source of propagules to other sub-populations – sinks (Pulliam 1996, Farina, 2007). This model is applicable if the distance and connectivity between sub-populations enables the dispersal from one habitat patch to another (from source to sink). However, several studies have shown that too great a distance between habitat patches is a main reason for poor colonization of patches in fragmented landscapes (e.g. Benitez-Malvido, 1998; Bennett, 2003; Petit et al., 2004; Laurance, 2008). As a result, in such a highly fragmented landscape, species composition, diversity and abundance patterns may be much more dependent on the existing seed bank within the patch, rather than long-distance dispersal from other patches (Piessens et al., 2005). If recruitment from the source seed is limited to the source habitat patch itself, the population may face local extinction due to demographic stochasticity and inbreeding depression (Sutherland, 2000). This type of crisis could be minimized by conserving all the remaining patches in sufficiently close proximity for dispersal within a fragmented landscape or, if necessary and feasible, increasing their number (see section 1.1.4 below). Therefore, quantification of the number of patches and their spatial structure and how this landscape pattern changes over time is important information for assessing the potential of the existing biota to survive landscape change, a crucial issue for conservation (Lindenmayer and Fischer, 2006, Rey-Benayas et al., 2007). In highly human-modified landscapes such as in south-west Ethiopia, it is expected that forest habitat patches are decreasing in size and number, and changing in shape, which is expected to have impacts on species richness, composition and distribution. In order to understand and conserve the biodiversity of forests in south-west Ethiopia, it is essential to identify the location and distribution patterns of these natural habitats and how this is linked to their species diversity, composition and structure.

1.1.4 Recommended approaches to reduce fragmentation effects

A number of authors (e.g. Honnay et al., 2002; Donald and Evans, 2006; Cortina et al., 2006; Farina, 2007, Bullock et al., 2011) have reported various options for reducing the fragmentation of forests. Patch area, patch number and connectivity between patches are the landscape structural characteristics that are most influenced by fragmentation (Forman, 1995). These are the principal conservation parameters that should be modified in order to minimize the negative effects caused by fragmentation (Lindenmayer and Fischer, 2006). The means most often discussed to achieve better modification of these conservation parameters include restoration of degraded habitat to increase the recovery of those ecosystem attributes and functional process that have been lost or reduced with degradation (Donald and Evans, 2006, Rey-Benayas et al., 2008, 2009; Bullock et al., 2011), or reducing patch isolation by establishment of stepping stones in the form of new woodland patches to increase the heterogeneity of landscape and habitat richness (McGarigal and Marks, 1994; Rey-Benayas et al., 2008), establishment of habitat corridors or increasing matrix woodland composition (Farina, 2007). In other words forest restoration is a key practice to improve biodiversity in altered landscapes, while the maintenance and creation of woody patches is fundamental to conserve species where dispersal is limited and large patches are required (Lindenmayer and Fischer, 2006). Another major focus for landscape ecology-based habitat conservation is the interaction at the edge between patch and matrix. Gascon et al. (1999) indicated that diversification of a matrix with different tree species can reduce pressure from this edge effect.

In human-dominated landscapes, before undertaking these conservation measures, it is likely that landscape pattern needs to be quantified since most processes on a landscape level are influenced or even controlled by habitat spatial pattern (Maestre et al., 2006). For long-term sustainability of the natural habitats in a modified landscape, the links between landscape pattern and ecological process and human activities also need to be clarified. For example, multiple conservation questions should be addressed such as: how landscape has changed through time and direction of change; determining whether the patterns of two or more landscapes differ from one another, and in what ways (Gergel and Turner, 2002). This form of analysis enables evaluation of alternative land management strategies in terms of landscape pattern. This can provide valuable evidence for recommendations of which structural characteristics of each patch should be maximized, and where new forest patches should be created (Gergel and Turner, 2002). Underpinning this is the understanding provided of the interactions between landscape spatial pattern and process at multiple scales: landscape, class and patch (Turner et al. 2001). For this, quantification of landscape pattern is key (McGarigal et al., 2002). Then, investigation of the origin and internal dynamics, e.g. changes in species composition due to change in landscape structure caused by fragmentation, may give important insights into the function and conservation potential of a given patch. Summation of this information can then be scaled up to the whole landscape.

1.1.5 Forest fragmentation in Ethiopia

Ethiopia accounts for the largest part of the "Eastern Afromontane" global biodiversity hotspot (Myers et al. 2000). It is a country with an estimated total area of 1.12 million km^2 with a diversity of ecosystems (CSA, 2007). The massive highlands with high and rugged mountains are divided by the Great Rift Valley which runs from northeast to southwest (Figure 1.2) giving a variety of climate, soils and vegetation types (Tadesse, 2003). The highlands on each side of the Rift Valley are categorized into the central lava highlands, and the massifs of the northern plateau and southeastern highlands (Woubeshet, 2001). The highlands form catchments for rivers in the extensive arid and semi-arid lowlands to the east, south and west of the country (Figure 1.2).

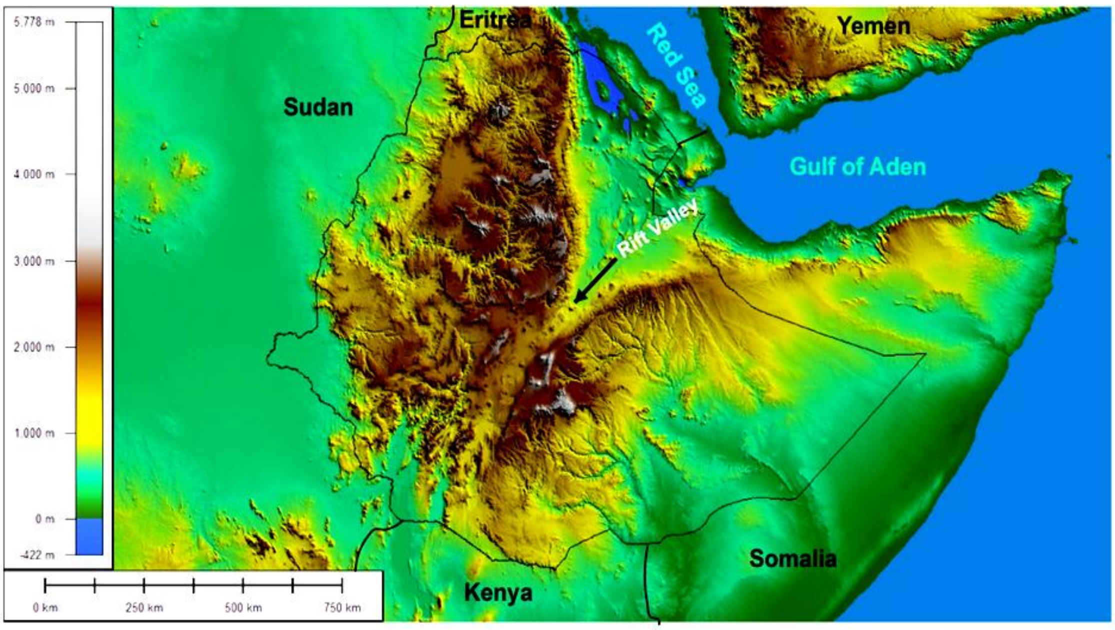


Figure 1.2 The highlands of Ethiopia divided into southeastern and northwestern blocks by the Rift Valley.

The diversity of ecosystems and the biogeographical isolation of the Ethiopian highlands due to the arid and semi-arid lowlands lead to their unique and diverse biological diversity (McKee, 2007). However, clearing of forests for agriculture (both subsistence and commercial plantations), cattle grazing, establishment of settlement areas represent a significant threat to the natural forests (Teketay, 2000). According to various sources (e.g. Logan, 1946; FAO, 1981; IUCN, 1991; EFAP, 1994), 40% of the country was covered by natural high forests up to the early 1900s. These natural forest habitats have undergone considerable loss of area and fragmentation through a long history of human activities (Teketay, 2000). Evidence for this is provided, for example, by change detection analyses based on satellite images of 1976 which indicate that natural high forest had been reduced to 6.08% of land area (FAO, 1978). Analysis of Landsat images 14 years later (1986 to 1990) showed that Ethiopia's forest cover had reduced even further to 3.9 % (Ministry of Water Resources, 1997; Reusing, 1998). By that time the only remnant natural forests were in church and monastery yards (Teketay, 2000; Binggeli et al., 2003; Alemayehu and Teketaye, 2006; Aerts, 2007), numerous traditional sacred forests (Desalegn, 2007), forests in inaccessible areas (Feoli et al., 2002) and blocks of afromontane rain forests in the southwest and southeast of the country (Reusing, 1998; WBISPP, 2000).

In the central and northern highlands of Ethiopia natural forest vegetation is now almost exclusively limited to church and monastery compounds (Aerts et al., 2006; Alemayehu and Teketaye, 2006). However, landscapes of the (Illubabor, GamoGofa, Kaffa and Sheka and southeast (Bale Mountain) highlands still contain considerable forest areas that exist in government protected areas, community forests and in the form of sacred groves, but these are being removed at alarming high rate estimated at 8% per year (FAO, 2003, Reichhuber et al., 2012). This loss of Ethiopian forest is of global importance since the Eastern Afromontane biome is one of the world's recognized biodiversity hotspots (Myers et al., 2000; Tadesse, 2003). Ecologically, these forests are home to many afromontane endemic species, species endemic to Ethiopia and other forest-specialist species (Tadesse, 2003). The forests of the south-west highlands are also the natural habitat of coffee (*Coffea arabica L.*) and support a unique pool of its genetic diversity (Fayera and Denich, 2006; Tadesse et al., 2008). Wild coffee populations in these forests serve as a source gene pool for coffee that is under cultivation (Tadesse et al. 2002). However, knowledge of the unique biodiversity of these forest ecosystems has not halted their increasing deforestation, fragmentation and habitat loss (Tadesse et al., 2008).

The dominant land-use system in this area used to be smallholder farming with agricultural production in small parcels for subsistence purposes which had a limited effect on natural forests (Mengistu, 2005). The communities used traditional farm tools that did not have a great impact on soils or extensive forest areas. Gathering of NTFPs from forests constituted a major component of their livelihoods (Tadesse and Nigatu, 1996). According to Desalegn and Yilma (2008), in 2008 90% of communities living in and around the remaining Bonga forest, which is part of south-west afromontane forests, were still engaged in the harvesting and production of NTFPs, including wild and semi-wild coffee, honey, spices, mushrooms, bamboo, wild fruits and tubers.

Since 1984 rapid expansion of the land area under agriculture has occurred in the Ethiopian south-west highlands due to the influx of people from the central and northern parts of the country due to both planned government resettlement programmes and spontaneous migration (Reusing, 1998, 2000; Tadesse et al., 2001; Mekuria, 2005; Mengistu, 2005). This has resulted in the loss of a significant amount of forest cover and fragmentation (Tadesse, 2003). The area was selected by the national government for resettlement without sufficient consideration of the potential impact on the natural ecosystem, people's livelihoods or the cultural practices of the indigenous population (Mengistu, 2005; McKee, 2007). Consequently, the natural forest habitat, and traditional livelihoods and management systems that were compatible with the environment were all negatively affected by the new settlers (Mengistu, 2005). The settlers have used slash-and-burn cultivation systems, carried out extensive drainage of wetlands and converted large areas of forest and wetlands to permanent or rotational agricultural land for food production. This has permanently changed the landscape pattern and ecological process (Bognetteau et al., 2003, 2007). The settlers also introduced large numbers of cattle, in addition to those already owned by the indigenous people. These cattle are largely for domestic use. Their grazing occupies large areas of marginal land and converted or degraded forest. The natural regeneration of forest tree species has been markedly reduced by the high intensity of grazing within many forests (Reusing, 1998).

A more recent major threat to forest ecosystems in south-west Ethiopia is commercial agriculture. Major external investment has been targeted in particular at the natural forest blocks with coffee growing in the understorey in the suiTable climate/microclimate conditions of temperature and rainfall. This medium-- to large-scale commercial farming has increased greatly in forest areas during the past decade (Mekuria, 2005; Tadesse and Masresha, 2007; McKee, 2007; Hiywot et al., 2012) both degrading and altering the spatial pattern of forest habitats. For example, two-thirds of the primary rainforest which covered 60% of Kaffa Zone in 1973 has been cleared to give way to cash crop plantations of coffee and tea, so that it covered only 20% of the zone in 2005 just 32 years later (Tadesse and Masresha, 2007; Bedru, 2007). The rapidly dwindling forest remnants remain important to local communities for harvesting of understorey coffee and other non-timber forest products (Tadesse and Nigatu, 1996). As a result, there are frequent disputes over this scarce forest land resource between outside investors and local communities and among local communities themselves (Wood, 1993; personal observation). The continuing conversion of these forests is increasing habitat fragmentation; together with resulting over-exploitation of forest products and other forms of forest degradation this is expected to decrease the species richness of the remaining patches.

The issue of land ownership, tenure and rights to access forest resources by local communities are major contributors to forest habitat loss and fragmentation in south west Ethiopia (Mekuria, 2005). Land and its resources, in Ethiopia in general, are state-owned property while local communities have only user rights (Abegaze, 2004). As forest use-rights are restricted to non-timber forest products, communities in south-west Ethiopia are not currently able to exercise their traditional full tenure rights. In some areas, the forests are largely considered to be state forests. As such they are subject to high rates of deforestation and timber looting because there is no strong state control and this is causing a major change in landscape pattern (Tadesse, 2007; personal observation). The regional and national governments have been unable to foster viable partnerships with all communities in the protection of the region’s forests.

The consequences of this deforestation and forest degradation in south-west Ethiopia for biodiversity conservation are not adequately documented as the area lacks adequate landscape-level ecological and floristic data, and there has been no analysis of the linkage between landscape pattern and floristic composition of these forests. This lack of evidence is constraining the prioritization of these habitats in conservation policy. One exception is that a small fraction of the forests in Illubabor Zone in Ethiopia have recently (in 2008) been designated as a Biosphere Reserve under UNESCO's World Heritage Programme (Tadesse et al., 2008). In addition, a small number of forest blocks (Masha Anderecha and the forests of the Bonga area) have also been designated as protected areas by the national government in an attempt to protect them from mass deforestation and fragmentation (Schmitt, 2006). However, the actual protection accorded to them is not proportionate to their conservation status. For example, Tadesse (2008) stated that a significant amount of timber continues to be illegally extracted by local people. Illegal tree cutting and timber extraction focuses on species such as *Pouteria adolfi-friedrici, Afrocarpus falcatus* and *Cordia africana* (Aseffa, 2007). Other studies by Schmitt et al. (2006) and Aseffa (2007) showed that these forests were under pressure from expansion of private coffee plantations and other land-uses. The expansion of intensive agricultural monocultures of tea, and other rain fed crops, as well as coffee, at the expense of forest cover could change the functional landscape pattern (Tadesse et al., 2002; Tadesse, 2003; Schmitt et al., 2006; Tadesse et al., 2008). The fragmentation of the forests may also harm provisioning ecosystem services, e.g. productivity of understorey *Coffea arabica L.* may be reduced under reduced shade levels when it is less able to compete for soil nitrogen (Prof. Ensermu Kelbessa, pers. comm.).

In summary, the forests of south-west Ethiopia have been used by local communities over centuries for a variety of uses and practices (Demel et al., 1998; Tadesse and Denich, 2001; Fayera and Denich, 2006). Livestock grazing, harvesting of fuelwood, fodder, timber and managed understorey coffee have coexisted with limited complete clearance of forest for agriculture in historic and more recent times (FAO, 2010). The recent rapid increase in these activities as human populations in the area have greatly increased creates substantial pressure on the forest resources which results in the fragmentation of patches and modification of spatial heterogeneity. This, in turn, results in a substantial change in landscape pattern which is likely to disrupt ecosystem function and ecological process (Gelgel and Turner, 2002). The impact of this disturbance is related to the resulting size and shape of individual patches and, at a landscape scale, to patch structure and spatial arrangement (Turner et al., 2001). The deforestation and forest degradation in south-west Ethiopia is likely to alter the structure of populations and thus the regeneration dynamics and species composition within forest patches (Mehta et al., 2008). Failure to maintain functional and resilient ecosystems is likely to have serious negative consequences for provision of ecosystem goods and services. However, quantitative studies on forests in south-west Ethiopia were relatively few. The majority of these studies have focused simply on description of vegetation composition (Kumelachew and Tamirat, 2002; Tadesse, 2003). In contrast, few studies have attempted an integrated ecosystem functional assessment of the impact of fragmentation in order to elucidate the linkage and feedbacks between, for example, change in forest spatial pattern and its impacts on species diversity, composition and the structure of populations.

1.1.6 Objective of the study

This study aims to provide the evidence required to inform the development of conservation policy for the remnant forests of south-west Ethiopia. It will quantify the fragmentation of forest habitat resulting from recent deforestation and degradation, the consequences for plant species biodiversity, and the resulting conservation value of the remaining forest patches. It will also investigate the relationship between forest protection and the culture and institutions of local communities. The objectives of this study are:

1. To evaluate the impact of land-use change on natural land-cover pattern and specifically the fragmentation of remaining forests in landscapes with contrasting land uses within the Illubabor and GamoGofa zones, south-west Ethiopia.

2. To investigate the impacts of fragmentation of afromontane rainforests in the Illuababor Zone on plant species composition, with a particular focus on small forest patches embedded in a matrix of agricultural land in contrast to larger forest blocks.

3. To evaluate how the response of plant species to fragmentation in afromontane forests is linked to their functional groups, focusing on responses to forest edge and interior environments, and the consequences of this for community composition of forest patches.

4. To determine the value of the sacred forest groves of the GamoGofa Zone of south-west Ethiopia for conservation of regional plant diversity, and the level of threat that these habitats are currently under.

CHAPTER 2

DESCRIPTION OF THE AREA AND GENERAL APPROACH OF THE STUDY

2.1. Description of the study area

The afromontane forest areas of south-west Ethiopia are predominantly under intensive management for coffee production. This is practiced by both local farmers and government-backed investors. The growing importance of forests for coffee production and the demand for more space by the rapidly increasing human population has put great pressure on the remnant natural forests (Schmitt et al., 2010). The understorey of the area’s forests is characterized by a large amount of wild and managed coffee and is considered a main national economic area. The area also contains a large number of indigenous sacred groves managed for religious purposes within a matrix of agricultural and settled land. The study was conducted in Illubabor and GamoGofa Administrative Zones of, respectively, Oromia and Southern Nations, Nationalities and Peoples Regional (SNNPR) states, in south-west Ethiopia (Figure 2.1).

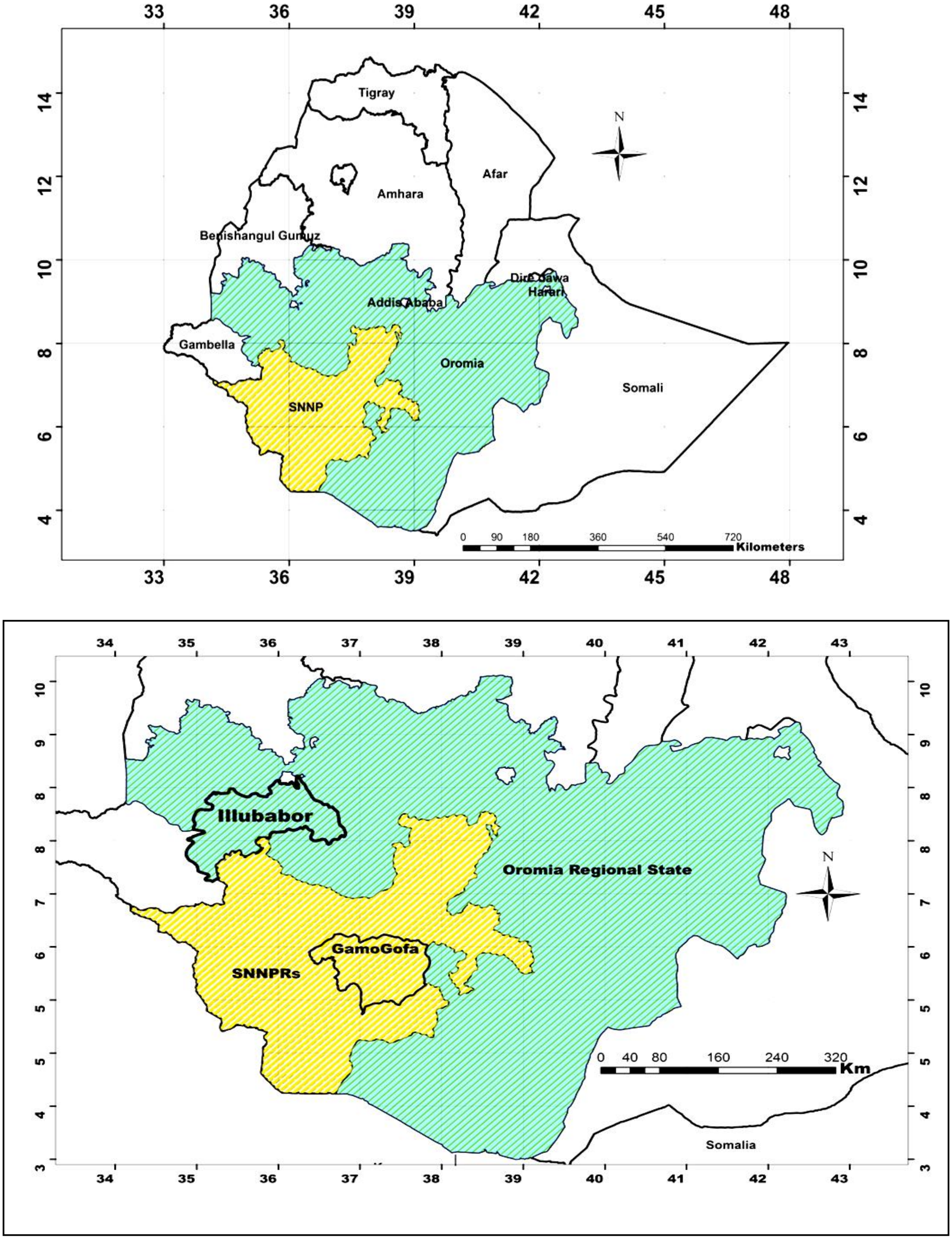


Figure 2.1 States and Administrative Zones in Ethiopia, showing the Oromia and Southern Nations, Nationalities and Peoples Regional (SNNPR) states (top panel) and, within them, the Illbuabor and Gamo Gofa Administrative Zones (respectively) (bottom panel) in south-west Ethiopia in which the study was carried out.

2.1.1 Illuababor study area

Illubabor Administrative Zone covers an area between latitude 8^0 01′ 50.06″ and 8^0 10′ 58.83″ N; longitude 35^0 11′ 11.90″ and 36^0 40′ 29.38″E and has a land area of 16,000 km^2. The topography of the area encompasses flat areas, river valley slopes and water-logged valley bottoms, with an elevation range of 1000-2500 m a.s.l. The geology is part of the south-west highlands block which are formed largely of Precambrian basement complex, Tertiary volcanic rocks from the trap series and quaternary sediments (Kazmin, 1975, Davidson, 1983). The Tertiary volcanic rocks mostly overly the Precambrian rocks (Friis et al 1982) and include agglomerates, rhyolites, trachytes, tuffs, ignimbrites and basalts (Tadesse, 2003). The soils of the area are mainly derived from alkali olivine basalt and tuffs (WBISPP, 2000; Mekuria, 2005) and are categorized as rhodic nitosols, eutric cambisols and chromic luvisols (Tafesse, 1996, FAO, 1998).

Illubabor is one of the most important coffee growing zones in south-west Ethiopia, and is undergoing one of the highest rates of deforestation and fragmentation (Tadesse et al., 2001). Within this area, functional landscape pattern has been altered by major modification and fragmentation of natural habitats due to commercial-scale coffee growing, farming of other crops and livestock, and settlement expansion. A major driver is rapid population expansion due to planned as well as spontaneous migration of people from other parts of the country. The spontaneous in-migration dates back to the 1930s (Tadesse, 2003; Dixon and Wood, 2007). However, most has occurred since the 1980s due to a government supported mass resettlement programme (Mengistu, 2005).

2.1.2 Climate

In Ethiopia the climate is mainly controlled by the seasonal migration of the Inter-Tropical Convergence Zone (ITCZ) and associated atmospheric circulations, as well as by the complex topography of the country (Gemechu, 1977). It has very diversified climate, ranging from the semi-arid desert type in the lowlands to the humid and warm type in the highland (Genechu 1977). The different parts of the country experience different distributions of rainfall and temperatures through the year (Woubshet, 2001).

The whole area of Illubabor Zone experiences widespread and prolonged monsoon rains from May through September reaching a peak in August (Figure 2.2). However, a rainfall gradient exists across the Zone from 1500 mm to 2200 mm, with a mean annual temperature range between 18.5 and 26.8 ^{0}C (Ethiopian Meteorological Agency unpublished data).

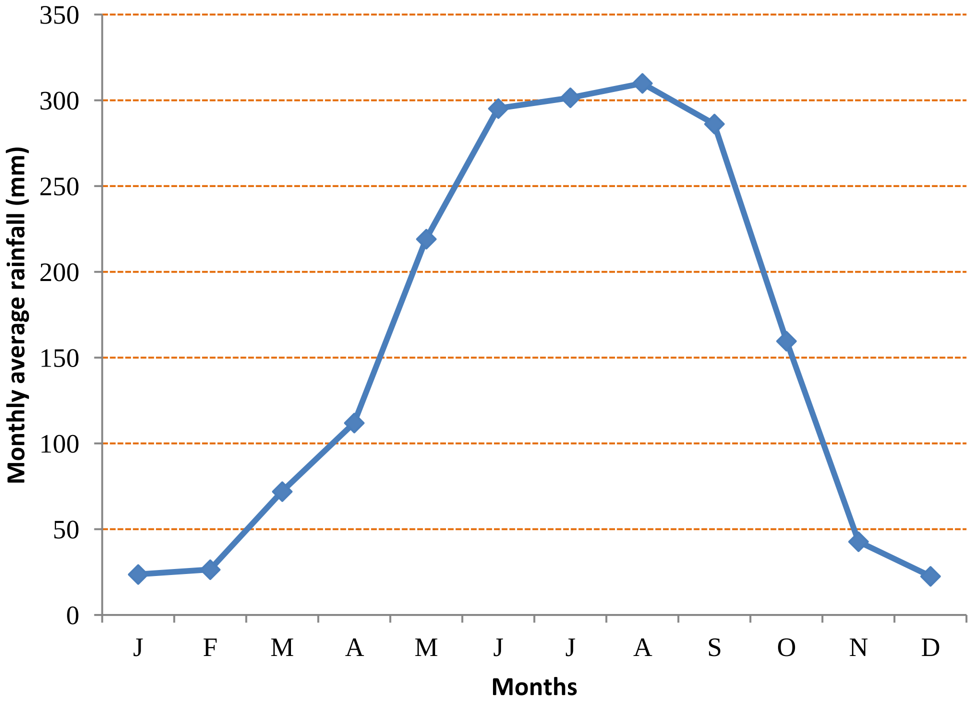


Figure 2.2 Annual rainfall distribution pattern in Illubabor Zone, average for whole zone (Source: Ethiopian Metrological Agency, unpublished data).

2.1.3 Spatial and temporal distribution of rainfall

The rainfall of a given area varies according to topography, altitude and orographic pattern, and has a complex relationship with vegetation cover (Buytaert et al., 2006, Suhaila and Jemain, 2009, 2012). Therefore the rainfall data taken at a single station cannot provide a reliable estimate for a large study area. To address this problem, a model of the spatial and temporal distribution of rainfall was obtained using monthly rainfall data recorded over 23 years (1986-2009) at 24 different locations distributed all over the Illubabor Zone, obtained from the Ethiopian Meteorological Agency. The locations of the rain-gauge stations selected for the study are shown in Figure 2.3. Before analysis, the data quality was checked and estimates of missing values were inserted based on the average rainfall recorded in the same period in other weather stations surrounding the study area. Time series of monthly average rainfall over different selected stations were prepared and the mean and coefficient of variation calculated for different years and stations. The mean rainfall of different years was compared to find out the temporal and spatial variability of the average rainfall pattern within the study period between 1986 and 2009 and amongst different stations. For each of the 24 metrological stations and each of the 30 forest patches that were sampled (as described in section 2.3.5 below), GPS readings were taken, plotted on a map, and used to determine which metrological station was closest to each patch. The mean annual rainfall for each patch was estimated to be equal to that of its nearest station. The rainfall value of some individual stations had to be used for more than two sampled forests (Figure 2.3).

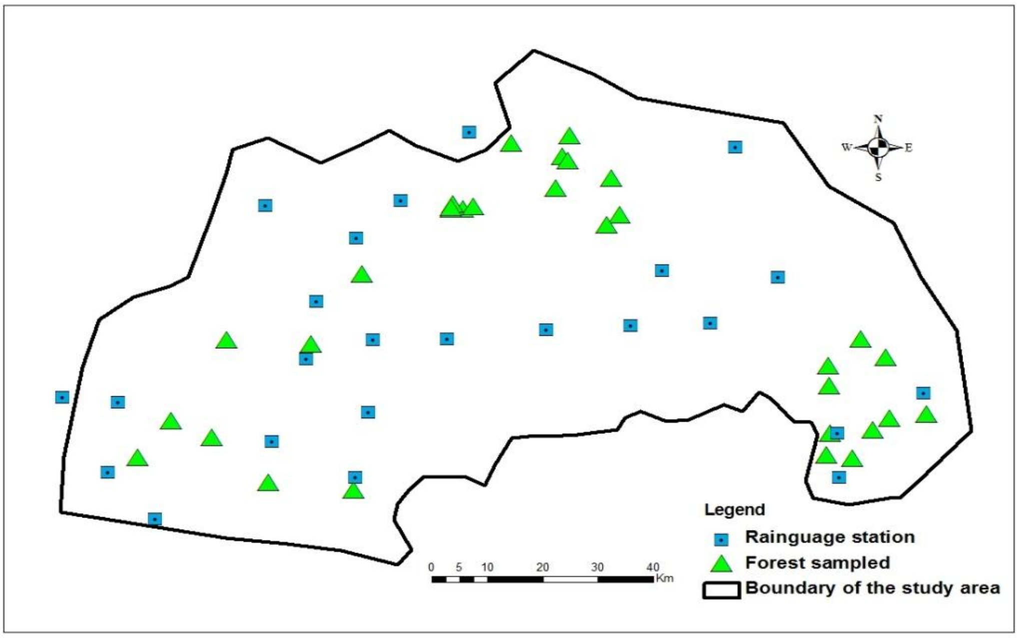


Figure 2.3 Location of the 24 rain gauge stations and 30 sampled forest patches in the study area in Illubabor Administrative Zone, Ethiopia. This Figure is not the full size of Illubabor Administrative Zone, it is confined to the area where the sampled forests are located. The boundary of this Figure was decided based on the area coverage of Satellite Image obtained.

The spatial distribution of rainfall across the study area showed high spatial variability. The spatial variability of mean annual rainfall from station to station is much higher in the lower altitude central south and south west of the Zone which are the areas with greater forest cover. The mean annual rainfall ranges from about 2412 mm in the south, gradually decreasing to 1121 mm in the west of the Zone with the largest annual rainfall recorded at Didu-Gordomo station (2412 mm) and the lowest at Bure station (1121 mm) (Figure 2.4). Only four stations recorded an average of less than 1500 mm annual rainfall. Average annual rainfall for the 24 stations was variable amongst the 23 sample years (Figure 2.5). It ranged from 2060.6 mm in 1988 to 935.5 mm in 2006. However, the inter-annual variability was generally not great (coefficient of variation = 0.15). The period 2002-2009 was notably drier (mean annual rainfall = 1432.2 mm) than the period 1986-2001 (mean annual rainfall = 1873.1 mm). In general, the temporal distribution of rainfall shows decreasing from 1986 to 2009. The years 2002-2009 constituted eight of the eleven driest years over the 23 year period. All areas of the zone have a strong seasonality of rainfall with a single prolonged monsoon rainy season (Figure 2.2).

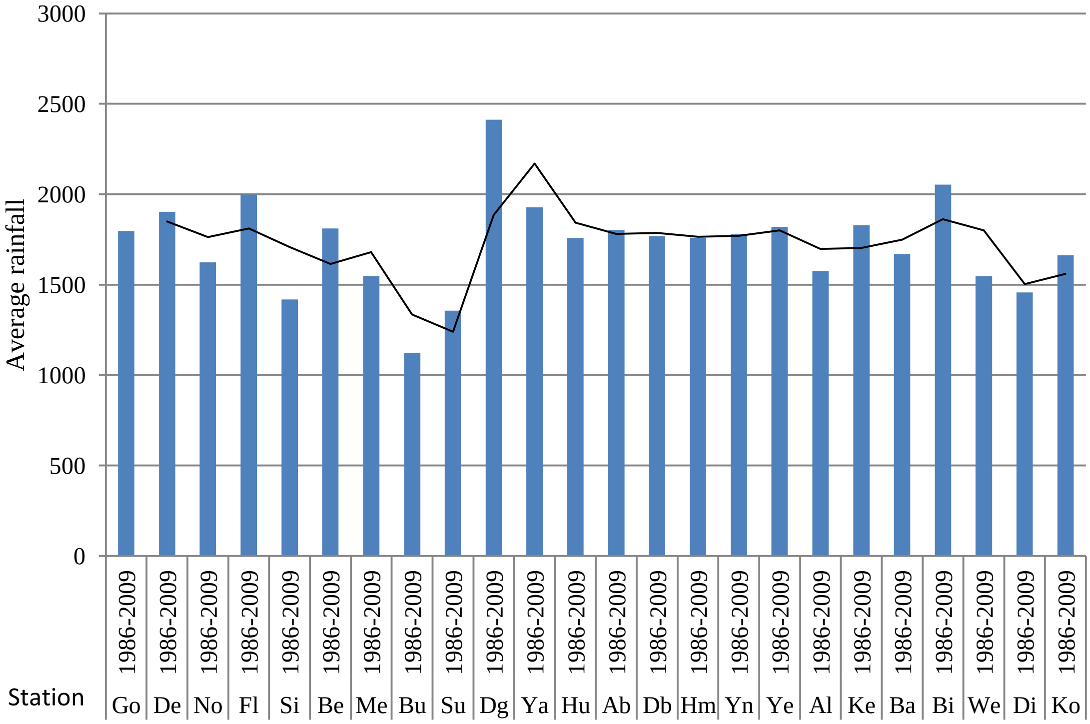


Figure 2.4 Mean annual rainfall during the 23 year period 1986-2009 for 24 meteorological stations distributed across Illubabor Zone, south-west Ethiopia. Source of data: Ethiopian Metrological Agency

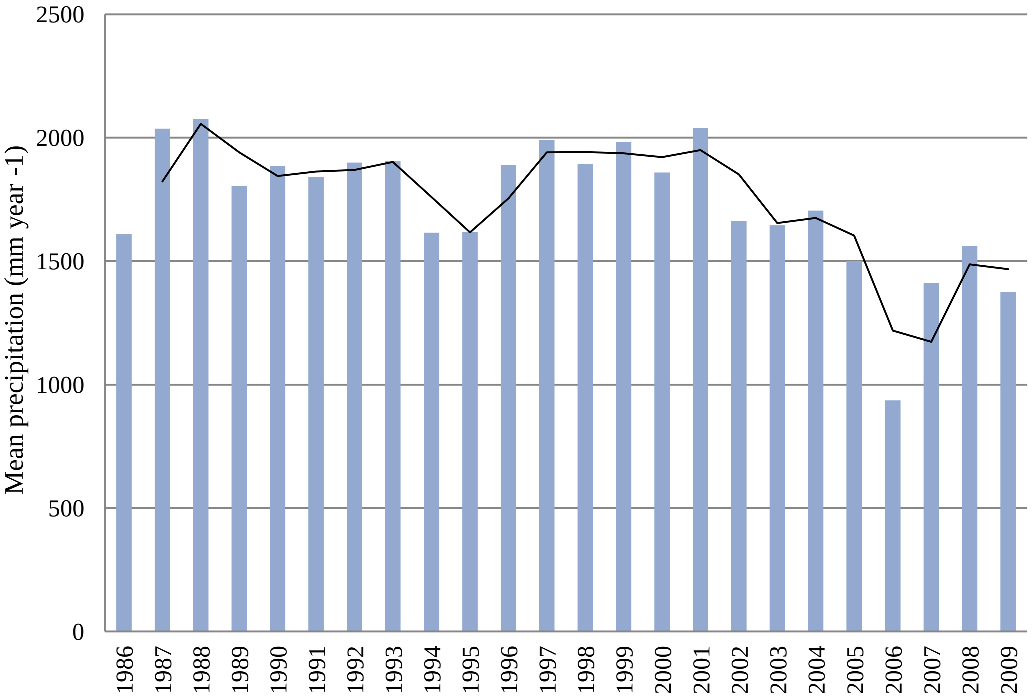


Figure 2.5Annual rainfall for each year 1986-2009 as the mean of 24 meteorological stations distributed across Illubabor Zone, south-west Ethiopia: Source of data: Ethiopian Metrological Agency

2.1.4 Temporal distribution of temperature

The average annual temperature was variable amongst the 23 sample years (Figure 2.6). The maximum ranged from 24.9 to 28.4 ºC and the minimum from 12.4 to 13.7 ºC (Table 2.2). The inter-variability was not great ($CV_{maximum} = 0.03$, $CV_{minimum} = 0.10$). However, the mean temperature was higher during the drier period (2002–2009), mean rainfall 1432 mm, mean temperature 27.05 ºC, than during the wetter period (1986-2001), mean rainfall 1873.1 mm, mean temperature 26.95 ºC). The highest monthly temperature was recorded in March (24.52 ºC) and November (24.50 ºC) and the lowest in May (23.78 ºC) and August (23.87 ºC).

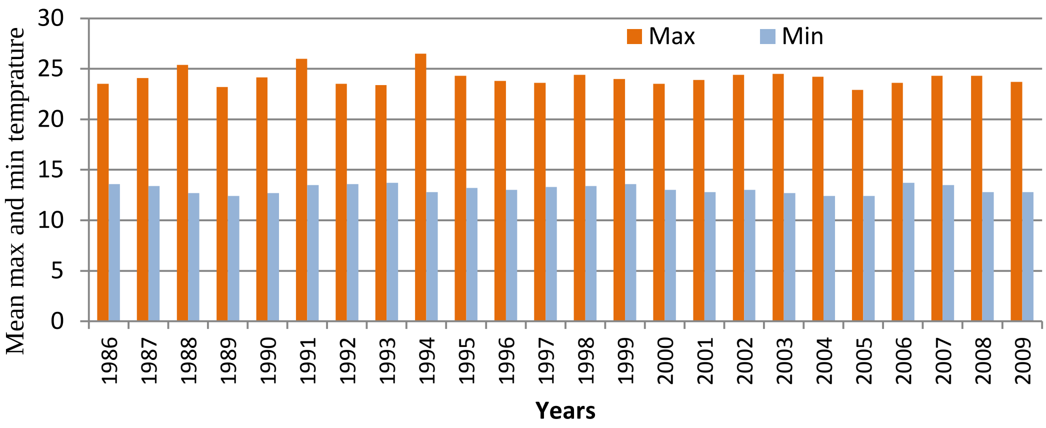


Figure 2.6 Mean annual maximum and minimum temperatures for each year in the 1986-2009 period as the mean of 24 meteorological stations distributed across Illubabor Zone, Ethiopia (station details in Table 2.1). Source of data: Ethiopian Metrological Agency

Table 2.1 Location details and mean annual rainfall over the 1986-2009 period for 24 meteorological stations in Illubabor Zone, Ethiopia. Source of data: Rain fall obtained from Ethiopian Metrological Agency, station altitude, latitude and longitude taken from Digital elevation models (DEM)

Station code	Station name	Station altitude (m)	Station position (latitude-longitude	Average annual rainfall for 24 years (mm)
Go	Gore	1936	8°69′01.08″-35°31′47.53″	1797
De	Dembi	1976	8°05′30.33″-36°27′40.41″	1903
No	Nopa	1652	8°24′28.15″-35°36′06.98″	1623
Fl	Leka	1952	8°65′11.17″-35°40′12.06″	1999
Si	Sibo	1617	8°13′16.54″-35°16′40.65″	1418
Be	Bedele	1999	8°27′25.94″-36°21′30.00″	1811
Me	Metu	1686	8°18′16.70″-35°44′08.36″	1548
Bu	Bure	1540	8°13′42.08″-35°11′14.95″	1121
Su	Supe	1663	8°31′15.85″-35°39′52.90″	1356
Dg	Didu-Gordomo	1565	8°60′49.33″-35°20′30.87″	2412
Ya	Yambero	2195	8°10′17.72″-36°27′34.02″	1928
Hu	Hurumu	1792	8°20′11.60″-35°42′00.63″	1758
Ab	Abdela	1930	8°22′25.77″-36°14′54.04″	1803
Db	Debena	2084	8°28′23.09″-36°15′39.84″	1768
Hm	Humbe	2004	8°21′53.89″-36°07′01.44″	1760
Yn	Yanfa	2123	8°14′52.34″-36°35′58.55″	1780
Ye	Yembo	2001	8°21′59.36″-36°07′02.98″	1820
Al	Alge	1829	8°35′31.30″-35°44′08.36″	1576
Ke	Kemisse	1973	8°27′57.87″-36°10′00.26″	1828
Ba	Baro	1381	7°41′39.77″-34°59′00.51″	1670
Bi	Bilambilo	2000	8°21′59.59″-36′07′01.40″	2053
We	Wetete	1944	8°22′11.99″-36°14′46.58″	1547
Di	Dippa	1671	8°34′52.99″-35°30′56.44″	1458
Ko	Kone	1987	8°41′22.56″-36°17′20.93″	1662

Table 2.2 Mean maximum and minimum temperatures and rainfall for each year in the 1986-2009 period as the mean of 24 meteorological stations distributed across Illubabor Zone, Ethiopia (station details in Table 2.1). Source of data: Ethiopian Metrological Agency

Year	Max. temp. (°C)	Min. temp. (°C)	Rainfall (mm)
1986	26.3	13.6	1609
1987	27.2	13.4	2037
1988	28	12.7	2075
1989	27.8	12.4	1805
1990	28.1	12.7	1885
1991	28.4	13.5	1841
1992	26.2	13.6	1899
1993	24.9	13.7	1904
1994	28.2	12.8	1615
1995	26.5	13.2	1619
1996	27.3	13.0	1891
1997	26.2	13.3	1991
1998	27	13.4	1893
1999	26.3	13.6	1982
2000	25.8	13.0	1860
2001	27	12.8	2040
2002	26.8	13.0	1663
2003	27.7	12.7	1645
2004	27.5	12.4	1705
2005	26.5	12.4	1501
2006	28.1	13.7	936
2007	27.9	13.5	1411
2008	26	12.8	1562
2009	25.9	12.8	1374

2.1.5 Forests

The forests of southwest/east Ethiopia are the only large block of intact forest that remains in the country. Their flora was among the least known in tropical Africa until recent years (Tadesse, 2003). Early biogeographical studies (Cheffy, 1942, Logan, 1946), recent detailed botanical and vegetation surveys by Friis (1992), and Friis and Sebsebe (2001), and studies of the ecology and floristics (e.g Abayneh, 1998; Kumelachew and Tamrat, 2002; Tadesse, 2003) together indicate that the forests of the Illubabor zone largely consist of floristically and structurally distinct afromontane rainforest and transitional rainforest vegetation types. The distribution of these two vegetation types (phytogeography) is influenced by elevation (Schmitt et al., 2010). The transitional rainforest occurs in the part of the Zone below 1500 m elevation and was sampled by Site I of the present study (Figure 2.11). In this forest type, species associated with Afromontane rainforest and those associated with lowland semi-evergreen forest both occur, as well as species unique to this forest type (Friis, 1992). The Afromontane rainforest occurs in the eastern and northern highland section of the Zone above 1500 m elevation and was sampled by Sites II and III in the present study (Figure 2.11). These forests are highly fragmented, existing as remnant patches of the once extensive natural forest now embedded in an agricultural landscapes (pers.obs).

2.1.6 Population and livelihoods

The Illubabor Administrative Zone has the largest population density in Oromia Regional State and has been subject to dramatic population growth in recent years (Tadesse, 2003). This growth accelerated with the large resettlement programme that has taken place since 1984, compounded by subsequent additional spontaneous migration to this area (Mengistu, 2005). Based on data from the Ethiopian Government Central Statistical Agency (CSA, 1996), the population density in the zone in 1994 was 80.3 persons per km^2. According to the 2007 national Ethiopian population and housing census, the population size of Illubabor Zone is about 1.3 million (CSA, 2007); it had increased by 66.3% compared with the 1994 census (847,048 population).

Different ethnic groups inhabit the Illubabor Zone of which the Oromo have the largest population (Tadesse, 2003). The major occupation is agriculture occupying more than 90% of residents (Tadesse and Denich, 2001). The major crops grown are wheat, barley, maize and coffee. Coffee production, processing and marketing are the major sources of employment for communities at lower altitudes (Tadesse, 2003). Cattle, sheep and goats are the most important livestock in the farming systems. They are usually kept for income and as a source of manure. Marginalized pastureland and forests are the major feed source.

Natural forests contribute to the livelihoods of most people in the Zone. They depend on the forests mainly for coffee, spice and honey production (Tadesse and Denich, 2001). Communities living in lowland areas depend most heavily on forest products. They manage and harvest in large quantities from the following species in semi-natural forests: *Coffea arabica, Piper capense* and *Aframomum corrorima* (Aerts et al., 2011). The other major component of their livelihoods is production of a few agricultural crops, in particular *Zea mays, Sorghum bicolour* and different *Brassica* species in small parcels of land which are managed as home gardens (the land around the houses) (pers.obs.). The livelihoods of communities living in the upland areas are largely based on production of field grain crops such as *Eragrostis teff, Triticum spp., Hordeum vulgare, Zea mays* and *Sorghum bicolour*, in addition to production of *Coffea arabica* at a small scale in their home gardens and in small areas inside forest patches.

2.1.7 Sampling strategy

Taking the biogeographical aspects and anthropogenic patterns described above this study uses a sampling strategy where the Illubabor Administrative Zone is stratified into a) lowland and b) upland. Then, the upland area was again classified into II) northern and III) eastern based on coffee production: Site I) the lowland area is mainly coffee production zone in the western part of the zone where people depend heavily on forests for the production of coffee which dominates their livelihoods (Figure 2.11); Site II) the medium-intensity coffee production area in the upland northern part of the zone where people's livelihoods depend primarily on production of other crops but forests are still important to supplement livelihoods through growing coffee and harvesting forest products (Figure 2.11); Site III) the area in the upland eastern part of the zone where people's livelihoods also depend heavily on production of agricultural crops supplemented by income from low levels of coffee production within forest patches (Figure 2.11). Sites II and III contain several disjunct blocks of highland afromontane rainforest separated by a matrix of land highly modified from its natural condition (Chapter 3). The three sites together are well distributed over the range of environmental conditions in the broader landscape of Illubabor Zone.

2.2 Gamo highlands study area

The Gamo highlands are also a section of the south-west Ethiopian highlands, and are located in the GamoGofa Administrative Zone in the Southern Nations, Nationalities and Peoples Regional state (SNNPRs) (Figure 2.7). The Gamo highlands are located on the western escarpment of the Great Rift Valley (Figure 2.7) between 5^0 53.8′ 17.52″ to 6^0 26′ 22.97″ N and 37^0 10′ 35.13″ to 37^0 42′ 31.89″ E. The topography of the Gamo highlands is characterized by steep slopes, up to undulating plateaus with gentle slopes, as well as detached steep-sided hills and valleys (Desalegn, 2007, Figure 2.7). The elevation of the highlands rise abruptly from 1183 m in the Maze lowlands to the west and from 1200 m at Lake Abaya and Chamo in the east to the central ridges with a maximum elevation of 3500 m at the summit of Mount Guge. The geology and soil types of the Gamo highlands are similar to the Illubabor Administrative Zone (Davidson, 1983). The Gamo highlands are the origin of a large number of streams and rivers, which are the primary source of water for people living in both the highlands and surrounding lowlands.

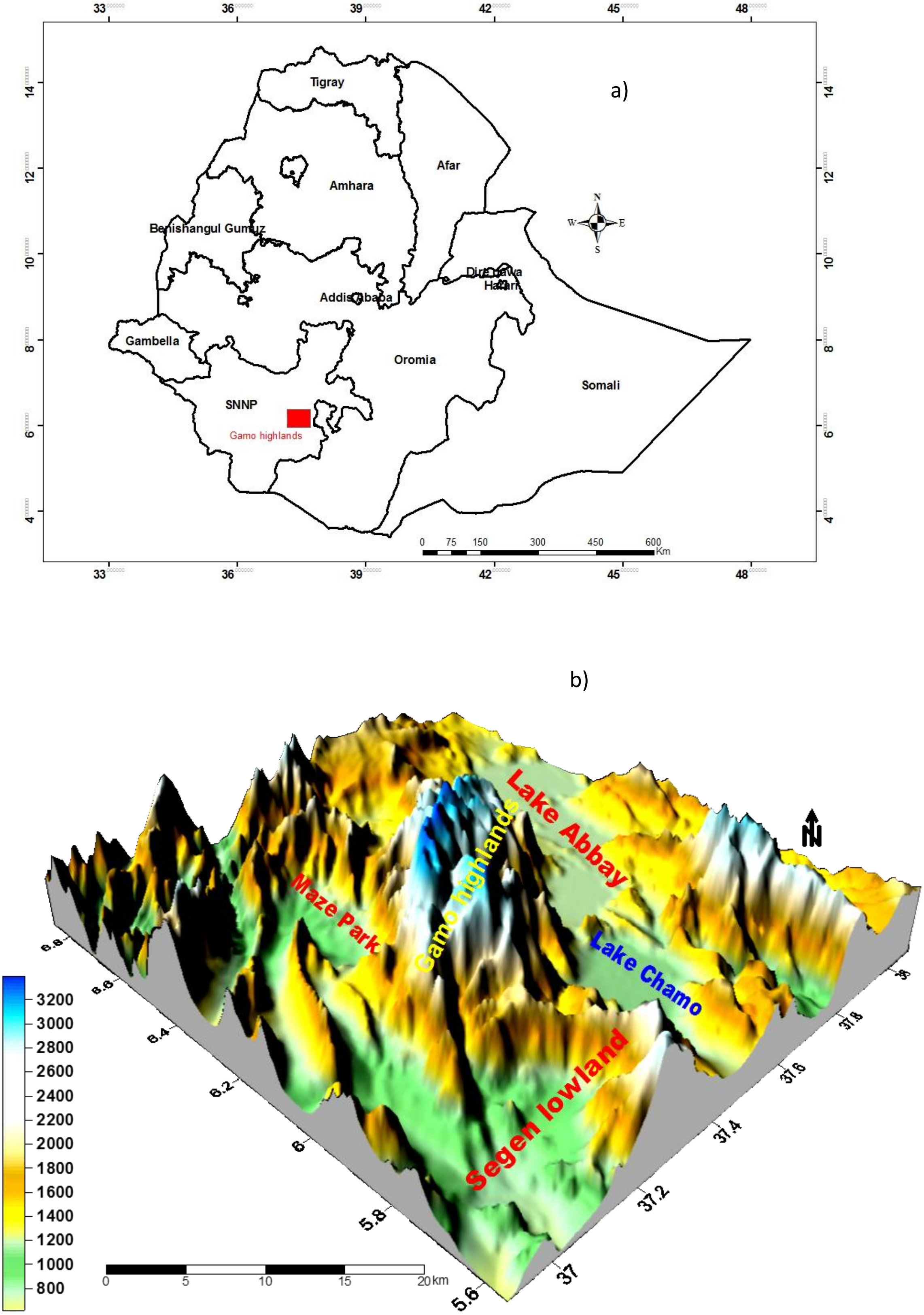


Figure 2.7 The location of the Southern Nations, Nationalities and Peoples Regional state (SNNPRs) and the Gamo highlands study area (a) coloured red (b) showing its elevation (m) topographic characteristics.

2.2.1 Population and livelihoods

The gentle slopes, valley bottoms and lowland plains of the Gamo highlands are inhabited by the Gamo people with a total population of 1,107,163 (CSA, 2007). The pattern of settlements is based on ritual and traditional political systems and divided into ten districts: Dita, Zada, Doko, Dorze, Ocholo, Ganta, Boroda, Qogo, Bonke and Kamba (Freeman, 2002; Weedman, 2008). Within each district, traditional houses are concentrated in flat parts of the mountainous landscape.

The livelihoods of the people are primarily based on livestock rearing and crop production (Belete, 2006). Local varieties of the following crops are cultivated: *Enset ventricosum, Colocasia esculenta, Dioscorea alata, Manihot esculenta, Ipomoea batatas* and *Rutha chalapensis*. Other areas on the plateaus, lower slopes and parts of the valleys are used during the rainy season to cultivate wheat, barley and maize (pers.obs). The people practice a polyculture with annual crops mixed with a large number of fruit-bearing trees and shrubs such as apple and grape. Other tree and shrub species characteristically occur in the managed land in hedgerows, e.g. *Justicia schimperiana*, *Vernonia amygdalina* and *Dovalyis cafra,* and as isolated sacred trees, e.g. *Erytheina brucie*, *Hagenia abyssinica* and *Afrocarpus falcatus*. Each household cultivates a home garden and a separate cultivated field, both of which have defined boundaries (pers. obs.). Land tenure is traditionally linked to extended families. Beyond the cultivated fields there are small “family reserve” lands, which are used for grazing of small animals, usually calves. Adjacent to these cultivated and small grazed lands are other sacred landscape features such as sacred *dubusha*, ultimately controlled by either the extended family or individuals. The location of sacred pasture land is not easily defined but it is usually surrounded by the village and is controlled by designated community custodians or the community at large (Shagire Shano and Mako Wario, community chief and religious leader respectively, pers. comm.). This sacred Site Is used by the whole community who come together to celebrate yearly thanksgiving and ritual purification ceremonies. Sacred groves, i.e. natural sacred relict forests as well as burial grounds, are either controlled by the community of a similar clan or by religous leaders locally known as *Maka*. They are usually located on top of hills. The rest of lanscapes were covered by *Euclaptus spp* and *Cupressus lucitanica*

2.2.2 Vegetation

The Gamo highlands have various ecosystems ranging from afro-alpine grassland on the high mountain Guge (above an altitude of 3400 m) to various dry and moist afromontane forests in the altitudinal range 1500-3000 m (Desalegn, 2007). The vegetation is typical of south-west Ethiopia with the moist and dry evergreen vegetation types in a complex mosaic. The northeastern sides of the main mountain ranges are the windward side for the prevailaing easterly winds that bring most rainfall to the area during the rainy season (Fiedler and Gebeyehu, 1988) and are consequently characterized by moist forest types. Sites with all other aspects (from east round to north) have relatively a much lower rainfall and are dominated by dry evergreen forest types with a similar flora to the central Ethiopian highlands (pers. obs.).

2.2.3 Climate

The climate of the area is characterized by a bimodal rainfall pattern. The area receives high intensity rainfall from June to September which is locally known as *Badhessa* and low intensity rains from February to April known as *Balgo*. The mean annual rainfall ranges from 500 mm in the lowlands to 1200 mm in the highlands, and the temperature varies from 10 °C in the highlands to 25 °C in the lowlands (FDRE, 2000). More detailed long-term rainfall and temperature data were unavailable for most of the Gamo highlands area due to lack of weather station. This makes it difficult to compare the spatial distribution of rainfall and temperature across the Gamo highlands, however a rainfall map covering the area is shown in Figure 2.8.

2.2.4 Site characteristics of sampled forests

Forests of two different management types (sacred groves and non-sacred forests) were purposefully sampled in this part of the study (as described in chapter 6). These two forest types are located in distinct geographical areas in the north and south of the Gamo highlands respectively, however these two areas have smilar average rainfall of 1600 mm (Figure 2.8). The average altitude for the sampled sacred groves (2259 m) was only slightly lower than that of the non-sacred forests (2331 m) (Table 2.3). The two forest types did, however, differ greatly in their slope angles. The six sacred grove sites had a slope range from 1.39 º to 11.12 º. In contrast the slope for the four non-sacred forest sites was all > 16.45 ° except for Oche which was 1.39 º. The average slopes for the sacred grove and non-sacred forest sites are 6.26 ° and 17.04 ° respectively. The surface geology and soil texture of the two sites are also different. The non-sacred forest sites are chracterized by sandy soil whereas sacred forest sites chracterized by fine-textured or organic matter (pers.obs).

Table 2.3 Site characteristics of the studied sacred groves and non-sacred forests in the Gamo highlands, SW Ethiopia

Site name	Altitude[1] (m)	Aspect[2]	Slope[3] (degree)	Geology[4]
Sacred groves				
Ula	2243	Flat	1.39	AOBTRR
Gufae	2248	W (247.5-292.5)	11.12	AOBTRR
Qimme	2246	N (337.5-360)	11.12	AOBTRR
Tele	2389	NW (292.5 -337.5)	11.128	AOBTRR
Akasie	2282	Flat	1.39	AOBTRR
Osha-Ocha	2146	Flat	1.39	AOBTRR
Average	2259	311.2	6.26	
Non-sacred forests				
Sora	2235	NW (292.5-337.5)	28.31	RTT
Shoa	2215	N (0-22.5)	22.06	RTT
Dhule	2438	NW (292.5-337.5)	16.45	RTT
Oche	2436	S (157.5-202.5)	1.35	RTT
Average	2331	205.3	17.0	

[1] Altitude measured using GPS, [2] Aspect and [3] slope derived from TIN (Triangulated Irregular Network) data created from contour. [4] Woody Biomas Inventory and Strategic Planning Project (WBISPP) (2000); AOBTRR, Alkali olivine basalt and tuffs and rare rhyolites; RTT, Rhyolites and trachytie tuffs (WBISPP, 2000). The average for aspect and slope were calculated using the median of each site range.

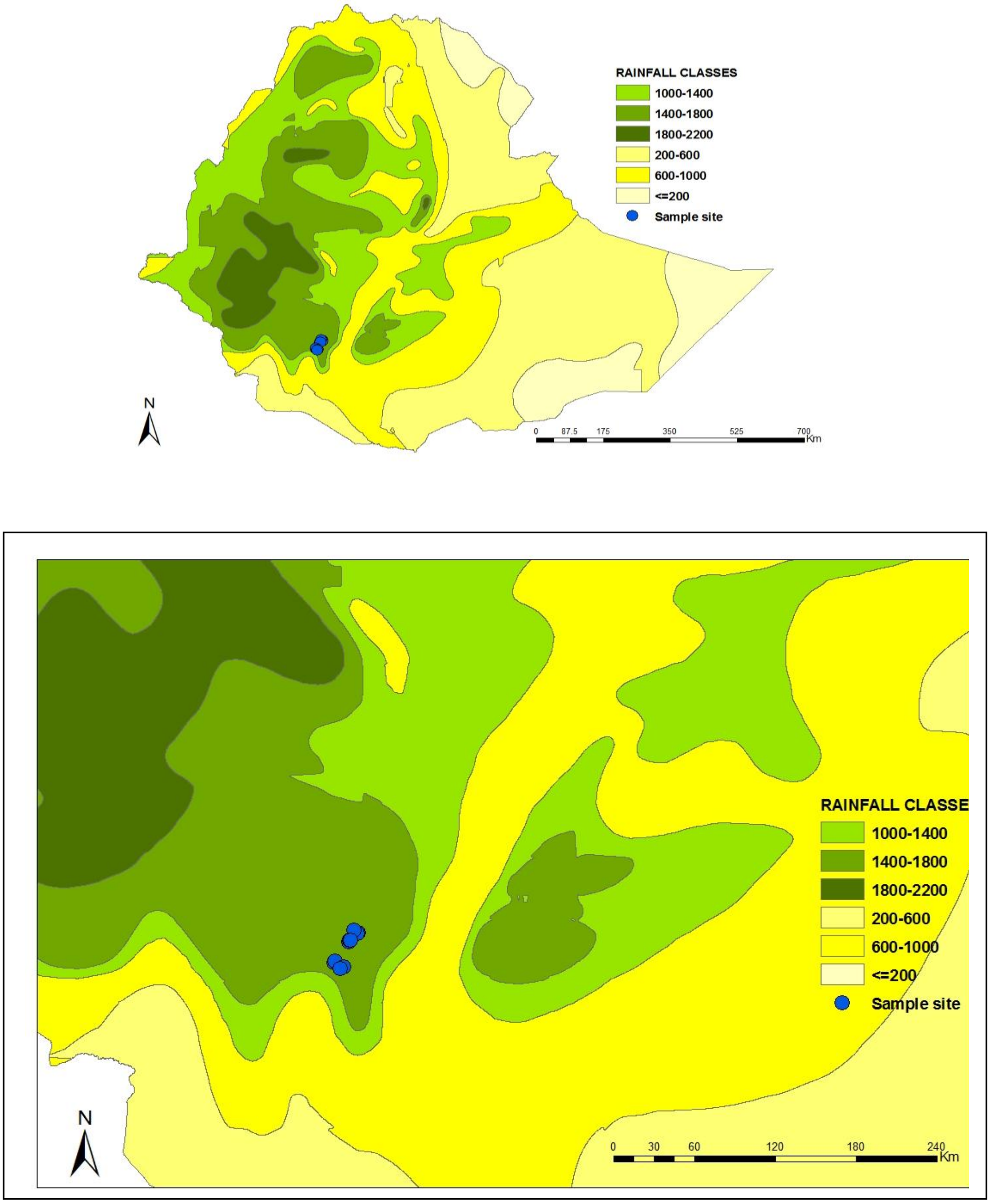


Figure 2.8 Class distribution of mean annual rainfall (mm) for Ethiopa, showing the country as a whole in the top panel and the Gamo highlands study area in the bottom panel. The position of the Gamo highalnds in the top panel can be ascertained from the location of the study sites shown in both panels. The rainfall distribution class was obtained from Ethiopian Ministry of Agriculture (2000). GPS locations for the study sites were taken during the study.

2.3. General approach of the study

Landscape pattern analyses can be conducted on land-use/land-cover spatial data produced from different data sources (Turner et. al., 2001). According to Gergel and Turner (2002), there are three main types of data source for landscape studies: aerial photography, digital remote sensing and published data and census data. In this study, digital remote sensing data were used because they cover larger landscape of the study area with multi-temporal records of the same landscape, allowing analysis of elements, units, and types of landscapes in different years. The study involved three general approaches:

1) Production of land-use/land-cover change maps from three satellite images covering a time period of 21years in Illubabor Zone and two satellite images covering 15 years in GamoGofa Zone,

2) Quantification of landscape structure using different landscape metrics to measure landscape composition and structure for analyzing the effect of land-use change on landscape pattern,

3) Ecological data collection (plot-based vegetation survey) in selected forest fragments to examine floristic composition, species richness and abundance.

2.3.1. Satellite remote sensing data and image processing

Satellite images and data have come to play an important role in the study, documentation and mapping of the world’s terrestrial and aquatic surface for the past three decades (Tso and Mather, 2009). Remote sensing techniques have contributed in various ways to the assessment of the physical properties of the environment. An aspect of growing importance is in the surveying and monitoring of the effect of various land-use and land-cover changes taking place in the environment including the response to land management (Foley et al., 2005; Reddy, 2008).

Landsat and Spot are two important satellites for earth resource observation (Lillesand and Kiefer, 1993). They have been available since 1972 and 1986 respectively, making possible long-term detection and monitoring of changes in land-use and natural vegetation (Campbell, 1996). Each of these satellites provides different image bands and resolution that have been widely used for surface classification (Reddy, 2008). Early Landsat (Landsat 1, 2 and 3, known as Landsat-MSS) had sensors that recorded two bands of reflected light in the visible spectrum (green and red) and two in the near infrared spectrum. This sensor scanned a continuous 185 km-wide swath with a ground resolution (spatial resolution) of 79 m. The sensor was later improved in the 1980s to the new sensor called Thematic Mapper (Landsat 4 and 5, known as Landsat TM) that recorded seven-band spanning from the blue band of the visible spectrum up to the thermal band. It has an improved spectral resolution over the Landsat MSS. It also has an improved spatial resolution of 30 m by 30 m for six bands and 120 m for the thermal band. In 1999 Landsat-7 was launched equipped with higher resolution instruments. It carries the Enhanced Thermal Mapper Plus (ETM+) which is an eight-band sensor. These bands are identical to those of the TM with the addition of a panchromatic band (Band 8) that records energy with a 15 m resolution, and an improvement of the spatial resolution of the thermal band to 60 m. Landsat TM and ETM+ are highly advanced sensors that are the most widely used remote sensing system for landscape survey (Campbell, 1996).

The SPOT satellite is similar to Landsat in many ways (Lillesand and Kiefer, 1993). It has similar multispectral and multitemporal options as well as a potential for large area coverage in a single scene. However, SPOT has a highly increased spatial resolution of 5-10 m in black and white (panchromatic) registration and 10-20 m for colour (multispectral) registration, and therefore makes it possible to obtain much greater numbers of picture elements per unit area.

For this study, the Landsat images of 1986 (TM) and 2000 (ETM+) and a Spot image of 2007 were acquired which cover the majority of the Illbuabor Administrative Zone. The Landsat images were obtained from the United State Geological Survey (USGS) archive and downloaded for free. They were previously, geometrically and radiometrically corrected and geo-referenced to real-world coordinates by USGS. The row images of the Illubabor Zone are shown in Figure 2.9 using the band combination of RBG 7, 4 and 2 for the Landsat images. The same quality level was available for the Spot 2007 image obtained from the Oromia Water Corporation Enterprise (OWCE). The row images for Spot 2007 are shown by band combination of RGB 3, 2, 1 in Figure 2.9. Before classification, the three bands of the Landsat images were layer stacked to merge them to produce a single image in ERDAS (2008) software. All images were already Orthorectified to a UTM (Universal Transverse Mercator) projection using the WGS (World Geodetic Systems) 84 datum by USGS. All three images were selected to have been taken during the dry season as this gives the best discrimination between land-cover classes (Reddy, 2008). The characteristics of the satellite data used in this study are summarized in Table 2.4.

Table 2.4: Characteristics of the satellite image data used in this study in Illbuabor Administrative Zone, SW Ethiopia.

Sensor	Spectral bands	Spatial resolution (Pixel size, m)	Spectral resolution	Date
Landsat TM	Band:2	30	0.45-0.52 μm	March 1986
	Band:4	30	0.63-0.69 μm	
	Band:7	30	0.52-0.60 μm	
Landsat +ETM	Band:2	30	0.45-0.52 μm	February 2000
	Band:4	30	0.63-0.69 μm	
	Band:7	30	0.52-0.60 μm	
Spot	Band:1	20	0.61-0.68 μm	February 2007
	Band:2	20	0.78-0.89 μm	
	Band:3	20	1.58-0.75 μm	

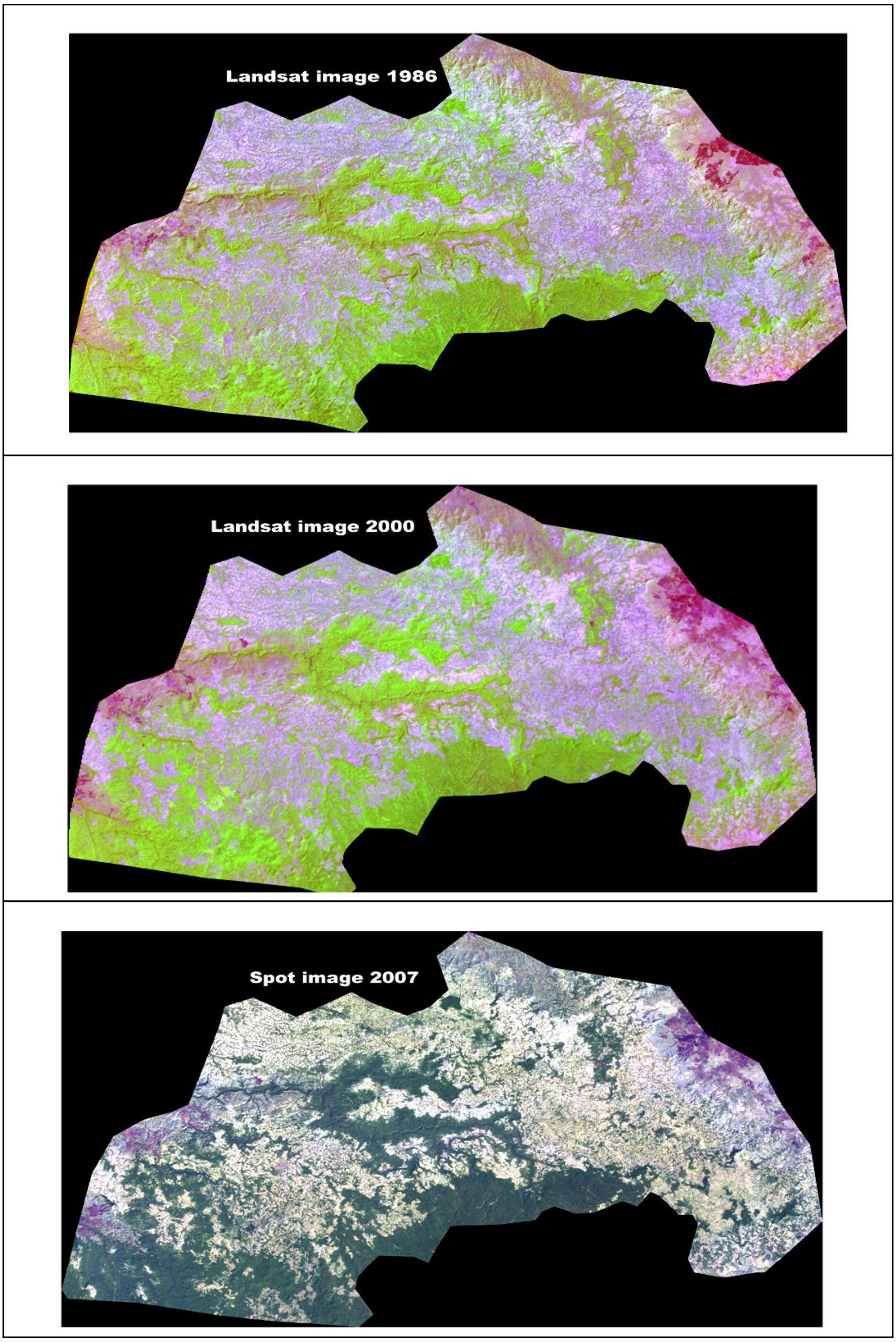


Figure 2.9 The raw satellite images masked to show the Illubabor Administrative Zone of SW Ethiopia which were used to produce land-use and land-cover maps for three years: 1986, 2000, 2007.

2.3.2. Land-use and land-cover cover classification

Land-use and land-cover classification requires a classification scheme and algorithm (Congalton and Green, 1999). The unsupervised classification method was selected to classify the Landsat images. It was used for the classification of the two Landsat images because use of unsupervised techniques is recommended for large areas that cover a wide and unknown range of land-cover types, and where landscape heterogeneity complicate identification of homogenous training sites (Cihlar, 2000; Krishna et al., 2010). In addition, there was no ground survey data with which to carry out a supervised classification. In this process for the two Landsat images the Iterative Self Organizing Data Analysis Technique (ISODATA) was applied to the eight classes of unknown land-covers with six iterations. The classified raster image was clumped using eight-neighbour rules (cells of the same edge that share a common edge or corner are considered part of the same patch) for delineating different land-use and land-cover patches (ERDAS, 2008; Geregel and Turner, 2002). A small minimum mapping unit of 0.5 ha was used to include smaller forest patches, narrow extensions or corridors as independent land cover. In contrast, supervised classification begins with defining the area that will be used as training sites for different land-cover classes. It involves a person assigning each raw pixel to prior-defined classes on the selected training sites (Kramer, 1997). The representative samples of each class that are assigned are then used by the computer to classify the whole image into the previously defined classes. Supervised classification was possible for the 2007 Spot image because it is a more recent image with high resolution and easy to identify different land-covers. The eight cover classes assigned to the 2007 image guided by a previous land-cover map obtained from Ministry of Agriculture (WBISPP, 2000), vector map (town, road, river), kebele map (small administrative unit at community level) and GPS ground data points taken post image classification in 2009 were: natural forest, farm and settlement, grazing land, wooded grassland, bare soil or rocky area, disturbed forest, plantation and wetland. Then, the eight classes obtained from the unsupervised classification were labeled using the result of supervised classification. The forest class was clear and easily identified and labeled. The other classes with similar spectral reflectance also fall nearby as a cluster. Such classes may represent the same category with a slight variation in reflection. The unsupervised and supervised classification was done using ERDAS 9.1 (ERDAS, 2008).

A reconnaissance field survey was carried out in 2010 using the supervised classification of the Spot image of 2007 and the unsupervised classifications of the Landsat images of 1986 and 2000. GPS reading were made for 355 land-cover classes labeled for both the supervised and unsupervised images, and their land-cover type checked in the field for verification. The 355 sample points were distributed between eight homogenous land-uses and land-cover types identified across the study area during the supervised classification. The sample points were collected using Garmin GPS 60. The sample size varied from 50 to 70 samples for each land-use and land-cover category (Jensen, 2005). These ground truth data were used in confirmation of cover classes identified with the supervised classification techniques, and the class identification and labeling process for images classified using the unsupervised classification technique. Even though the images classified using the unsupervised classification techniques were labeled using the results obtained from supervised classification techniques, they still needed class identification and labeling during the ground truthing.

2.3.3 Image reclassification

During ground survey I identified that some classes were locally misclassified and intermixed with neighboring classes and such misclassified cover-classes/pixels were identified using ground truth data points (Krishna et al., 2010). For example, most of the class "grazing land" was classified as "wetland", most of class "natural forest" was classified as "disturbed forest" or "plantation" and most of the class "farm and settlement" was classified as "bare or rocky area". Therefore, to improve on the classification of grazing land, wetland and grazing land was reclassified in both images (supervised and unsupervised) as grazing land since both areas are observed to be used for grazing land (pers. obs.); to improve on the classification of natural forest, disturbed forest and plantation was reclassified in both images as forest (again reflecting my personal observations in the field); to improve the classification of the farm and settlement cover class, bare or rocky area and farm and settlement was reclassified as farm and settlement (again based on pers. obs., see Section 3.2.3.2, Chapter 3 for details).

2.3.4. Quantification of landscape pattern

Landscape patterns are the structural and functional arrangements of landscape elements (patch, corridor and matrix) across landscapes (Gorodon, 1986; Forman, 1995; Turner et al., 2001). These arrangements develop continuously in space and time (Sih et al 2000). They are formed by the complex interaction between natural environments and human activities, resulting in the change of landscape spatial pattern and process (Turner et al., 2001). Description and quantification of landscape pattern using landscape metrics helps understanding of the relationship between pattern and process (Gergel and Turner, 2002) and improves the awareness of landscape patches associations and help for study of ecosystem functions, sustainable resource management and effective land-use planning (Matsushita et al., 2006) and restoration. Landscape metrics are quantitative indices used to quantify landscape pattern and their relationships (McGarigal and Marks, 1994; Herold et al, 2003). They are algorithms used for quantifying spatial characteristics of patches, classes of patches, or the entire landscape mosaic (McGarigal et al., 2004; Cabral et al., 2005). They were developed in the 1980s and include measures from information, percolation and fractal theory (Forman, 1995; Herold et al., 2003; Olsen, 2007). They numerically describe spatial configuration such as patch number, patch size, shape complexity, isolation, connectivity, contagion and interspersion of patches (Forman, 1995).

In practice, there are many landscape metrics used to analyse the spatial features of landscape pattern. In this study, I selected eight landscape metrics for the derivation of landscape pattern. The selection was based on their high frequency of use in landscape analysis by different authors (Bierwagen, 2007) as well as their close firm with the objective of the study. Table 2.5 lists the landscape metrics used and their definitions (see McGarigal and Marks (1995) for detailed definitions). These metrics are categorized into size metrics (Class Area (CA), Mean Patch Size (MPS), Core Area (COA) and Number of Patches (NUMP)), shape metrics (Area Weighted Mean Shape Index (AWMSI) and Area Weighted Mean Fractal Dimension (AWMPFD)), Patch Size Coefficient of Variation (PSCOV)) and edge metrics (Edge Density (ED)). The eight metrics were calculated for each of the four land-use/land-cover classes for each of the three images.

The Class area (CA, total area), for each of the four classes combine to define landscape composition (the percentage of the area comprised by each land-use/land-cover class (Weng et al., 2007)). This is important for comparing among landscapes of varying sizes (Gergel and Turner, 2002; McGarigal et al., 2002). Mean patch size (MPS) measures the average area of all patches for each class (Gergel and Turner, 2002). It is the major index of natural habitat pattern that affects biodiversity conservation and species composition and diversity (McGarigal and Marks, 1995) therefore it has a particular relevance as an indication of patch viability for the forest class. Core Area (COA) is the size of interior habitat that is not affected by the edge of the patch. It is important as an indicator of the area available for forest interior habitat-dependent species. The fragmentation and shrinkage of this area has an influence on interior biodiversity (Farina, 2007). An edge width of 50 m was used in this study to delineate the interior area for the calculation of COA. This width was chosen based on values proposed to define habitats for forest interior habitats and birds by Gergel and Turner (2002). The number of patches (NUMP) was used to quantify the number of individual patches of each of the four land-use/land-cover classes in the landscape and its increase or reduction indicates fragmentation or loss of habitat. The patch shape metrics (AWMSI and AWMPFD) are based on the perimeter to area ratio of each patch, and are indicators of the influence of edge effects within the patch, therefore (again) these were of particular interest for the forest class in the present study. AWMSI measures the complexity of average patch shape in the landscape compared to a standard shape (Mandelbrot, 1983; Gustafson et al., 1992). It equals the sum of patch perimeter (m) divided by the square root of patch area (m^2) for each patch in the landscape, adjusted by a constant to adjust for a square standard. The value varies between 1 and 2 and low values are derived when a patch has compact rectangular form with a relatively small perimeter to the area. AWMFD is also used as a measure of shape complexity of each patch (Mandelbrot, 1983; Gustafson et al., 1992). It is obtained by regressing 2 x ln (P/4) on ln(A), where P= perimeter and A= area of patch. A value of fractal dimension greater than 1 indicates a complex shape. Edge Density (ED) is a measure of the total length of the edge of all of the patches in each land-use/land-cover class divided by the total landscape area. A large value of edge density indicates a high level of human disturbance and fragmentation of the class. A number of packages for use with geographic information systems (GIS) permit the analysis and characterization of landscapes in terms of their patch composition and patch structure. Patch Analyst 4.2.1, a GIS extension which is designed to compute a variety of landscape metrics for categorical map patterns (Elkie et al., 1999), was used to calculate and compare the landscape metrics in the present study (Table 2.5). Patch Analyst was chosen because it provides integrated GIS environment for spatial analysis.

Table 2.5 Selected landscape indices used to analyze the landscapes of the Illubabor Administrative Zone in SW Ethiopia in the present study.

Metrics	Formula	Description
Class Area	CA = $\sum aij$, aj = area of patch j in the landscape of patch class i	Total area of the same class
Core Area	COA = $\sum_{j=1}^{n} aij^{c}$ aj = core are of patch j in the landscape of patch class i , where c is the buffer size	Area of interior habitat
Edge Density	ED = $\frac{E}{A}$, where E = total edge, A = total area	Perimeter-area ratio
Number of Patches	NP = ni, ni = total number of patches of class I in the landscape	Total number of patches in the landscape
Mean Patch Size	MPS = $\frac{\sum_{j=1}^{n} aij}{ni}$ aij the area of each patch j where ni is the number of patches of class i	Spatial configuration of patches
	AWMP S= $\frac{\sum(aij^{2})}{(aji)}$ aij = the area of patch j in the landscape class i	
Patch Size Coefficient of Variation	PSCOV = $\frac{\sum_{j=1}^{n}\left[aij-(\frac{\sum_{j=1}^{n} aij}{ni}\right]}{\frac{\sum_{j=1}^{n} aij}{ni}}$ or $\frac{PSSD}{MPS} X100$	Variability in patch size
Area weighted Mean Shape Index	M SI= $\sum_{j=1}^{n}\frac{\frac{pij}{2\sqrt{\Pi * aij}}}{ni}$ =pij=the perimeter of patch j in the landscape of patch class i, aij =the area of patch j in the landscape of patch class i, ni = number of patches in the landscape of patch class i, ln = natural logarithmic	Complexity of shape
Area weighted Mean Fractal Dimension	MPFD= $\sum_{j=1}^{n}\frac{\frac{2\ln pij}{\ln aij}}{ni}$ pij=the perimeter of patch j in the landscape of patch class i, aij = the area of patch j in the landscape of patch class i, ni = number of patches in the landscape of patch class i	Ratio of perimeter per unit area

Sources: McGarigal and Marks; 1995, Turner et al., 2001; Gergel and Turner, 2002; Farina, 2007

2.3.5 Vegetation sampling and plot layout

In order to obtain vegetation data in each sampled forest patch a plot-based inventory was carried out. The sampling was carried out in two stages: (i) a detailed inventory of forest interior habitat and (ii) a comparison of forest interior and edge habitats. All area within a forest patch at least 50 m from the nearest patch edge was classified as interior habitat. The interior coordinate of each patch was taken from the map using GIS and feed into GPS to navigate latter during sampling. Before establishing the sample plots the interior habitat was per-assessed by observation to identify the representative vegetation types. Then, a single main plot of 60 x 100 m was established in the interior habitat of each sampled forest patch (Figure 2.10). This was sub-divided into 15 sub-plots using a 20 x 20 m grid. Eight of these sub-plots, based on a systematic sampling regime, were used for vegetation inventory. Systematic sampling (described below) was selected as it maximizes the distance between plots and therefore minimizes spatial correlation among observations, thus increasing sampling efficiency and the accuracy of estimating the true population mean, though at the cost of reducing the accuracy of assessing true variance around that mean (Barry et al., 1995).

In order to investigate the mechanisms by which fragmentation of these forests (increase in the edge to area ratio of patches) influences their biodiversity, comparative data were required of the composition of the forest edge habitat (which is more liable to be exposed to grazing, trampling, microclimate and other environmental influences from the surrounding matrix). Thus, an additional eight plots of size 20 x 20 m were established systematically around the edge habitat of each patch (Figure 2.10) in forest patches that were large enough to accommodate both interior and edge plots (detail of plot layout described under section 5.1.2, chapter 5). The number of plots in the edge (eight) was set to sample as equal number as the 20x20 m sub-plots in the interior habitat. For both interior and edge habitat plots, five nested 5 x 5 m sub-sub plots were used within each 20 x 20 m sample unit, one at each corner and one at the centre to record saplings and seedlings of tree species and presence/absence of angiosperm herbaceous species.

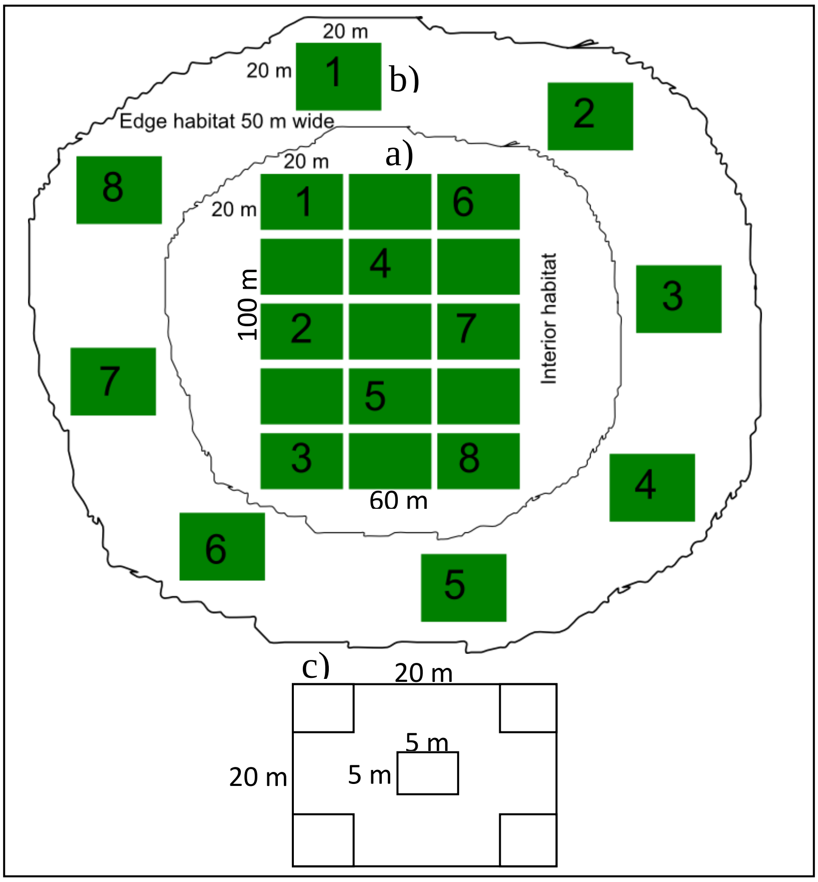


Figure 2.10 Layout of (a) 20 X 20 m sub-plots within the main plot in the forest interior habitat, (b) 20 X 20 m plots in the forest edge habitat, and (c) 5 X 5 m sub-sub-plots in the sampled forest patches in Illubabor administrative in SW Ethiopia. Numbers 1-8 indicate the sampled plots

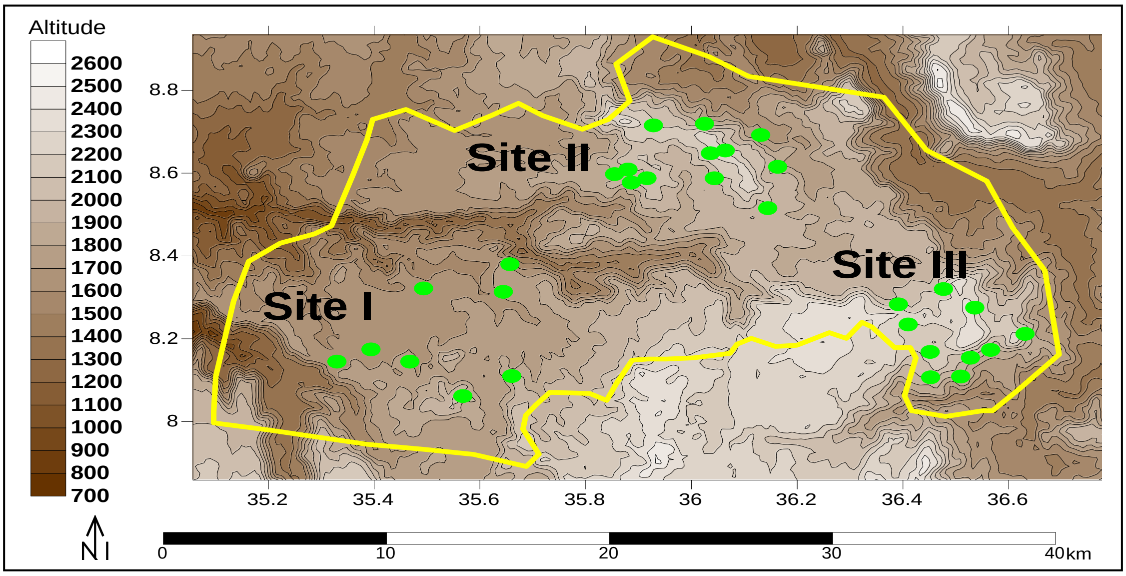


Figure 2.11 Spatial distribution of the sampled forest patches in the Illubabor Administrative Zone of south-west Ethiopia.

CHAPTER 3

DETECTION OF LANDSCAPE PATTERN CHANGE AS A RESULT OF HUMAN LAND-USE IN THE ILLUBABOR ZONE, SOUTH-WEST ETHIOPIA OVER THE PERIOD 1986 - 2007

Abstract

This study examines land-use changes in three landscapes of different land-use intensity in the Illubabor Administrative Zone, in south-west Ethiopia, and how these have affected landscape pattern and ultimately forest cover. Land-use change in the three landscapes was determined by interpretation of 1986 and 2000 Landsat images and Spot images of 2007.

The three sites (Site I, II and III) are located in different land-use characteristics. Site I is located in mid-altitude and its land-cover was dominated by natural forest. Site II and III are located in higher altitude and their pattern of land-use was very similar and dominated by farm and settlement. In two of the three sites the decrease in overall forest cover was only slight: in Site I from 56.6% in 1986 to 53.1% in 2007; in Site III from 22.1% in 1986 to 20.0% in 2007. However, in the third site, forest cover halved over the period: in Site II from 29.8% in 1986 to 14.8% in 2007. The cover of wooded grassland increased in Site I from 0.1% in 1986 to 0.4% in 2007, however it decreased in the other two sites: in II site from 1.6% in 1986 to 0.3% in 2007 and in Site III from 3.9% in 1986 to 1.7% in Site III. The cover of farming and settlement land decreased, to a variable extent, in all three sites over the period: in Site I from 36.8% in 1986 to a slightly lower 36.5% in 2007; in Site II from 48.0% in 1986 to 42.2% in 2007; in Site III from 53.9% in 1986 to 45.9% in 2007. In contrast, the cover of grazing land increased, to a very variable extent, in all three sites: in Site I from 6.5% in 1986 to 10% in 2007; in Site III from 20.1% in 1986 to 32.4% in 2007; and in Site II it more than doubled from 20.6% in 1986 to 42.7% in 2007. Comparing the trends in the two interim periods (1986-2000 and 2000-2007) there was no consistency of pattern amongst the three sites or land cover types. In sites I forest cover increased during 1986-2000 but decreased to a greater extent during 2000-2007, whereas in Site III it showed the opposite pattern, increasing during 1986-2000 but decreasing (to a greater extent) in 2000-2007. In

contrast, in Site II it showed a small decrease in 1986-2000 and a large decrease in 2000-2007. In both sites II and III the cover of farm and settlement land increased in the first period and then decreased to a greater extent in the second period, whereas in Site I it decreased in the first period and then increased in the second. Somewhat mirroring this, in sites II and III the cover of grazing land decreased during the first period and then increased to a greater extent during the second period, whereas it increased in both periods in Site I.

The number of patches of forest land increased in the first period in Site I, while increasing in sites II and III, however it decreased greatly in the second period in all three sites, so that in all there was a net decline over the whole 1986-2007 period: by 80% in Site I, 56% in Site II and 45% in Site III. In contrast there was a general increase in the mean size of forest patches. Mean patch size increased greatly in Site I over both periods, from 8.2 ha in 1986 to 37.9 ha in 2007 due to a high proportion of the smaller patches being converted to other land-use types. In contrast in both sites II and III it decreased during the 1986-2000 period, but increased during 2000-2007, to give a more modest net increase from 1986 to 2007 of 6.7 ha to 7.5 ha in Site II and 6.4 ha to 12.4 ha for Site III respectively. The area-weighted shape index of forest patches showed contrasting trends amongst the three sites and two periods. In Site II it decreased in both periods to give an overall 1986-2007 decrease from 6.8 to 4.1. In Site III after a slight increase there was a greater decrease giving an overall small change from 6.4 to 5.6. In contrast, in Site I (which started with a much larger area-weighted mean shape index) a very small decrease was followed by a large increase giving an overall change from 17.1 to 20.0. Forest patch edge density showed a different contrasting pattern across the periods and sites. In Site II it increased in both periods from 22.9 m ha^{-1} in 1986 to 54.7 m ha^{-1} in 2007. In both sites I and III it decreased during 1986-2000 but increased during 2000-2007, however the net 1986-2007 change was a decrease in Site I from 66.5m ha^{-1} to 52.2 m ha^{-1} but an increase in Site III from 33.1 m ha^{-1} to 41.7 m ha^{-1}.

The total core area of forest patches was defined in this study as the sum of all interior forest areas inside a 50 m buffer zone from the patch edge. This also showed an inconsistent trend across the sites and periods, though for each Site Its trend between the two periods was the exact opposite of that for edge density. The net change, from 1986 to 2007, in Site I was a small decrease from 17926 ha to 15399 ha, in Site III a similar small decrease from 6003 ha to 5524 ha, and in Site II a larger decrease from 7257 ha to 3647 ha . The number of patches containing any core area showed a slight increase over the same period in Site III from 351 to 375, however it decreased in Site I from 733 to 622 and greatly in Site II from 697 to 385.

These results indicate a complex landscape pattern of deforestation across the contrasting parts of Illubabor Zone. There was a general increase in land area under grazing, but this was only associated with a high rate of net deforestation in two out of three studied landscapes. However, there was clear evidence of forest fragmentation effects though their detail varied between landscapes. In the lower altitude area dominated by coffee cultivation forest became consolidated into fewer larger patches, while there was an associated decrease in patch edge density, there was also a small decline in core (patch interior) forest area. The two higher altitude landscapes showed contrasting trends, with one experiencing high rates of net deforestation and decrease in number of forest patches, with just a small increase in mean patch size, and a large increase in edge density and decrease in core (patch interior) forest area. The second upland landscape did not experience high net deforestation, but it did show considerable loss of smaller patches with the large decrease in patch number being paralleled by a doubling of mean patch size and just a modest increase in edge density and reduction in core forest area. While the differences between the landscapes can be explained by contrasts in processes of land-use change they share common underlying causes, including changes in population characteristics and in forest management responsibilities.

3.1 Introduction

Human land-uses have dramatically changed the landscape pattern by altering the relative abundances of natural and human-dominated habitats. In recent decades, the earth has experienced the fastest ever expansion and intensification of land cultivation (Leff et al., 2004), conversion to pasture and urbanization of land (Luck and Wu, 2002). These rapid land-use and land-cover changes caused by humans have become a major concern due to their influence on ecosystem processes and biodiversity (Lindenmayer and Fischer, 2006; Farina, 2007), global climate systems (Ramankutty et al., 2002) and landscape patterns (Gergel and Turner 2002). Thus, analyzing land-use/land-cover change (LULCC) is a fundamental step in order to capture and understand the impact of human activities on landscape pattern and process.

Landscape processes are heavily influenced by spatial pattern (Turner 1989), which includes type, number and spatial distribution and arrangement of natural and human modified landscape units (Ji, 2008). Changes to these spatial patterns are widely discussed by many authors (e.g. Forman. 1989; Turner et al., 2001; Gergel and Turner, 2002, Lindenmayer and Fischer, 2006; Farina, 2007) as they can influence ecosystem structure and function and there is a consensus that conservation and management is required at a landscape scale rather than at a patch level. In spite of this requirement for conservation and management at a landscape scale, anthropogenic effects continue to modify the natural habitat units with an ever increasing pace (Forman, 1995), and the rate of conversion of natural habitats due to direct human use is accelerating worldwide, particularly in regions such as Africa which exhibits rapid population growth (Turner et al., 2001; Malcolm et al., 2007). Large-scale land conversion, slash-and-burn agriculture for subsistence farming, grazing, settlement expansion, road construction, logging, and the expanding network of power lines all affect the spatial pattern of natural habitats (FAO, 2003; Hennig, 2006).

These types of human influence can affect landscape pattern in three main ways. Firstly, they reduce the total area of forest by removing forest cover permanently (Kramer, 1997; FAO, 2003). As a result some forest types have disappeared locally and there is a reduction in the total amount of habitat, thereby affecting landscape pattern and process. Secondly, the spatial structure of forests is altered by dividing forest cover into fragments (Dale et al., 1994; Forman, 1995, Turner et al., 2001). Fragmentation results in the disruption of existing ecological connections between spatially separated habitats (Gergel and Turner, 2002; Lindenmayer and Fischer, 2006). Thirdly, land-use activities between fragmented patches largely separate them from each other and increase isolation (Saunders, 1991, Turner, 1996; Bender et al., 2003; Lindenmayer and Fischer, 2006). An isolated patch can also be subjected to edge effects from the surrounding matrix (Annette and Martin, 2004) and the amount of edge habitats in an isolated patch can be increased relative to interior habitats (Arroyo-Rodríguez and Mandujano, 2006).

The combination of the above pressures has resulted in the decline, loss and fragmentation of forest habitats in large areas of Africa impairing the ecological functionality of landscapes. For example, between 1990 and 2000, Africa lost about 52 million hectare of forest, accounting for about 56% of the global reduction in forest cover through deforestation and fragmentation (FAO, 2003). The net loss of forest area remains high at an estimated four million hectares per annum between 2000 and 2005 (FAO, 2006). The recent FAO (2010) reports also indicate that the forest reduction between 2005 and 2010 continued in the same trend. Despite these facts, frequent monitoring of the changing landscape pattern and reduction in forest resources is lacking in African countries, making trend analysis at a landscape level difficult for conservation and restoration planning.

Natural forest habitats in Ethiopia are intensively deforested and extensive afromontane forest remnants only exist in the south-west and south-east of the country (EFAP, 1994) Figure 3.1. In recent years, however, there has been rapid agricultural expansion around and inside these remaining forests due to the influx of people from central, eastern and northern parts of the country due to planned government resettlement programmes and spontaneous migration (Reusing, 1998, 2000; Tadesse et al., 2001; Mekuria, 2005; Mengistu, 2005). This has contributed to the loss of a significant area of natural habitat and fragmentation of much of the remaining forest (Tadesse, 2003) resulting in a changed landscape pattern from predominantly natural to highly human-modified.

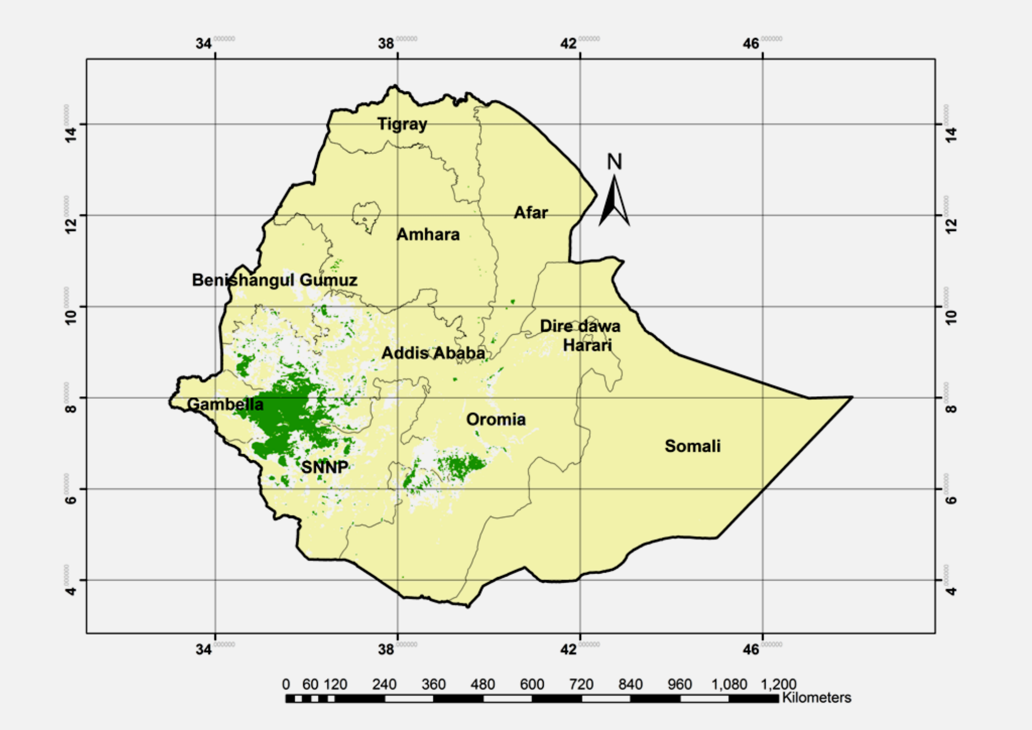


Figure 3.1: Map of Ethiopia with remaining afromontane forest distribution predominantly in the south-west part of the country: source WBISPP, 2000

These landscape pattern changes in Ethiopia and their possible effects have not yet received any major research attention. Although there have been some studies on LULCC in Ethiopia (e.g. Reusing, 1998; WBISPP, 2000; FAO, 2001; Reid et al., 2000; Bedru, 2007), these have focused only on landscape cover proportion. None of them report on landscape composition and structural characteristics of remaining patches within the larger landscape. To improve this evidence base, a major stage in resolving natural habitat fragmentation has to be the identification and analysis of the LULCC which will require assessment at the landscape scale to determine changes both in proportion and structure at landscape, class and patch levels. An exemplar landscape for such analysis is the Illubabor Zone in south-west Ethiopia. This area has undergone rapid land-use change as a result of socio-cultural change, unprecedented population growth and unsustainable forest use. However, no research has been carried out there which analyses the spatial character of forest habitat patches at landscape, class and patch level.

The main objective of this study is to evaluate the impact of land-use change on natural land-cover pattern, especially of forest, within Illubabor Zone, south-west Ethiopia. The specific objectives were to determine the nature and extent of changes in land-use in three landscapes of different land-use characteristics between 1986 and 2007, to establish the causes of these changes and to assess their impact on forest cover and its spatial pattern. To achieve these objectives, the change in land-cover proportion, and forest patch area, shape, number and edge density were investigated by analyzing land-use/cover changes at landscape and class level.

3.2 Materials and methods

3.2.1 Data used for land-use and land-cover map production

Landsat images from 1986 and 2000 and a Spot image from 2007 were used for LULCC map production (see Table 2.3 for their details and acquisition date). Landsat images were obtained from the United States Geological Survey (USGS) archive and downloaded for free (www.glovis.usgs.gov). They were previously geometrically and radiometrically corrected by USGS. The same quality level was available for the Spot 2007 image which was obtained from the Oromia Regional State Water Corporation Enterprise (OWCE). Both Landsat images were geo-referenced to Universal Transverse Mercator (UTM) projection using the WGS (World Geodetic Systems) 84 datum by USGS. The three bands (2, 4 and 7) of each Landsat image were layer stacked to merge them together to produce a single image using ERDAS 9.1 (ERDAS, 2008). The Spot image was obtained in a layer stacked using three available bands (RGB 3, 2 and 1). All images are available for each month of every year of the study area. From the available images, those in the dry seasons were selected to better discriminate land-cover classes. The dry season of the study area is from November to March (Woubshet, 2000, Chapter 2). Images of the area containing 0% cloud cover were selected for each year for land-use production.

3.2.3 Land-use and land-cover classification and accuracy assessment

3.2.3.1 Land-use and land-cover classification

The detailed description of supervised and unsupervised classification technique used in this study was given under section 2.3.2 (chapter 2). Classification of the 1986, 2000 and 2007 images was performed on a sub-scene I clipped from the full image on the basis of a vector frame covering the study area (Figure 2.9, chapter 2). Then, the classified images were exported to ARC GIS 9.2 (ESRI, 2007). Using its GIS tools, different land-use and land-cover raster polygons identified on the satellite images were converted to vector polygons and then to shape formats. The land-use and land-cover classes identified and labeled for each image were taken to the field and GPS reading were made for 355 land-cover classes or polygons created for all images and their land-cover type checked in the field for verification. As described under section 2.3.2, these 355 sample points were distributed between eight land-uses and land-cover types identified across the study area. The sample points were collected using Garmin GPS 60 in 2010 and 50-70 sample points were selected for each land-use and land-cover category (Jensen, 2005). Land-use and land-cover polygons of 1986 and 2000 labeled using results from supervised technique of image 2007 were compared with land-cover from ground-truthing data.

3.2.3.2 Image reclassification

Initial classification of the images using the two classification methods (unsupervised for 1986 and 2000 Landsat images and supervised classification for 2007 Spot image) revealed a wide range of spectral confusion among land-cover types. Spectral confusion can occur when several land-cover classes have similar spectral response (Yang and Lo, 2002). To reduce this problem, I closely examined to highlight major area of misclassified land-covers. This was achieved with the aid of existing land-cover map produced by WBISPP (2000) and sample points taken during ground survey. Interviews of local elders and farmers were also conducted during ground survey for five sample points where a clear relationship between the Landsat and Spot images could not established. Most farm and settlement pixels were classified as bare soil or exposed rocky area; most natural forest pixels were classified as plantation and disturbed forest; most grazing land pixels were classified as wetland. During ground survey I also recognized that there was no clear boundary between undisturbed and disturbed forests. Therefore, to improve on the classification the following cover types were reclassified into three land-cover classes in both unsupervised and supervised images: (a) bare soil or exposed rocky area and farm and settlement, (b) wetland/swampy area and grazing land, (c) undisturbed forest, disturbed forest and plantation. The fourth class, wooded-grassland land-cover, was classified correctly during initial classification. Finally, the eight land-use/cover polygons initially identified were combined after field verification and the four basic classes of land-use/cover were identified from the classified images for all years (Table 3.1).

Table 3.1: Land-use/cover classes and definitions used in this study

No	Land-use/cover class	Definition
1	Farm and settlement	Agricultural land currently under cultivation, areas cleared of vegetation which have not yet been planted, urban and rural settlements, gravel as well as asphaltic roads, bare soil or exposed rocky area.
2	Forest	High canopy forest, including montane evergreen forest and closed evergreen lowland forest, disturbed forest, highly coffee managed forest, plantation forest (*Cuperssus lucitanica, Pinus spp* and *Eucalyptus spp* forests).
3	Grazing land	Areas under cover of grasses and other herbaceous vegetation, wetlands (normally used as grazing ground in Ethiopia), may include some land under fallow after crop cultivation.
4	Wooded grassland	Areas with both a high percentage of grasses and scattered trees and shrubs.

3.2.3.3 Accuracy assessment

Classification accuracy associated with the maps is critical to reliably characterize spatial pattern, detect change, and relate pattern to process (Iverson, 2007) since the land-use/cover maps derived from remote sensing data always contain some errors due to factors which range from classification technique to method of satellite data capture (Shao and Wu, 2004; Muzien, 2006; Shao and Wu, 2008). These errors could spread to subsequent landscape pattern analysis. Without knowing the magnitude of these errors in the landscape data, the characterization of landscape pattern is not reliable. The inaccurate result obtained may affect the inferences of pattern-process relationships and thus recommendations for conservation and restoration planning (Shao and Wu, 2008). In order to use the derived land-use/cover maps for pattern change analysis, therefore, the errors need to be quantified in terms of classification accuracy.

The accuracy assessments of both supervised and unsupervised techniques were made through a confusion or error matrix (Hasmadi et al., 2009). A confusion matrix usually represented in terms of producer's accuracy, user's accuracy and overall accuracy (Shao and Wu, 2008) and the Kappa coefficient (Congalton and Green, 1999, Tso and Mather, 2009). I compared pixels categorized under different land-cover polygons by software to the same Site In the field (Hasmadi et al., 2009). This process was used to estimate the accuracy of the image classification by comparing the classified map with the reference map based on field sample points (Caetano et al., 2005). The technique was used to provide a statistical and analytical approach to examine the accuracy of the classification. The producer's accuracy was calculated as the percentage of each land-use/cover type on the ground correctly classified in the 1986, 2000 and 2007 maps. It was calculated as the ratio of the number of correctly classified pixels for a class to the total number of pixels for that class from the field survey (column total). User's accuracy was calculated by dividing the number of correctly identified pixels to the classified total (row total which includes both correctly classified and misclassified samples) (Foody, 2002). The overall accuracy was computed by dividing the total correct number of pixels (the total diagonal value) by the total number of pixels in the matrix.

Kappa coefficient (K) is another measurement used to assess maps accuracy in this study. It is a method for validating LULCC results by using the random independent validation points (Foody, 2002). It expresses proportionate reduction in error generated by a classification process compared with the error of a completely random classification (Congalton and Green 1999). The Kappa coefficient proposed by Congalton and Green (1999) was calculated as:

$$\kappa = \frac{N\sum_{i=1}^{r} X_{ii} - \sum_{i=1}^{r}(X_{i+} \times X_{+i})}{N^2 - \sum_{i-1}^{r}(X_{i+} \times X_{+i})}$$

where: K = Kappa coefficient, r = the number of rows in the error matrix, X_{ii} = the number of observations in rows i column i (along the major diagonal), X_{i+} = is the marginal total of row i (right of the matrix), X_{+i} = the marginal total of column i (bottom of the matrix), N = the total number of observations included in the matrix (total GPS points for ground truthing).

The value of Kappa is always less than or equal to 1. A value of 1 implies perfect agreement in this case the maps classification accuracy would be precise. Most studies indicate that values of Kappa is usually less than 1 but value < 0.20 is poor agreement and values between 0.7 and 1 is very good agreement. Though the classification accuracy is critical for landscape pattern analysis, there is no universally applicable standard based on which the adequacy of classification accuracy can be quantified (Shao and Wu, 2008). However, the producer’s and user’s accuracy values ranging from 60% to 100% are accepTable (Shao and Wu, 2008).

I used ground truthing of 355 sample points and existing land-use and land-cover maps to assess the classification accuracy of the land-use and land-cover maps in this study. The result of my classification accuracy assessment indicates that both producer’s accuracy and user’s accuracy for each of the three images are range from 60% to 100% (Tables 3.2). Therefore the reliability of the results that will be obtained from landscape pattern analysis is expected to be high. When individual land-use and land-cover of each image year is considered, in 2007 the highest producer’s accuracy exists in the land-use/cover class of farm and settlement, while the highest user’s accuracy exists for natural forest (Table 3.2). The farm and settlement class has such high classification accuracy as it was the dominant matrix and continuous landscape in 2007 (Figure 3.6). In contrast, for 2000 the highest producer’s accuracy was for natural forest and user’s accuracy was for grazing land (Table 3.2). In 1986, the highest producer’s accuracy was for natural forest and user’s accuracy for wooded grassland (Table 3.2). The overall accuracy was 79.1% for the TM (1986) data, 82.8% for +ETM (2000) data and 80.6% for Spot (2007) data. The Kappa Coefficients for 1986, 2000 and 2007 were 0.71, 0.77

and 0.73 respectively. The large values of Kappa coefficient (> 0.7) for each of the three years indicate that there is a very good agreement between producer's (developers of the map) and user's accuracy. Therefore, the classification carried out in this study produced an overall accuracy and Kappa coefficient that fulfils the accuracy threshold (> 60%) required for further LULCC analysis.

Table 3.2 Classification accuracy assessment for Landsat Satellite images of the years 1986 and 2000, and Spot image of 2007.

	Landsat 1986		Landsat 2000		Spot 2007	
Land-cover class	Producer's (%)	User's (%)	Producer's (%)	User's (%)	Producer's (%)	User's (%)
Forest	95.0	71.9	91.0	76.5	83.0	96.5
Grazing land	60.0	75.0	72.0	90.0	70.7	62.4
Farm &settlement	77.5	83.8	84.2	85.6	87.5	78.9
Wooded grassland	78.0	92.2	80.0	82.1	75.0	90.0
Overall accuracy	79.1%		82.8%		80.6%	
Kappa statistic	0.713		0.763		0.734	

3.2.3.4 Landscape pattern analysis by landscape metrics

After land-use and land-cover classification and accuracy assessment was done for the larger part of the Illubabor Administrative Zone study area, specific areas of interest were clipped for detailed study. The first stage was to classify the study area into two land-use categories based on dominant productive land use in 2007: coffee or mixed farming (Figure 2.11). Based on data of the land area under coffee and food crop production collected from the Zone Agricultural Office, it was decided to locate one site for detailed study in the land category dominated by coffee cultivation. The delimitation of this study site was determined by altitude. Because of the large heterogeneous area within the zone predominantly under mixed-farming food crop production at higher altitudes, it was decided to locate two sites for detailed study in this land category.

The second stage was to define the spatial scale of the map. The concept of spatial scale in landscape ecology encompasses both extent and grain (Forman and Godron, 1986): spatial extent is the total area of the map being considered for analysis; grain is the size of the individual units of observation, i.e. land-cover classes. Extent and grain define the upper and lower limits of resolution of the study (Wiens, 1989, Gergel and Turner, 2002) and was dictated by the scale of the maps obtained by the processing of Landsat and Spot imagery (1: 250,000). The extent of the newly clipped land-use and land-cover maps from whole study area for each site for spatial analysis was purposefully aimed to cover most of the area encompassing the sampled forests for vegetation analysis. The boundary of each map was defined by a vector polygon created by edit tools in GIS. I created the polygons using edit tools purposely for each site and overlaid on each land-cover map and subjectively increased the size by dragging all sides of the polygon until it covers the whole sampled forest patch and each gives an identical area before the maps are clipped. These created equal-area polygons help to analyze spatial data in similar area in all sites (Wu, 2004). Making spatial analysis in equal areas of polygon reduces the problems that may arise due to different scales of analyzed spatial data because this has been shown to greatly influence values of pattern metrics (e.g. Turner et al., 1989; Gustafson and Parker, 1992; Wickham and Riitters, 1995; Saura and Martinez-Millan, 2001; Wu, 2004).

The third step was the definition of landscape unit or land-cover patches within the defined map boundaries. The land-cover patches were defined by the pixels belonging together to one land-cover class of the classified images and delimited by patch boundary. Two adjacent patch polygons can have the same meaning and the boundary between the two polygons should be dissolved (Rempel, 2012), otherwise the landscape pattern detected using landscape metrics will have little meaning. In the present research, before analysis of change at each site for each year, the land-use and land-cover polygons with similar meaning were dissolved by dissolve feature described by Rempel (2012) and available in Patch Analyst (ArcGIS extension) to remove boundaries between polygons. This method is useful to reduce the number of patches that will be artificially high caused by undissolved boundaries of same-class polygons (Rempel, 2012).

The location of the three sites is shown in Figure 2.11. Site I covers an altitude range of 1500 to 1990 m a.s.l. (Table 4.1, Chapter 4). Its land cover in 2007 was dominated by natural forest (53.1%), of which a high proportion was revealed to be subject to understorey coffee cultivation during the field survey. Site II covers an altitude range of 1878 to 2319 m a.s.l. Its land cover in 2007 was dominated by farm & settlement land cover (42.1%). During the field survey it was ascertained that this land was dominated by the cultivation of field crops such as teff, wheat, and barley as well as some coffee. In addition, 14.8% of this site was classified in 2007 as natural forest, and the field visit revealed that this was also subject to some understorey coffee cultivation. Site III covers an altitude range of 2000 to 2422 m a.s.l. Its land cover in 2007 was also dominated by farm & settlement land cover (45.9%). In the field survey the pattern of its land use was found to be very similar to Site II including understorey coffee cultivation in some areas of natural forest (which occupied 20% of the Site in 2007). Informal interviews with more than five elders of the long-established local communities were conducted during the field work at each of the three sites. The elders at Site I reported that some of its current population migrated to the area before 1986 either spontaneously or during a planned resettlement programme. The elders at Site II reported that about a quarter of its current population were members of households which migrated to the area during a resettlement programme in ca. 1990, 20 years ago. The elders at Site III reported that some of its current population migrated to the area in approximately 2003. According to the elders, the land-use practices of these migrants are different from the indigenous people living in the area for a long time in that they remove forests to cultivate crops and for expansion of settlement area while the indigenous people use the forest only to collect some non-timber forest products, i.e. *Coffea arabica*, *Piper capense* and honey. The elders added that the indigenous people also use wood and lianas for house and beehive construction.

3.3 Results

3.3.1 Landscape pattern change in Illubabor Zone, ern Ethiopia from 1986 to 2007

3.3.1.1 Land use/land cover in Site I

The maps of the distribution of the four land-use/land-cover (LULC) classes of 1986, 2000 and 2007 for Site I are displayed in Figures 3.4, 3.5 and 3.6; Figure 3.2 provides details of the area and proportion of different LULC classes in each year and Figure 3.3 shows the change in these areas between the years.

In Site I there were only small changes in the area of each of the four LULC classes over the 21 year study period (Table 3.3). Forest remained the completely dominant class (> 50%) throughout the period (Figure 3.2): a small increase in area of 3.8% from 1986 to 2000 was followed by a larger decline of 9.7% between 2000 and 2007 to give a net decline in area of 6.2% (Table 3.4). The second most abundant LULC class remained farm and settlement throughout the period, with a very similar area (37%) at the start and end. Grazing land and wooded grassland both increased in area in both time periods, giving each a total increase of 53% and 526% respectively. Therefore, while the forest area declined between 2000 and 2007 that of all the other three classes increased. In this sense the forest constituted the “matrix” component of the landscape throughout the period. The fluctuation in proportion of farm and settlement LULC class may be due to fallow land inclusion or exclusion under farm each year.

Table 3.3 Area occupied by each land use/land cover class in hectares and as a percentage of the total land area in the years 1986, 2000 and 2007 in Site I in Illubabor Zone outhwestern Ethiopia

	1986		2000		2007	
Land-use and land-cover classes	Area					
	ha	%	ha	%	ha	%
Forest	26980.2	56.6	28005.9	58.78	25300.4	53.1
Grazing land	3114.2	6.5	3353.0	7.0	4776.6	10.0
Wooded grassland	29.3	0.1	70.0	0.1	183.3	0.4
Farm and settlement	17550.9	36.8	16218.8	34.0	17394.2	36.5
Total	47,647.7	100	47,647.7	100	47,647.7	100

Table 3.4 Change in area of land-use/land-cover categories in time periods between 1986 and 2007 in Site I in Illubabor Zone southwestern Ethiopia

Land-use and land-cover classes	1986-2000	% change	2000-2007	% change	1986-2007	% change
Forest	1025.7	3.8	-2705.5	-9.7	-1679.8	-6.2
Grazing land	238.8	7.7	1423.6	42.5	1662.4	53.4
Wooded grassland	40.7	138.9	113.3	161.9	154.0	525.6
Farm and settlement	-1572.8	-7.6	1416.1	7.2	-156.8	-0.9

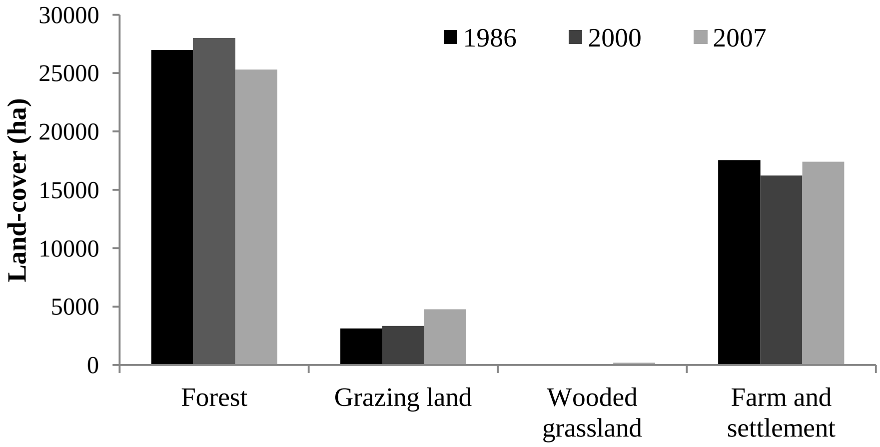


Figure 3.2 Total area of land (ha) occupied by each land-use/land-cover class in 1986, 2000 and 2007 in Site I

Figure 3.3 Net change in area (ha) of each land-use/land-cover class in each of three periods (1986-2000, 2000-2007, 1986- 2007) in Site I

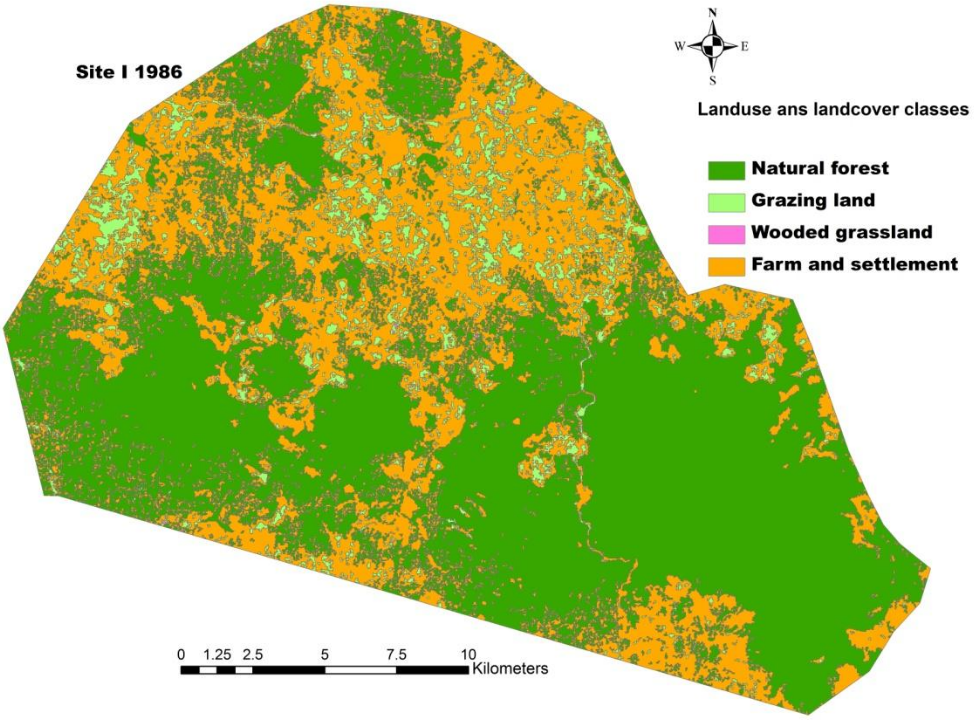


Figure 3.4 Final land-use and land-cover map of Site I in 1986

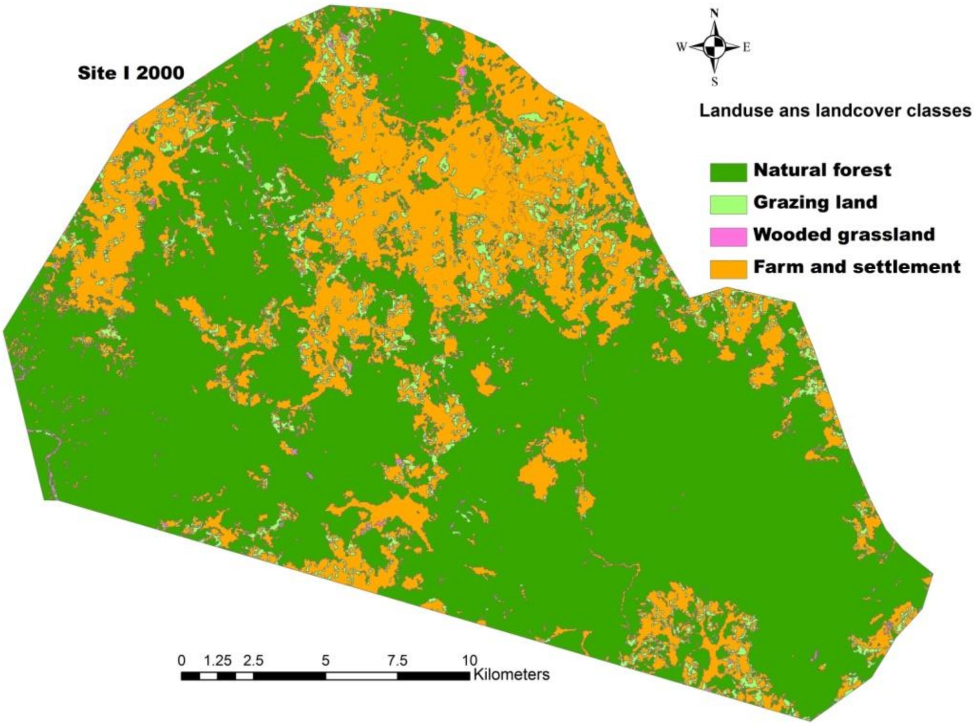


Figure 3.5 Final land-use and land-cover map of Site I in 2000

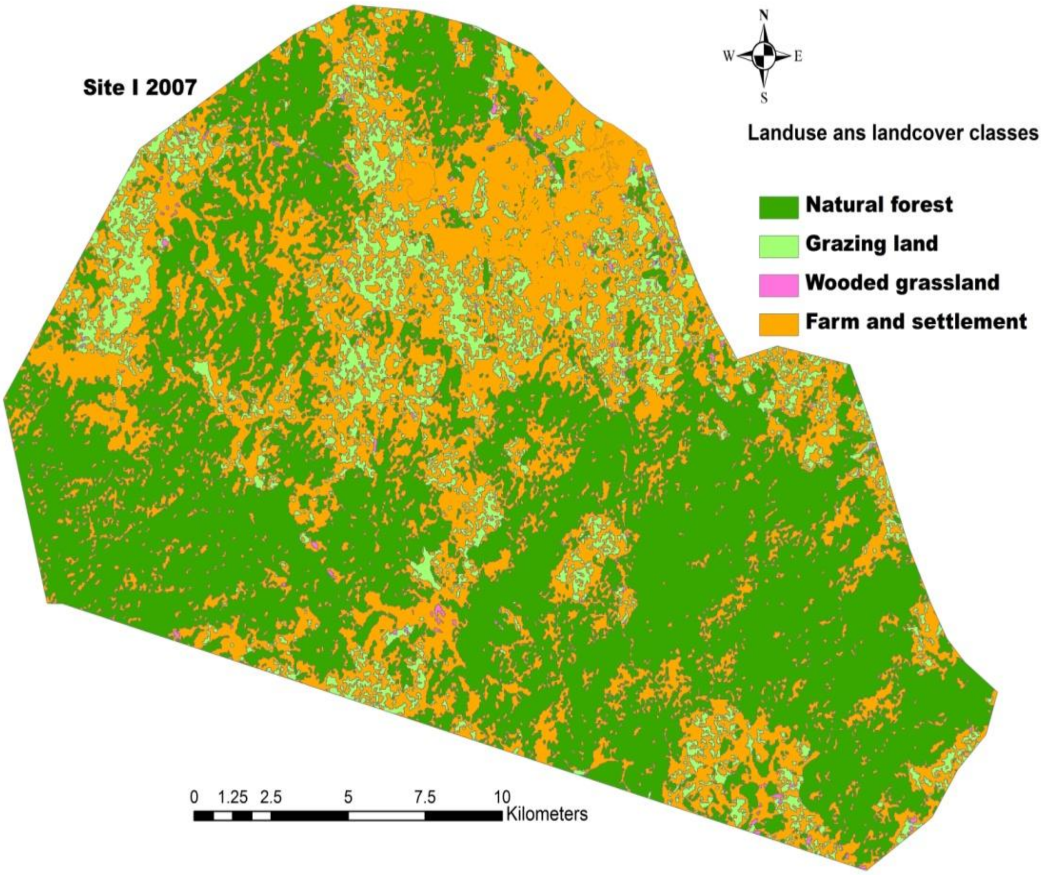


Figure 3.6 Final land-use and land-cover map of Site I in 2007

3.3.1.2 Land use/land cover in Site II

Land cover changed much more in Site II than in Site I during the study period (Figures 3.9, 3.10 and 3.11). The proportion of each land-use and land-cover is indicated in Figure 3.7. Farm and settlement started as the dominant land cover (48% of the total land area) in 1986, increased greatly to 62% in 2000, before declining to an even greater extent (to 42%) by 2007 (Table 3.5). In contrast, grazing land decreased greatly between 1986 and 2000 before a huge increase (of 285%, to 43% of the total land area) by 2007 (Table 3.6). Forest cover declined in both periods, especially the second (Figure 3.8), producing an overall halving of its area from 30% of land area in 1986 to 15% in 2007. Wooded grassland declined greatly in the first period and then remained at only 0.3% of land area.

Table 3.5 Proportion of land-use/cover class of the study area at Site II for the year 1986, 2000 and 2007

Land-use and land-cover types	1986		2000		2007	
	Area					
	ha	%	ha	%	ha	%
Forest	14186.4	29.8	12664.2	26.6	7052.0	14.8
Grazing land	9817.3	20.6	5287.4	11.1	20351.5	42.7
Wooded grassland	778.4	1.6	137.4	0.3	158.8	0.3
Farm and settlement	22865.5	48.0	29558.7	62.0	20085.4	42.2
Total	47647.7	100	47647.7	100	47647.7	100

Table 3.6 Land-use and land-cover area and their change at Site II between 1986 and 2007

Land-cover change between 1986 & 2007						
Land-use and land-cover classes	1986-2000	% change	2000-2007	% change	1986-2007	% change
Forest	-1522.2	-10.7	-5612.2	-44.3	-7134.4	-50.3
Grazing land	-4529.9	-46.1	15064.1	284.9	10534.2	107.3
Wooded grassland	-641.0	-82.3	-21.4	15.6	-619.6	-79.6
Farm and settlement	6693.2	29.3	-9473.3	-32.1	-2780.1	-12.1

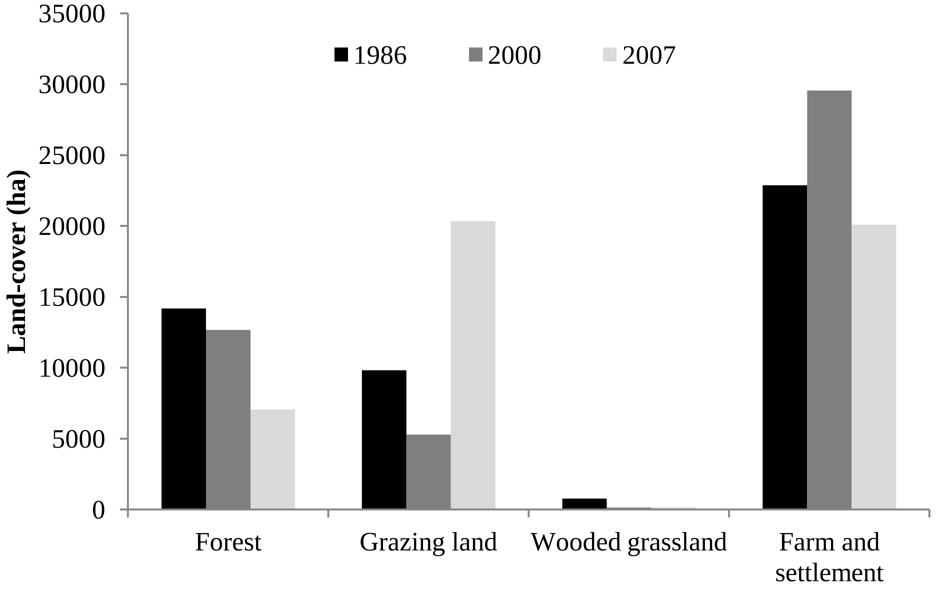


Figure 3.7 Total area of land (ha) occupied by each land-use/land-cover class in 1986, 2000 and 2007 in Site II

Figure 3.8 Net change in area (ha) of each land-use/land-cover class in each of three periods (1986-2000, 2000-2007, 1986- 2007) in Site II

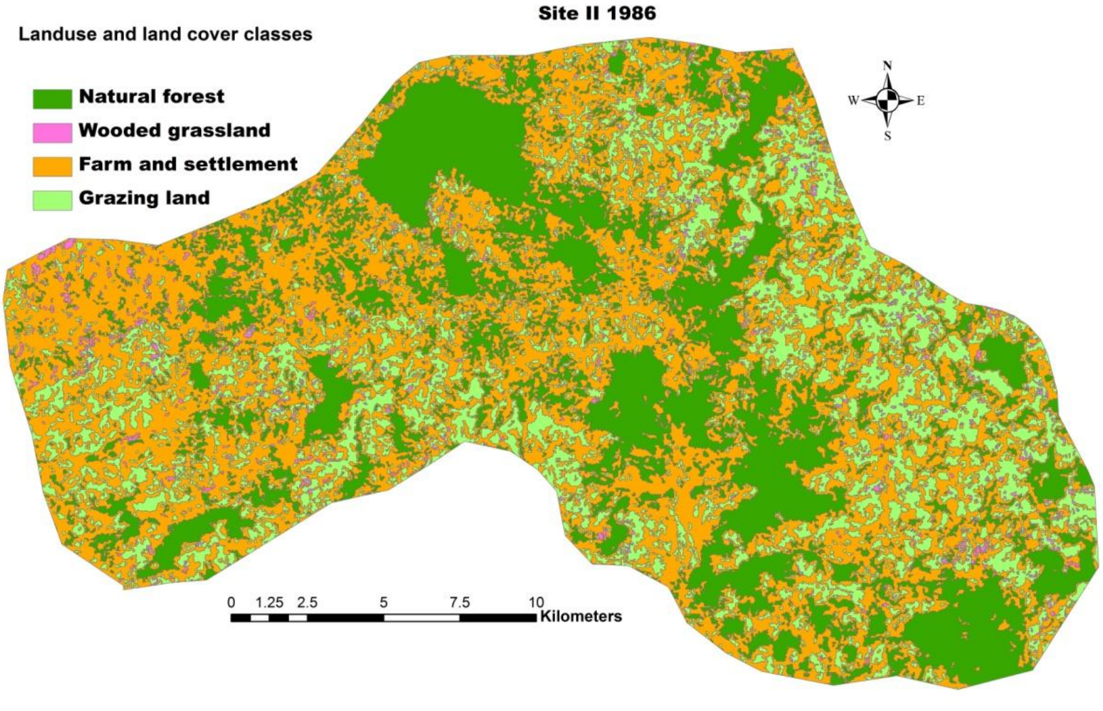


Figure 3.9 Final land-use and land-cover map of Site II in 1986

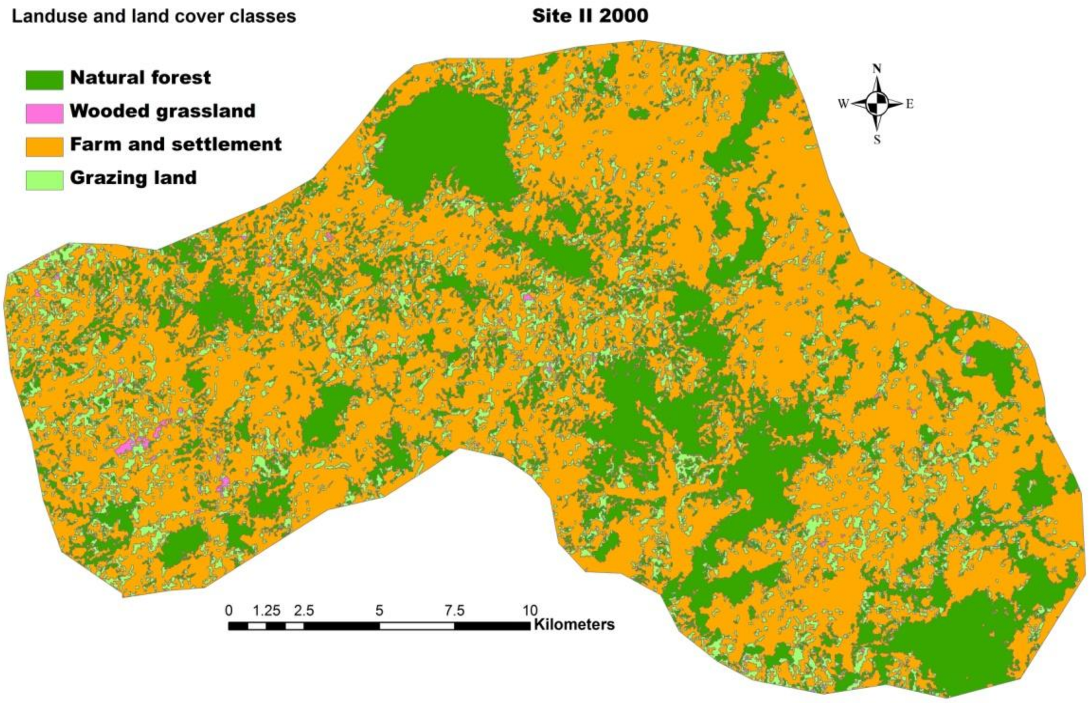


Figure 3.10 Final land-use and land-cover map of Site II in 2000

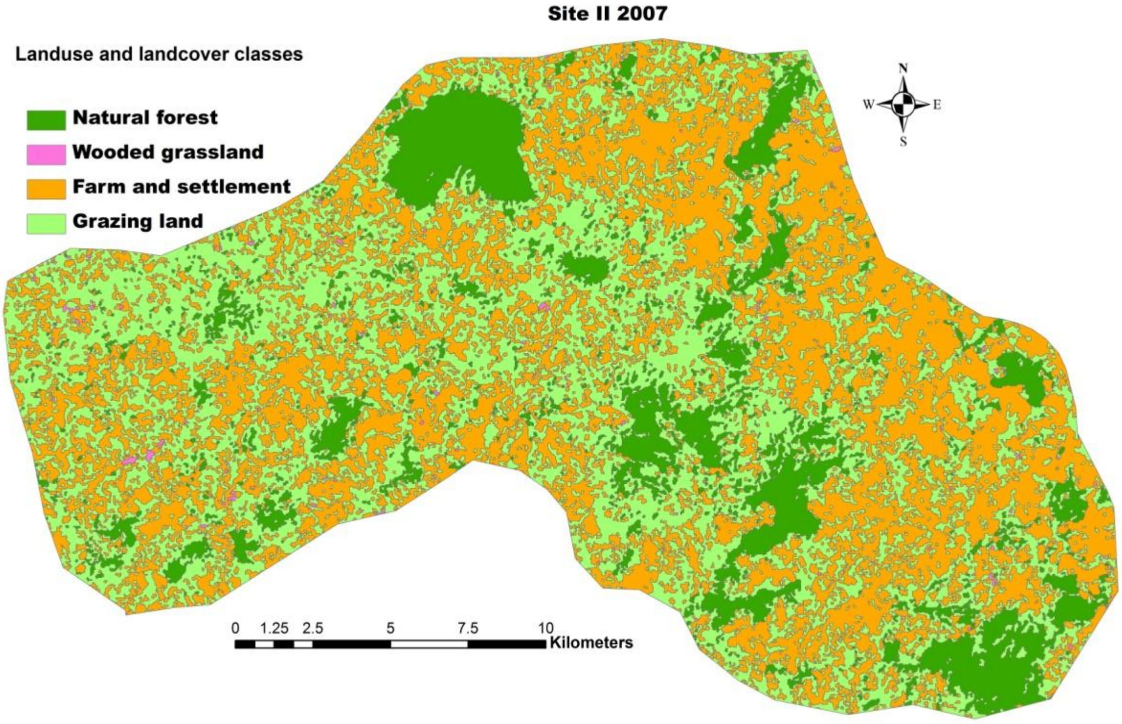


Figure 3.11 Final land-use and land-cover map of Site II in 2007

3.3.1.3 Land use/land cover in Site III

The land-use and land-cover distribution maps for Site III are displayed in Figures 3.14, 3.15 and 3.16. The cover proportion and net cover change of each land cover class is indicated in Figures 3.12 and 3.13 respectively. The magnitude of changes in area of the LULC classes in this site was intermediate between those of Sites I and II. The pattern of trend between the two periods for forest was more similar to Site I, but for farm and settlement, for grazing land and for wooded grassland it was more similar to Site II. In summary, farm and settlement remained the dominant land cover throughout (Table 3.7) but following an initial increase its area declined greatly (to 46%) in 2007. In contrast, after an initial decline, grazing land increased massively (to 32%) in 2007 (Table 3.7). Like farm and settlement, forest initially increased in area before a large decrease (to 20%) in 2007. The net cover change of these land-cover classes also declined (Table 3.8). The forest cover class gained area during the first period and lost large amount of area during the second period with a net area loss of 9.4% in 2007 (Table 3.8)

Table 3.7 Proportion of land-use/cover class of the study area at Site III for the year 1986, 2000 and 2007

	1986		2000		2007	
	Area					
Land-use and land-cover classes	Ha	%	Ha	%	Ha	%
Forest	10522.5	22.1	13284.3	27.9	9530.2	20.0
Grazing land	9588.9	20.1	6579.3	13.8	15439.6	32.4
Wooded grassland	1841.0	3.9	282.6	0.6	802.3	1.7
Farm and settlement	25695.3	53.9	27501.5	57.7	21875.5	45.9
Total	47647.7	100	47647.7	100	47647.7	100

Table 3.8 Land-use and land-cover area and their change at Site III between 1986 and 2007

	Land-use and land-cover change between 1986 & 2007					
	1986-2000	% change	2000-2007	% change	1986-2007	% change
Forest	2761.8	26.2	-3754.1	-28.3	-992.3	-9.4
Grazing land	-3009.6	-31.4	8860.3	134.7	5850.7	61.0
Wooded grassland	-1558.4	-84.6	519.7	183.9	-1038.7	-56.4
Farm and settlement	1806.1	7.0	-5625.9	-20.5	-3819.8	-14.9

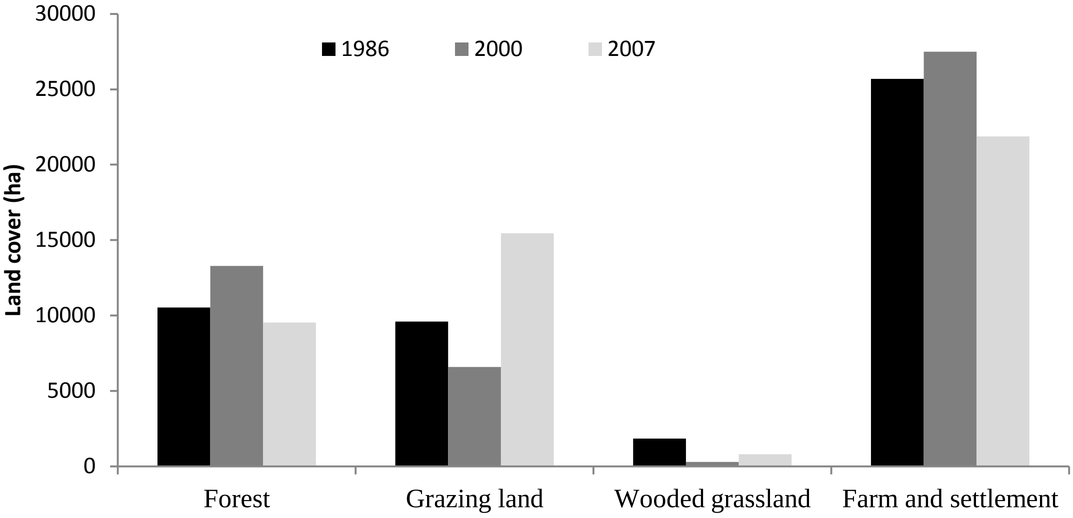


Figure 3.12 Total area of land (ha) occupied by each land-use/land-cover class in 1986, 2000 and 2007 in Site III

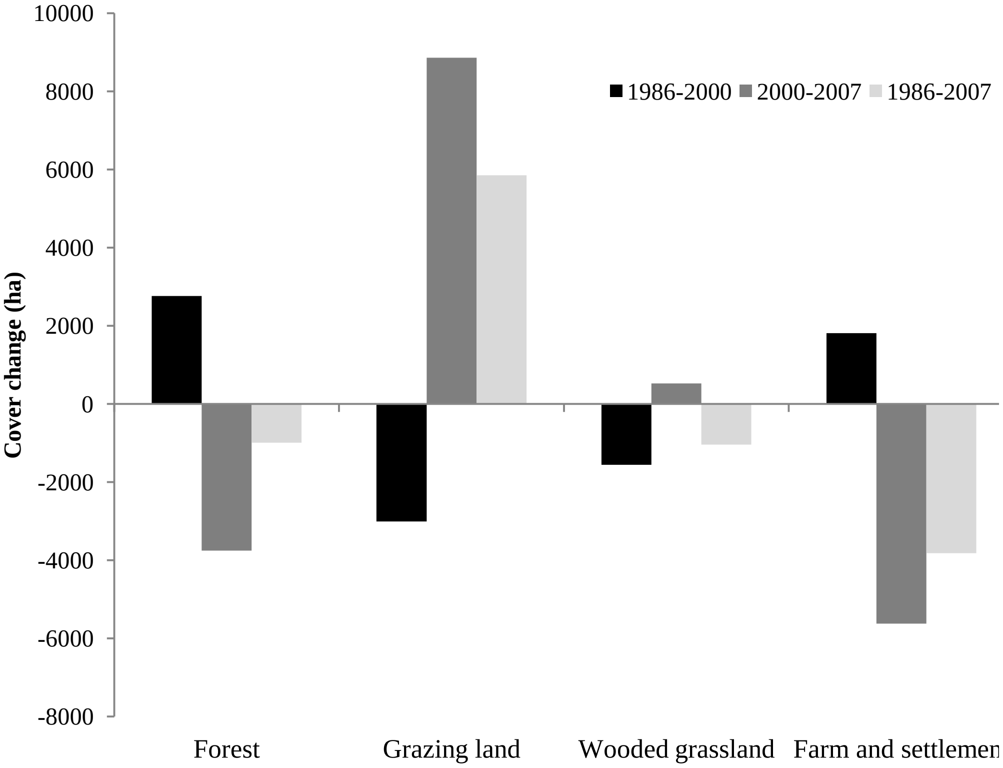


Figure 3.13 Net change in area (ha) of each land-use/land-cover class in each of three periods (1986-2000, 2000-2007, 1986- 2007) in Site III

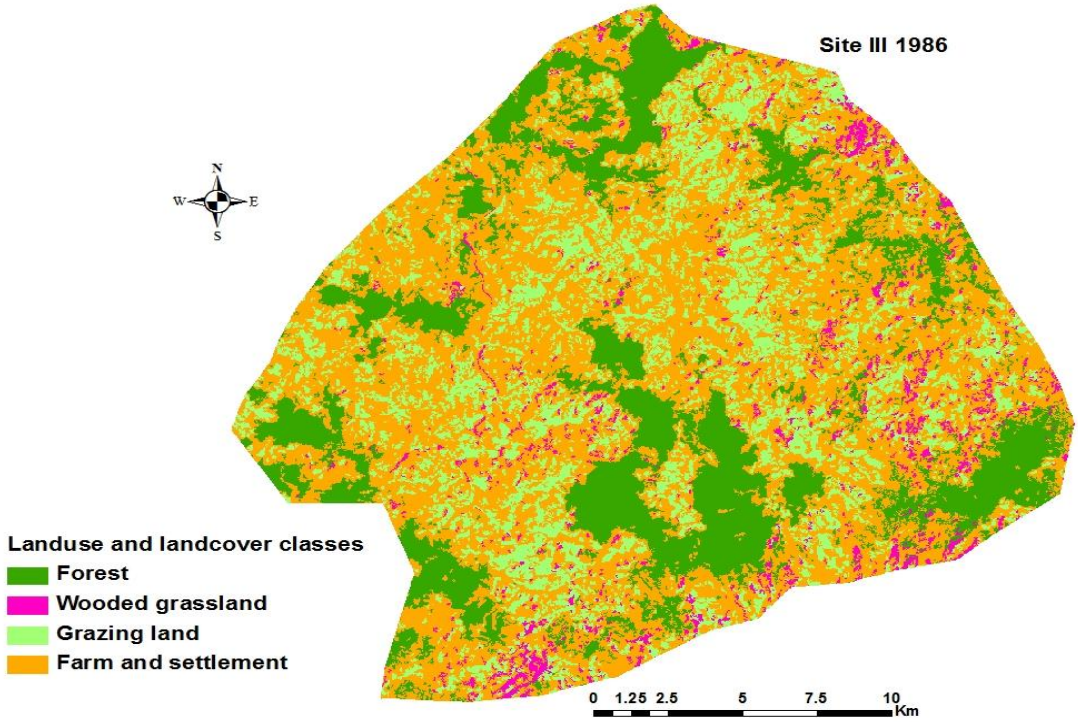


Figure 3.14 Final land-use and land-cover map of Site III in 1986

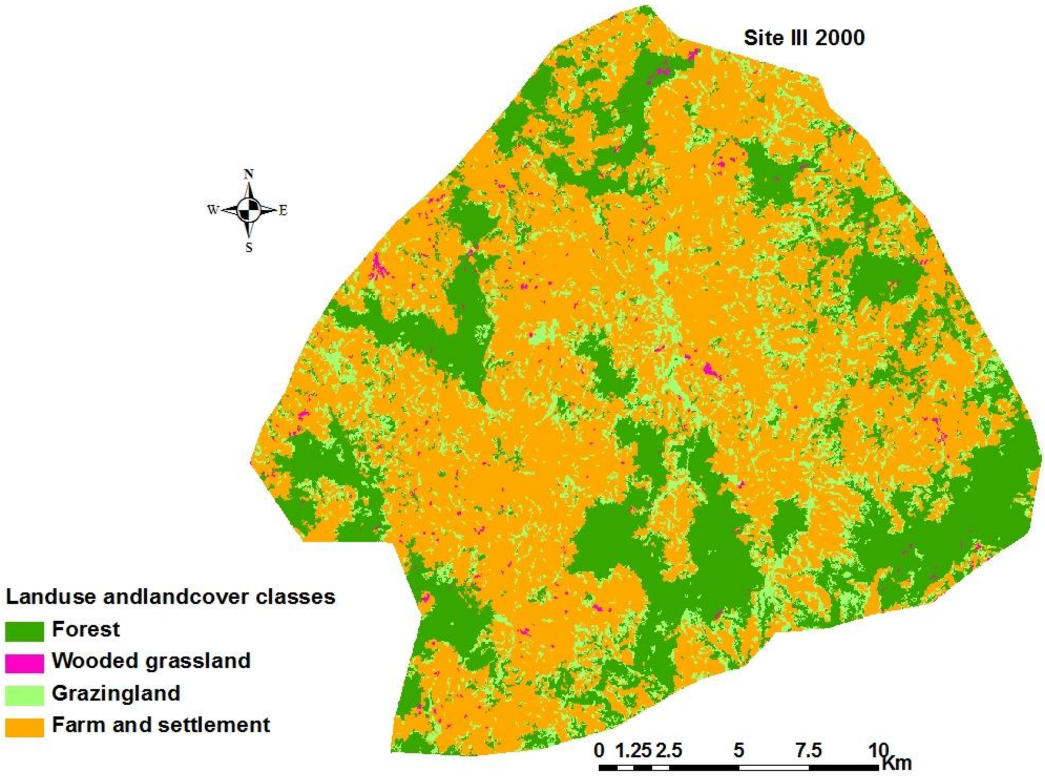


Figure 3.15 Final land-use and land-cover map of Site III in 2000

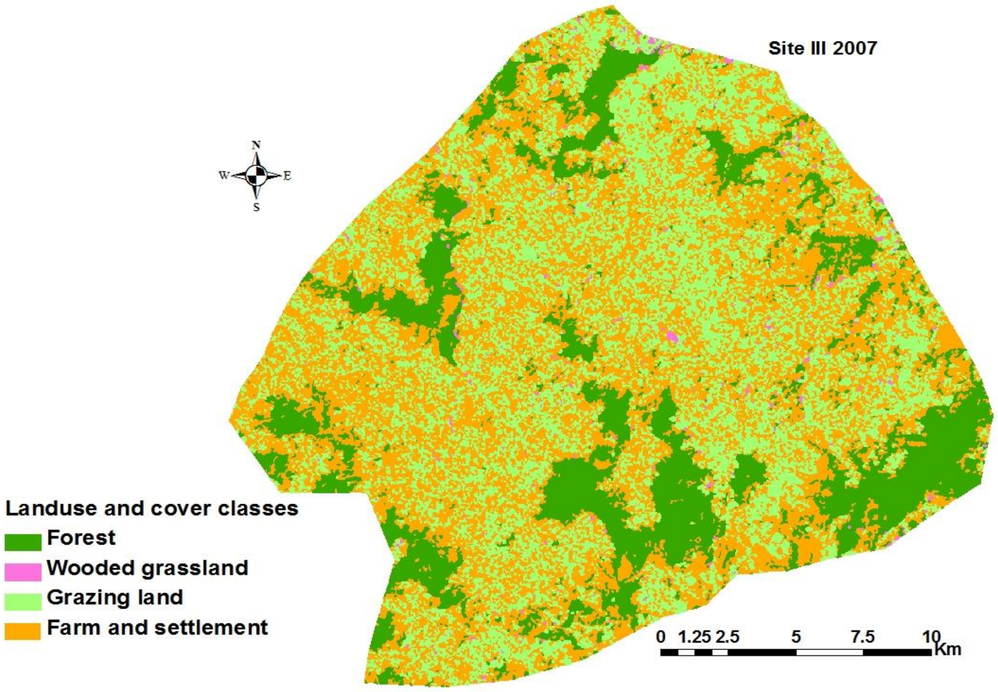


Figure 3.16 Final land-use and land-cover map of Site III in 2007

3.3.2 Land-cover spatial characteristics at each site between 1986 and 2007

3.3.2.1 Patch area and patch number

For the dominant LULC classes change in mean patch size (MPS, as the metric of patch area) was found to be very closely related to number of patches over each of the two time periods in all three sites (a decrease in number of patches being associated with an increase in MPS). Therefore the two metrics are dealt with together. The trends over time in these two metrics did vary greatly amongst the four land use/land cover classes within each site. However, within each LULC class there were similar trends in both metrics between the time periods across the three sites (especially for sites II and III). In all three sites the structure of the landscape became much more coarse-grained over time. In Site I, for natural forest, in both time periods patch number greatly decreased (to reach in 2007 just 20% of the 1986 number) and mean patch size greatly increased (to reach in 2007 462% of the 1986 size) (Figures 3.17 & 3.18, Appendix 3.1). Similarly, for farm and settlement, in both time periods patch number greatly decreased (to reach in 2007 just 31% of the 1986 number) and mean patch size greatly increased (to reach in 2007 317% of the 1986 size). Over the whole period the trend was the same (but the magnitude less) for grazing land, with a decreased in number of patches (to reach in 2007 64% of the 1986 number) and an increase in mean patch size (to reach in 2007 243% of the 1986 size). In contrast, the rare LULC class, wooded grassland, massively increased its number of patches (to reach in 2007 1470% of the 1986 number), but also had a modest increase in mean patch size (to reach in 2007 130% of the 1986 size).

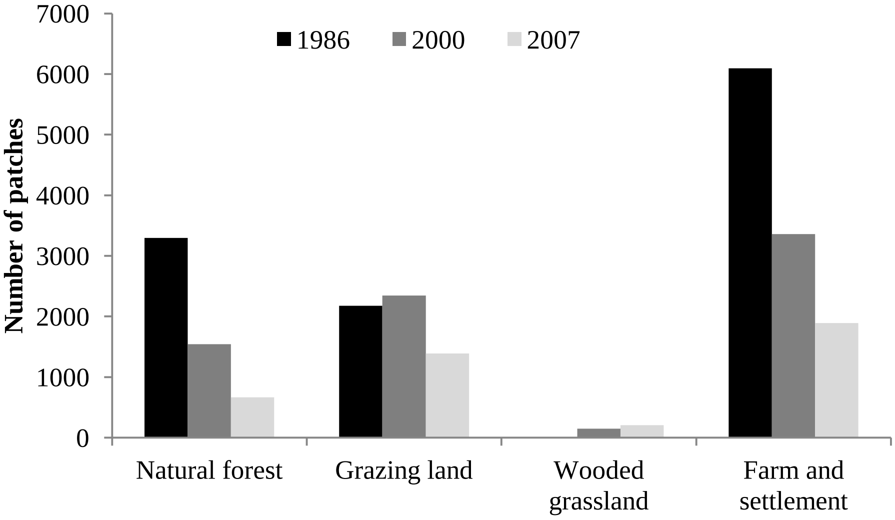


Figure 3.17 Number of patches of each of four land use/land cover classes in 1986, 2000 and 2007 in Site I of the study area

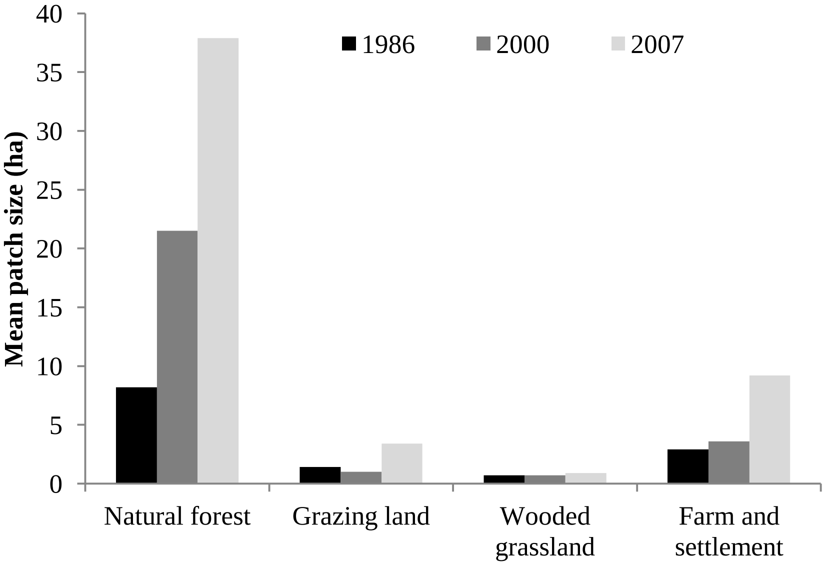


Figure 3.18 Mean patch size (ha) of the four land use/land cover classes in 1986, 2000 and 2007 in Site I of the study area

For Site II, the overall trend over the whole 1986-2007 period was comparable for the three dominant LULC classes to Site I, however there was greater variation between the two component periods. For natural forest, patch number increased then greatly decreased (to reach in 2007 44% of the 1986 number) and mean patch size decreased and then greatly increased (to reach in 2007 112% of the 1986 size) (Figures 3.19 & 3.20, Appendix 3.2). Similarly, for farm and settlement, in both time periods patch number greatly decreased (to reach in 2007 64% of the 1986 number) and mean patch size increased and then slightly decreased (to reach in 2007 136% of the 1986 size). Grazing land showed the same trend as forest, with patch number increasing then greatly decreasing (to reach in 2007 73% of the 1986 number) and mean patch size decreased then greatly increased (to reach in 2007 285% of the 1986 size). In Site II, wooded grassland showed the same trend as the other classes with a massive decrease in number of patches (to reach in 2007 just 12% of the 1986 number) and a doubling of mean patch size between 1986 and 2007.

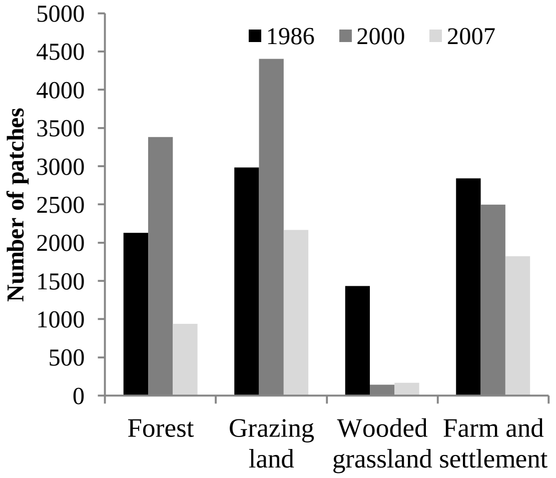


Figure 3.19 Number of patches of the four land use/land cover classes in 1986, 2000 and 2007 in Site II of the study area.

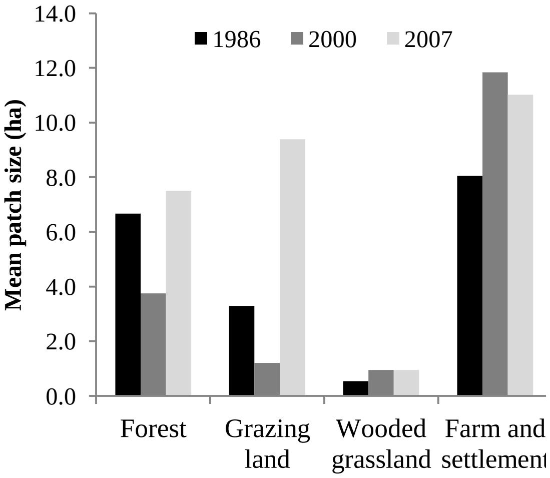


Figure 3.20 Mean patch size (ha) of the four land use/land cover classes in 1986, 2000 and 2007 in Site II of the study

The trends in Site III mirrored those in Site II very closely. For natural forest, patch number increased then greatly decreased (to reach in 2007 47% of the 1986 number) and mean patch size decreased and then greatly increased (to reach in 2007 194% of the

1986 size) (Figures 3.21 & 3.22, Appendix 3.3). Similarly, for farm and settlement, patch number decreased and then slightly increased (to reach in 2007 76% of the 1986 number) and mean patch size increased and then decreased (to reach in 2007 112% of the 1986 size). Grazing land showed the same trend as forest, with patch number increasing then greatly decreasing (to reach in 2007 91% of the 1986 number) and mean patch size decreased then greatly increased (to reach in 2007 183% of the 1986 size). In Site III, wooded grassland showed a very similar trend to Site II with a massive early decrease in number of patches (to reach in 2007 just 21% of the 1986 number) and an increase in mean patch size (to reach in 2007 138% of the 1986 size).

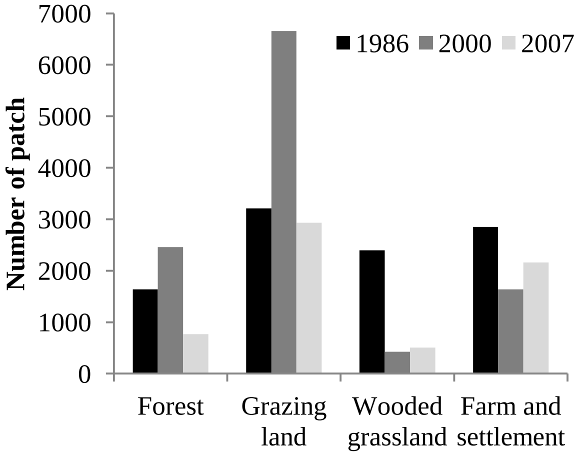


Figure 3.21 Number of patches of the four land use/land cover classes in 1986, 2000 and 2007 in Site III of the study area.

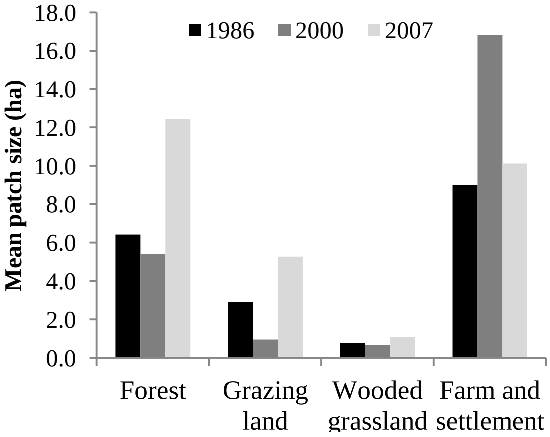


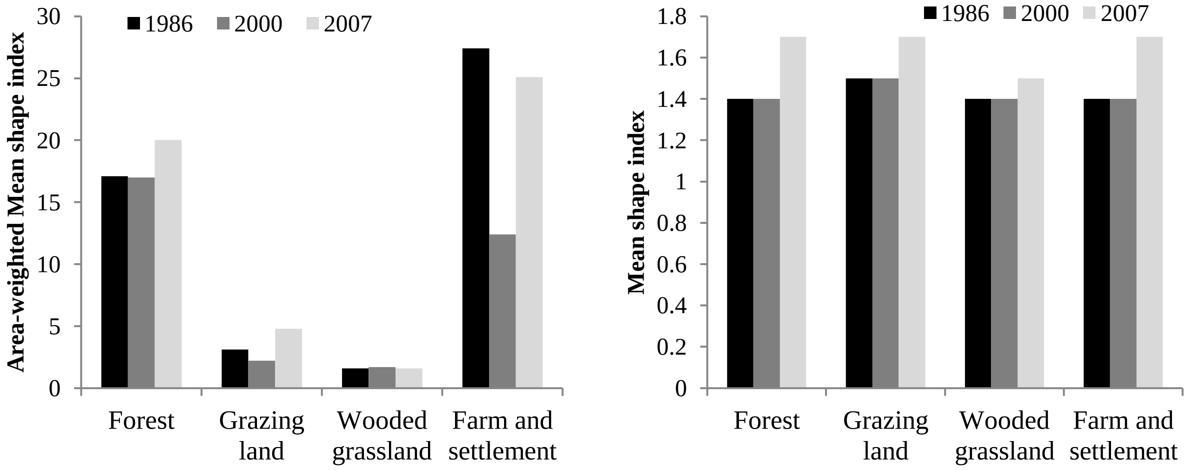
Figure 3.22 Mean patch size (ha) of the four land use/land cover classes in 1986, 2000 and 2007 in Site III of the study area

3.3.2.2 Patch shape and edge density

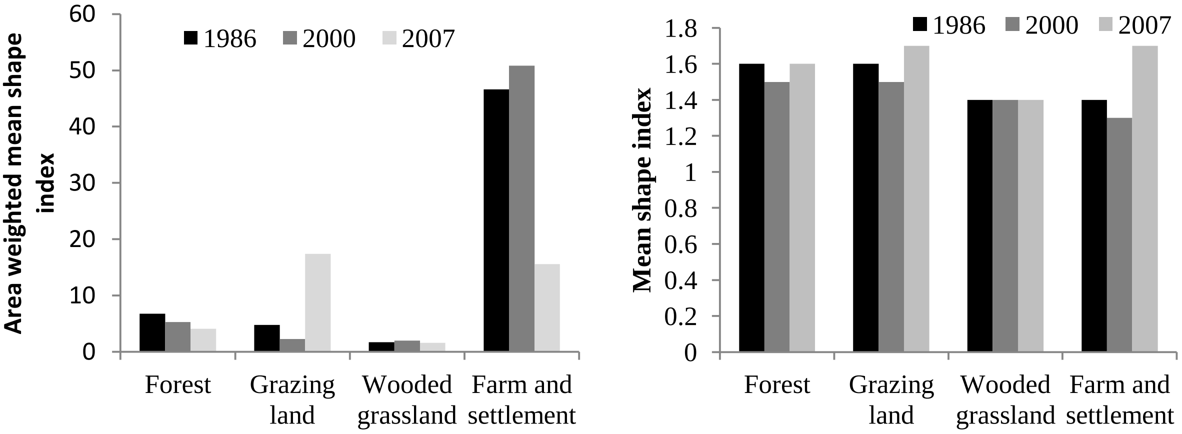
Of the standard patch shape and edge metrics produced by the Patch Analyst analysis of the four LULC classes in the three study sites just three metrics, i.e. area-weighted mean shape index (AWMSI), Mean shape index (MI) and edge density (ED, perimeter to area ratio) are selected for presentation in the main results because they are commonly used to characterize patch spatial pattern (they are reported in more detail in Appendices 3.1-3.3). Area-weighted mean shape index increases with greater irregularity of patch shape from a circle, while ED increases with greater complexity or elongation of patch shape. Of these two indices, AWMSI showed a slightly greater consistency for each LULC class across the three sites, than did ED. Between 1986 and 2007, for all three sites AWMSI increased for grazing land and decreased for farm and settlement (Figure 3.23, Appendices 3.1-3.3). However, for forest AWMSI decreased in sites II and III while it increased in Site I and, similarly, for wooded grassland it decreased in sites II and III while it remained the same in Site I. There was little evidence of consistency amongst the sites or LULC classes in the pattern of their trends between the two time periods (1986-2000 and 2000-2007). Forest decreased in AWMSI in two out of three sites in both periods. Grazing land decreased in AWMSI in all three sites 1986-2000 while increasing in all three sites 2000-2007. Wooded grassland increased in two out of three sites 1986-2000, while it decreased in all three sites 2000-2007. In sites II and III, farm and settlement AWMSI increased in 1986-2000 and decreased in 2000-2007, while in Site I it showed the opposite change in both periods. Across the three sites 1986-2007 AWMSI increased by an average of 158% in grazing land, while it decreased by an average of 11.7% for forest, 12.1% for wooded grassland and 27.4% for farm and settlement.

When mean shape index (MSI) of each class is considered it showed a much greater consistency for each LULC class across the three sites, than did ED. Between 1986 and 2007, MSI increased for all four LULC classes in both Site I and Site III; in Site II it increased in farm and settlement and in grassland but stayed the same in the other two classes (Figure 3.23, Appendices 3.1-3.3). In all three sites, for all LULC classes there was a tendency for MSI to decrease or stay unchanged in 1986-2000 but to increase greatly in 2000-2007. Across the three sites the average 1986-2007 increase in MSI was 21.4% for farm and settlement, 9.4% for forest, 8.6% for grazing land and 4.8% for wooded grassland.

Site I



Site II



Site III

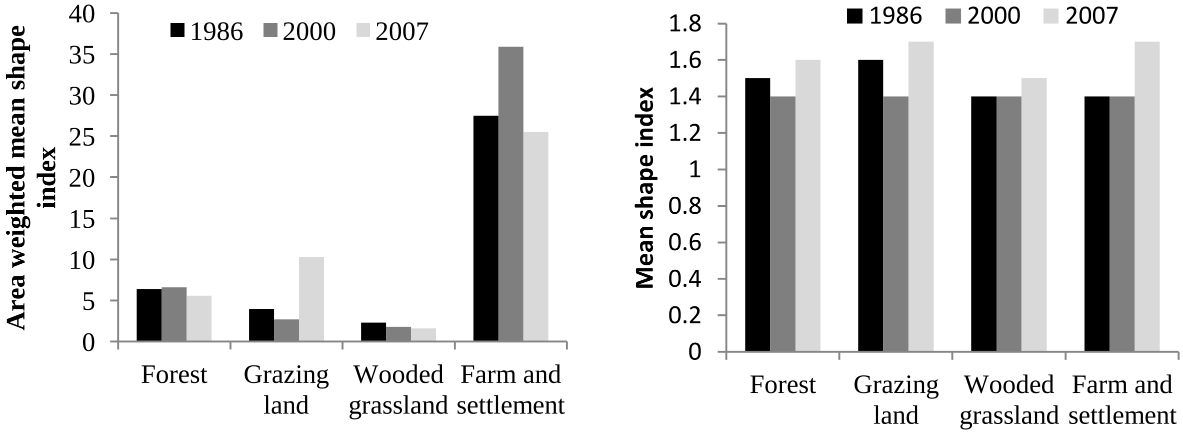


Figure 3.23 Area-weighted mean shape index (AWMSI) and mean shape index (MSI) for all four land-cover classes for sites I, II and III in 1986, 2000 and 2007.

For ED there was much less consistency amongst the three sites in its change between 1986 and 2007 (Figure 3.24, Appendices 3.1-3.3). Edge density did increase in wooded grassland in all three sites, for forest it increased in sites II and III but decreased in Site I, for grazing land it increased in sites I and III, but decreased in Site II, and for farm and settlement it increased in Site III, but decreased in sites I and II. The trend between the two time periods was also very variable amongst the three sites. In Site I patch ED tended to decrease 1986-2000 but increase 2000-2007 across the three dominant LULC classes. In Site II it increased during 1986-2000 (except for farm and settlement), but decreased during 2000-2007 (except for forest). In Site III it showed inverse temporal trends between two groups of LULC classes: in farm and settlement it decreased during 1986-2000, while increasing during 2000-2007; however in grazing land and in wooded grassland it increased during 1986-2000, while decreasing during 2000-2007. Across the three sites the average 1986-2007 increase in ED was 660% for wooded grassland, 48% for forest and 1% for grazingland, while it decreased by 12% for farm and settlement.

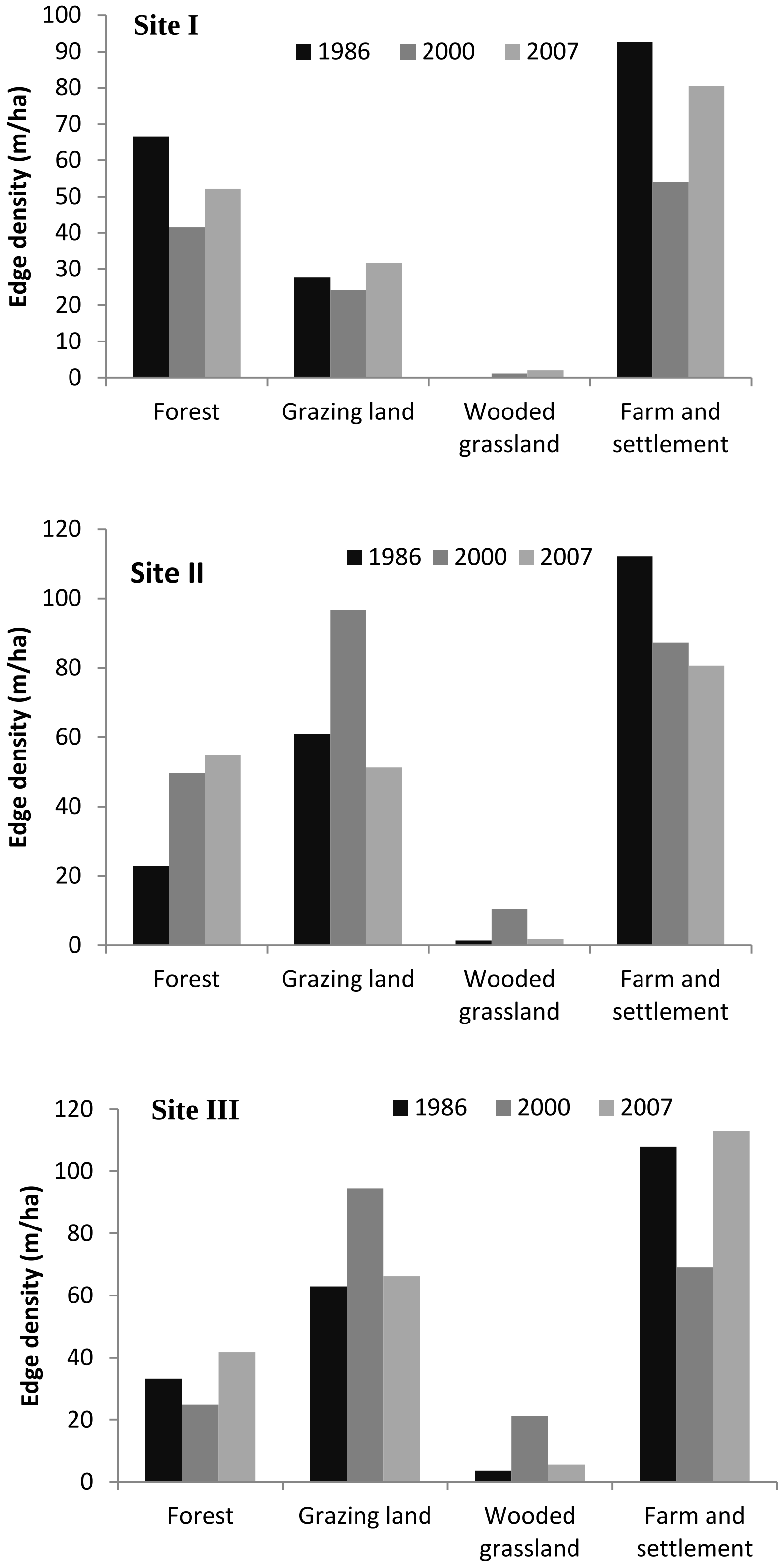


Figure 3.24 Patch edge density (ED) for all four land-cover classes for sites I, II and III in 1986, 2000 and 2007.

3.3.2.3 Landscape level analysis of fragmentation at each site

Combining all four land use/land cover classes to enable overall analysis at the landscape level, Site I was characterized by an increase in patch size during both study periods, from 4.11 ha in 1986 to 6.44 ha in 2000 and 11.48 ha in 2007 (Figure 3.25, details in Appendix 3.4). Mean landscape patch size in both sites II and III slightly decreased during 1986-2000 (from 5.08 ha to 4.57 ha in Site II and 4.69 ha to 4.24 ha in Site III respectively), and then drastically increased during 2000-2007 (to 9.35 ha for Site II and 7.44 ha for Site III respectively). The landscape of Site I was the most fragmented in 1986 as is indicated by its high number of patches which then decreased markedly during both time periods (Figure 3.25, Appendix 3.4). In the landscapes of both sites II and III patch number slightly increased during 1986-2000, but then (like Site I) sharply decreased during 2000-2007. The net Shannon diversity index of all patches of four cover classes showed a slight decrease in all sites to the end of the period (Appendix 3.4). Landscape area-weighted mean shape index showed contrasting trends amongst the three sites. In Site I it decreased during 1986-2000 and then increased by the same amount during 2000-2007. In Site II it increased and then greatly decreased, while in Site III it decreased markedly in both periods. In contrast landscape edge density (ED) showed the same pattern across the three sites: a decrease followed by an increase. The absolute variation of patch size (PSSD) increased in sites I and II while it decreased in Site III, however the patch size coefficient of variation (PSCoV) increased during 1986-2000 in all sites (Appendix 3.4). These two metrics both decreased in all three sites during 2000-2007 (Figure 3.25).

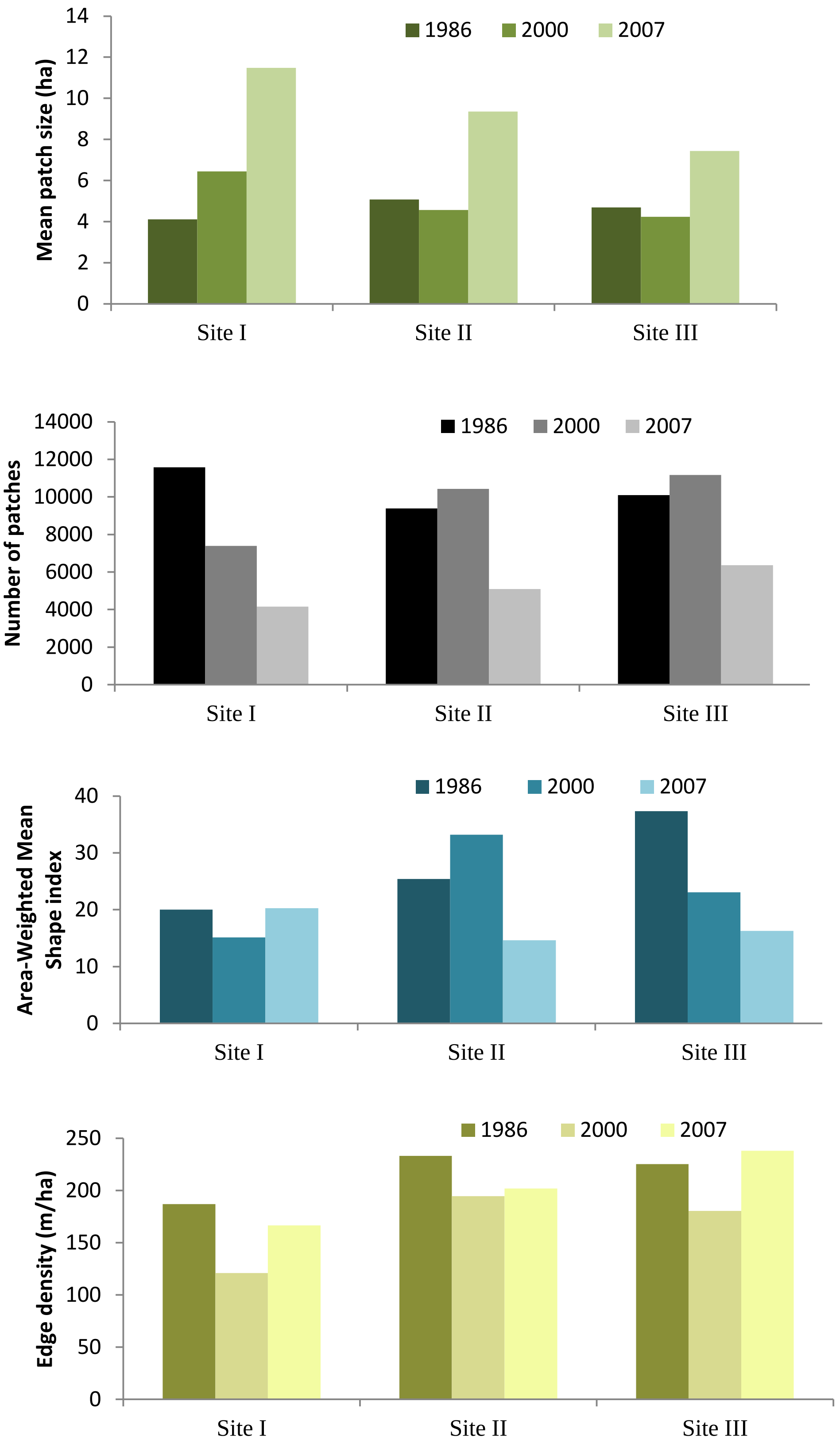


Figure 3.25 Number of patches (Nump), mean patch size (MPS), area-weighted mean shape index (AWMSI) and edge density (ED) for all classes combined at the landscape level for sites I, II and III in 1986, 2000 and 2007.

3.3.2.4 Forest core area

Of the total area of forest patches about 60.9% in Site I, 51.7% in Site II and 57.9% in Site III were core (interior habitat) area (Appendix 3.5). The largest total forest core area occur was in Site I in all three years, however it showed a marked variation during the period, increasing greatly during 1986-2000, before an even greater decrease during 2000-2007, giving a net loss over the whole period of 2526 ha (14.1% of the 1986 area) (Figure 3.26). Site III showed the same pattern, with a net loss over 1986-2007 of 478 ha (7.9%), however Site II had a decrease in both periods (especially 2000-2007), so it had lost a net decrease in forest core area of 3610 ha during the whole period (49.7% of 1986). The mean size of forest core area per patch showed the same trends as total core area for sites I and III: an increase during 1986-2000 followed by a decrease during 2000-2007 (Figure 3.26, Appendix 3.5). Both changes were of a very large magnitude for Site I, though they almost cancelled each other out (from 24.5 ha in 1986 to 81.0 ha in 2000 and 24.8 in 2007), and comparatively slight, though leading to a net decrease, in Site III (from 17.1 ha in 1986 to 18.8 in 2000 and 14.7 ha in 2007). Site II showed the opposite trend (and intermediate magnitude of change), from 10.4 ha in 1986 to 8.8 ha in 2000 and 9.5 ha in 2007.

The number of forest patches large enough to contain any core area (i.e. with any area at least 50 m from the nearest edge) showed a different trend from the other two metrics. In Site I the number decreased markedly from 733 in 1986 to 328 in 2000 before increasing by slightly fewer to 622 in 2000. In Site II the number decreased very slightly from 697 in 1986 to 667 in 2000 and then decreased much more to 385 in 2007. In Site III the changes in number were smaller than the other two sites and with a different trend: an increase from 351 in 1986 to 410 and then a decrease to 375 in 2007.

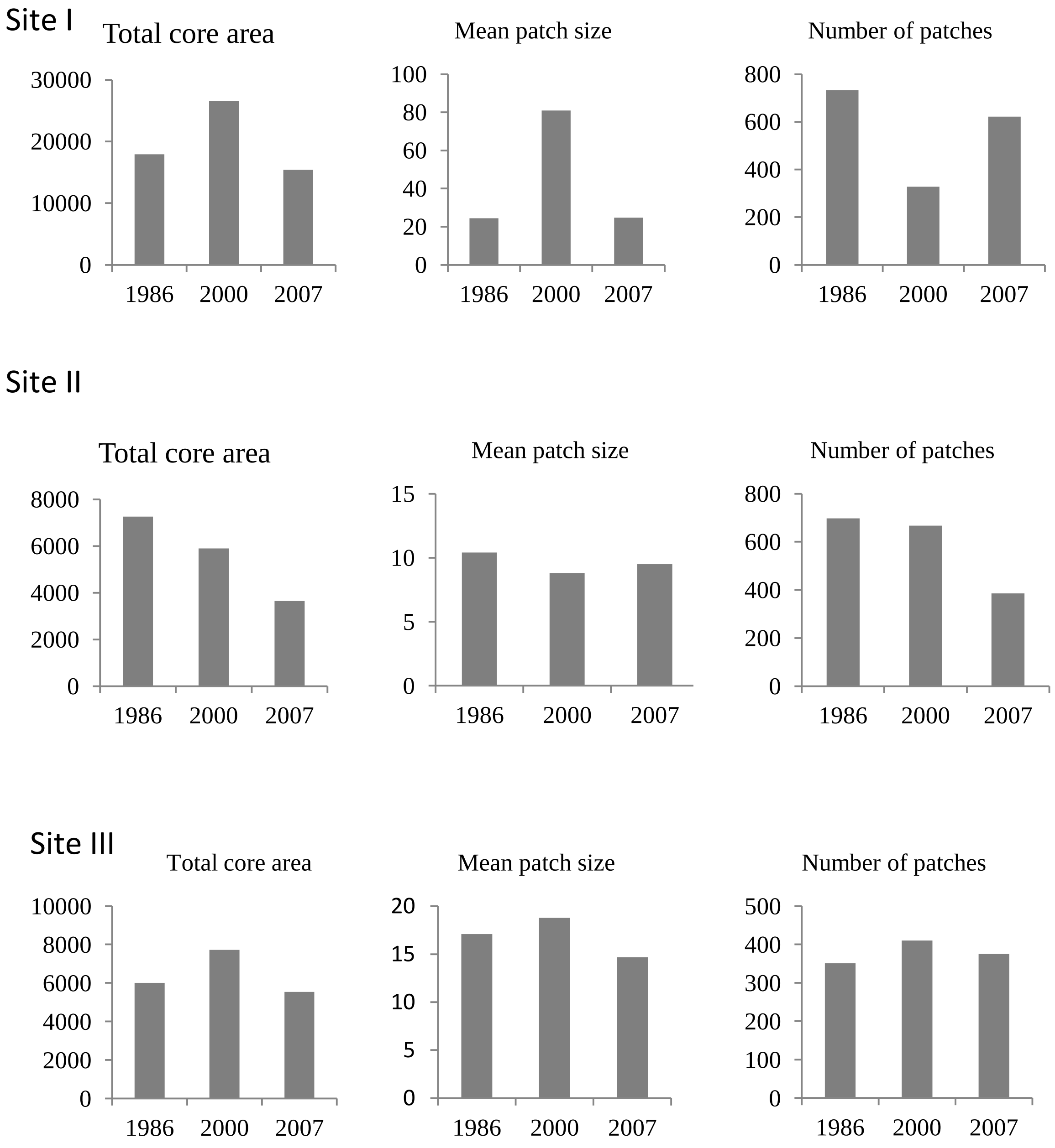


Figure 3.26 The total forest patch core area, mean size of core area per patch and number of patches containing core area for sites I, II and III in 1986, 2000 and 2007.

3.4. Discussion

In this study, analyses of the changes over time in four land-use/land-cover classes were first performed at a landscape scale and then more detailed landscape structural analyses were conducted covering the metrics of most relevance to habitat fragmentation: patch size, shape and edge density, with a more detailed assessed of the changes in core area of forest habitats.

3.4.1 Land-use/land-cover proportion

The high percentage of highland landscapes were converted into farm and settlement (Site II = 42.2% and Site III = 45.9%) and grazing land (Site II = 42.7% and Site III = 32.4%) at the end of the study period. While the dominant land-cover class in Site I was the forest (53.1%). The forest cover class was the third in proportion in high altitude (Site II = 14.8% and Site III = 20.0%). The increasing of human population and expansion of land under crop agriculture likely to be the most important factor influencing forest cover in high altitude. Mid-altitude areas have been used for coffee production which can retain most tree species as coffee shade (Aerts et al., 2011). The wooded grassland was the least proportion of cover classes in all study landscapes (Table 3.3, 3.5 and 3.7). It was increasing in Site I due to conversion of forests into farmland, then short grass fallow land under sparsely distributed trees in old agricultural field. In contrast to mid-altitude it was decreased in Site II and III at the end of the period due to conversion to agricultural land.

3.4.2 Influence of land-use/land-cover change on the forest spatial pattern

Forest area as a proportion of total land area gives a good overview of the contrast between forest cover and other land-uses. However, the spatial patterns and distribution of forests varied with the gradient of land-use pressure. In Site I, which was dominated by the relatively low impact land-uses of traditional coffee management system, these highly modified natural forests remained the dominant land cover (53%), still occupying 25300.4 ha, with a mean patch size (MPS) of 37.9 ha in 2007. Forest cover had only declined slightly from 56% in 1986. On the assumption that this area had once been under continuous forest cover, the large number of separate forest patches in 1986 (3297) indicates that there had previously been high levels of forest fragmentation. Some of this may have been very recent, triggered by the creation of a new resettlement area in 1984 near and inside the existing forests; this programme was designed to exploit this natural resource that at that time was seen as ‘under utilized’(Mengistu, 2005). Since 1986, the mean patch size of forest increased greatly (from 8.2 ha) to 37.9 ha in 2007), which could be taken as indicating a reversal of the previous fragmentation. However, much of this change is attribuTable to the reduction in number of forest patches by half between 1986 and 2000. This is attribuTable to the complete loss of many small patches as well as due to the merging of some nearby forest patches isolated in 1986 (Figure 3.4), e.g. due to secondary succession after abandonment of agriculture (either food crops or open-grown coffee). Widespread forest restoration by tree planting was not observed or reported by the local informants. The fall of world coffee market prices between 1986 and 2000 had a major impact in this area (Dr Tadesse Woldemariam, pers. comm.). It may have both led to abandonment of some coffee farming and, according to local elders, reduced the rate at which coffee farmers cleared natural forests. The importance of the combination of these processes was indicated by the increase in total class area (CA) of forest from 26,980 ha in 1986 to 28,006 ha in 2000. However, during the period between 2000 and 2007 there was a rise in world coffee prices. Consequently there was a resumption of net deforestation in Site I: forest area reduced from 28,006 ha in 2000 to 25,300 ha in 2007 (-6.2%).

The landscape of Site II in 1986 was characterized by a moderate cover of forest (14,186 ha, 29.8%), however this reduced to 12664 ha (26.6% of the area) by 2000. The period 1986-2000 was also characterized by a high rate of forest fragmentation, as reflected by an increase in the number of patches and decrease in mean patch size. The anthropogenic pressure on forest habitats in Site II continued after 2000 and led by 2007 to large reduction in total forest area (by 44%) and removal of 72% of the number of forest patches existing in 2000, as indicated by the LULCC map (Fig 3.11 and 3.28). Visual inspection of the 1986, 2000 and 2007 maps (figs. 3.9, 3.10 and 3.11) shows the increasing extent to which, with the exception of four larger blocks, remaining forest comprised patches increasingly isolated from each other within a matrix of human-dominated farm and settlement, and increasingly grazing land. The landscapes of Site III showed fluctuation in its forest cover, number of patches and mean patch size between 1986 and 2007. Over the whole period overall forest cover only declined slightly from 22.1% to 20.0%, but it did show a considerable loss of smaller patches with the large decrease in patch number (from 1639 in 1986 to 9530 in 2007) being paralleled by a doubling of mean patch size. It was noTable that the overall net decrease in forest cover over the whole period was much greater in Site II than in sites I or III. This is attribuTable to deforestation at the periphery of remnant forests and loss of large number of patches

3.4.3 Changes in patch size and spatial pattern from 1986 to 2007

Overall there was a net decrease in forest cover area in each study area between 1986 and 2007, although variation in the timing and spatial pattern of the decrease was a more striking result than the magnitude of net reduction in area. For example, forest cover for Site I and Site III increased between 1986 and 2000, and then decreased from 2000 to 2007, whereas forest cover in Site II decreased in both periods.

Land-use change obviously affects spatial pattern and ecological process. The five spatial processes in forest alteration identified by Forman (1995) include fragmentation (breaking up of habitat into patches of different size), attrition (disappearance of patches), perforation (the process of making holes in habitats), dissection (subdivision of habitat using equal width lines) and shrinkage (decreasing of size of habitat patches). The comparison of spatial processes between different periods within the same site and between different sites can be achieved in the present study by inspecting the vector forest maps for the years 1986, 2000 and 2007 as shown in Figure 3.4 to 3.6, 3.9 to 3.11 and 3.14 to 3.16 in combination with the calculated landscape-patch metrics. These indicate that the spatial pattern of forest cover in each study area has changed since 1986. Breaking apart of forest area, loss of forests area, decrease in patch size and increase of forest edge are all clearly shown providing evidence of the occurrence of all the main spatial pattern processes of fragmentation, attrition, shrinkage, perforation and dissection. This analysis is further assisted by Figure 3.27 which shows in greater detail changes in land use/land cover class distribution in one exemplar area selected from each site. This illustrates well the processes of forest expansion and then contraction (through attrition and perforation) in Site I, and more continuous attrition and perforation in sites II and III leading to the measured changes in patch number, mean size and shape.

A high number of separate patches of a habitat indicates high fragmentation (Cambin and Lachavanne, 2002). Therefore, the peak of forest fragmentation for the Site I landscape was in 1986 and for the Site II and III landscapes was in 2000. For Site I this interpretation was supported by its low mean patch size and high edge density in 1986. For all three sites the higher rate of overall deforestation occurred during 2000-2007. For Site I the pattern of forest cover change during both 1986-2000 and 2000-2007 was characterized by loss of patch number and increase in mean patch size. Whereas this was associated with a slight net increase in total forest area in the first period, there was a net decrease in the second. Site II experienced net deforestation in both periods, but this occurred in two distinct phases. It went through a period of increased patch number and reduced mean size, indicating the fragmentation of large existing forests (Griffiths and Lee, 2000), followed by an even greater reverse of a reduction of patch number by two thirds with an associated more modest increase in mean size, as the previously fragmented smaller patches were completely lost through conversion to other land use/cover. Forest cover change in Site III also went through two phases, but the pattern was more similar to Site I. Between 1986 and 2000 the net increase in forest area was associated with an increase in number of patches but a smaller decrease in mean patch size. Then the high rate of net deforestation during 2000-2007 was, like Site III, associated with a massive reduction in patch number (by two thirds), but also a large increase in mean patch size. In all three sites, the whole 1986-2007 period was characterized by fragmentation of the largest forest blocks with area reduction both through attrition at the edge and perforation, as can be seen in figs 3.4-3.16 and 3.27. In addition, in all three sites some many forest patches were totally converted to grazing land or farm and settlement land.

The increase in length of edge between forest and other land-cover classes indicates a high rate of human land-use activity (Turner et al., 2003; Bennett, 2003). This interpretation was supported by the results of the present study in the way that changes of edge density coincided with other indicators of land-use induced deforestation and forest fragmentation. In Site I, it was highest in 1986 (66.5 m ha^{-1}) and decreased (to

41.5 m ha^{-1}) before increasing to 52.5 m ha^{-1} in 2007. Site III showed the same pattern, a decrease from 33.1 m ha^{-1} to 24.8 m ha^{-1} followed by an increase to 41.7 m ha^{-1}. In contrast, in Site II edge density increased in both periods from 22.9 m ha in 1986 to 54.5 m ha^{-1} in 2007.The trends of shape complexity weighted by the area of patches for forest (AWMSI) gave a different perspective on patterns of deforestation/fragmentation from the other indices. In Site I the expectation that AWMSI would positively correlate with edge density in showing an opposite trend to forest cover was met. However, in both sites II and III the trends in AWMSI were positively correlated with forest cover and negatively with edge density. This difference between the higher altitude and lower altitude sites can be attributed to large impact of forest in higher altitude

3.4.4 Spatial characteristics of forest patch core area (CoA)

The core forest areas are the interior of forest patches, which are not influenced by the edge effect caused by interaction with the surrounding matrix (Turner et al, 2001). It is the central portion of a patch that remains after removing the specified edge buffer (Leitao et al., 2006). The edge buffer is the transition zone between non-forest and interior forest around the patch perimeter. To delineate the edge buffer for species conservation studies, the distance is usually defined by the user to be relevant to a specific organism under investigation or objective of the research. Since the present study was addressing the quality of forest habitat as a whole, a more general edge width had to be selected, 50 m which is widely considered as an adequate distance to detect edge influence on plant species. A high proportion of forest plant species are considered to be sensitive to microclimate, therefore loss of core forest loss is expected to have a major impact on conservation of these species decreasing their within-patch population size, and the suitability of habitat patches to support their meta-populations, thus increasing extinction risk (Farina, 2007). Change in core forest area is expected to be broadly correlated with mean patch size, however it will also be influenced by changes in patch shape and edge density at a landscape level.

The spatial pattern of change in core area is illustrated in Figure 3.28. Those core areas present in 1986 that had been lost due to conversion to other land-cover types are shown in red. The scale of the Figure is such that loss of core area by it entering the buffer zone due to incremental shrinkage of the area of forest patches cannot be seen. The loss of core area during 1986-2007 was greatest in Site II, followed by Site III and least in Site I. The 1986 landscape in Site I had a cumulative core area of 17,926 ha out of the total 26,980 ha forest area (66.4%). Paralleling the trends in total forest area, the core area increased by 28.4% between 1986 and 2000, and then reduced by 5.6% between 2000 and 2007. These trends were amplified in the mean size of patches containing any core area which increased from 24.5 ha to 81.0 ha before reducing to 24.8 ha. In Site I 66.4% of the total forest patches were large enough to contain core forest area in 1986, but by 2007 this had reduced to 60.8% reflecting the high rate of loss of small patches. Taking these results together, in Site I there were large fluctuations in the habitat area available for forest-dependent species, which occurred so rapidly that there was probably a lag in the capacity of their populations to track these changes.

The fluctuation in core forest area was less in Site III. From 6002.5 ha in 1986 (57% of the total forest area) this increased to 7720.1 ha (a similar 58.1%) by 2000, before falling to 5524.4 ha (57.9%) by 2007. The mean size and the number of patches containing core forest showed very comparable changes. Matching the overall trends in deforestation, the changes in core forest habitat were greater in Site II. The total core area fell from 7256.8 ha in 1986 (51.1% of the total forest area) to 5890.7 ha (46.5%) in 2000 and at a greater rate to 3647.1 ha (51.7%) in 2007. The number of patches containing core forest showed the same trend with a reduction in percentage of total patches containing core forest area from 39.9% in 1986 to just 22.0% in 2007. However, their mean size increased slightly during 2000-2007 after having fallen during 1986-2000. The latter result probably reflects a high rate of loss of area of patches of the scale of 1-5 ha (pushing them below the threshold size to contain core forest habitat). Figure 3.28 shows the pattern of the high rate of loss of core forest area present in 1986 and 2000 through conversion to other land-uses at the edge of larger forest patches and complete loss of core forest area in small or even medium-sized patches. Overall the loss of core forest area from Site II was acute during the study period, posing a severe threat to the conservation of forest-dependent biodiversity.

3.4.4 Implication for habitat conservation

In the present study the rate of LULCC has been high over the 21 year study period in all three sites. The pattern varied amongst the three sites. The lower altitude site where land-use had been dominated by coffee farming under forest, after an initial expansion of the dominant forest cover suffered high rates of fragmentation by other land uses. One of the two higher altitude sites suffered high rates of deforestation and fragmentation during the study period which markedly accelerated during the final seven years. This high rate of deforestation (> 1% per year) and fragmentation during the 2000-2007 periods was a feature of all three sites and should be a matter of high concern for biodiversity conservation. Habitat fragmentation always goes together with habitat loss and decrease in metapopulations (Lindenmayer and Fischer, 2006). In all three sites, the landscape has lost many forest patches and there has been a noTable increase in mean patch size due to a high rate of loss of small patches, indicating a shift in the underlying landscape matrix with lower connectivity of more fragmented forest due to expansion of the area of other land uses, especially grazing land. Despite the increase in mean patch size over the whole 1986-2007 period in all three sites, such was the high overall rate of deforestation that they did have a reduction in the area of core (interior) forest habitat. This rate of loss was particularly marked for the 2000-2007 period indicating the high current level of threat to forest-habitat dependent species across this whole area.

At the same time, these fragmented forest patches are largely natural ecosystems that provide local communities with timber, food, fuel, wild fruit, spices and medicinal plants. Their role as the location of forest coffee production is particularly important economically. More fundamentally, these forests have a potentially important role in delivering the ecosystem services of regulating climate, purification of air and water, and the supporting service of maintaining soil fertility by controlling of erosion, and pollination of crops (MEA, 2005).

The loss of these forests can be reversed in two main ways: conservation of existing isolated patches of habitat and restoration of degraded habitat. On the assumption that core forest area that has most recently been lost should be the highest priority for the restoration of forest habitat and its biodiversity because these areas will have the highest potential for restoration due to retention of some root stock and presence of some similar tree species and ultimately contribute to increasing core forest area. Figure 3.28 provides a valuable guide to the priority areas in each of the three study sites (those lost since 2000 coloured light green, followed by those lost since 1987 coloured red).

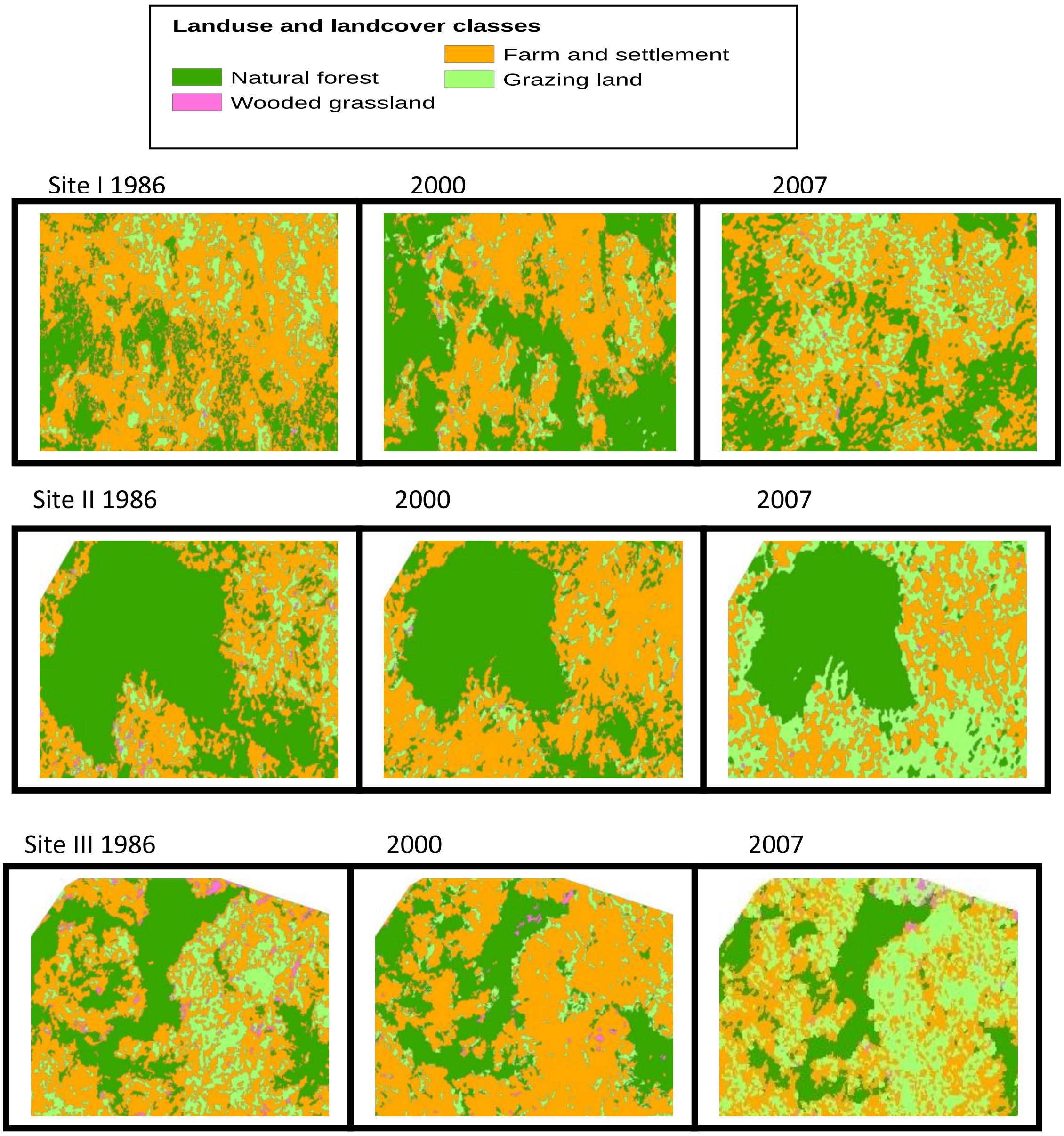
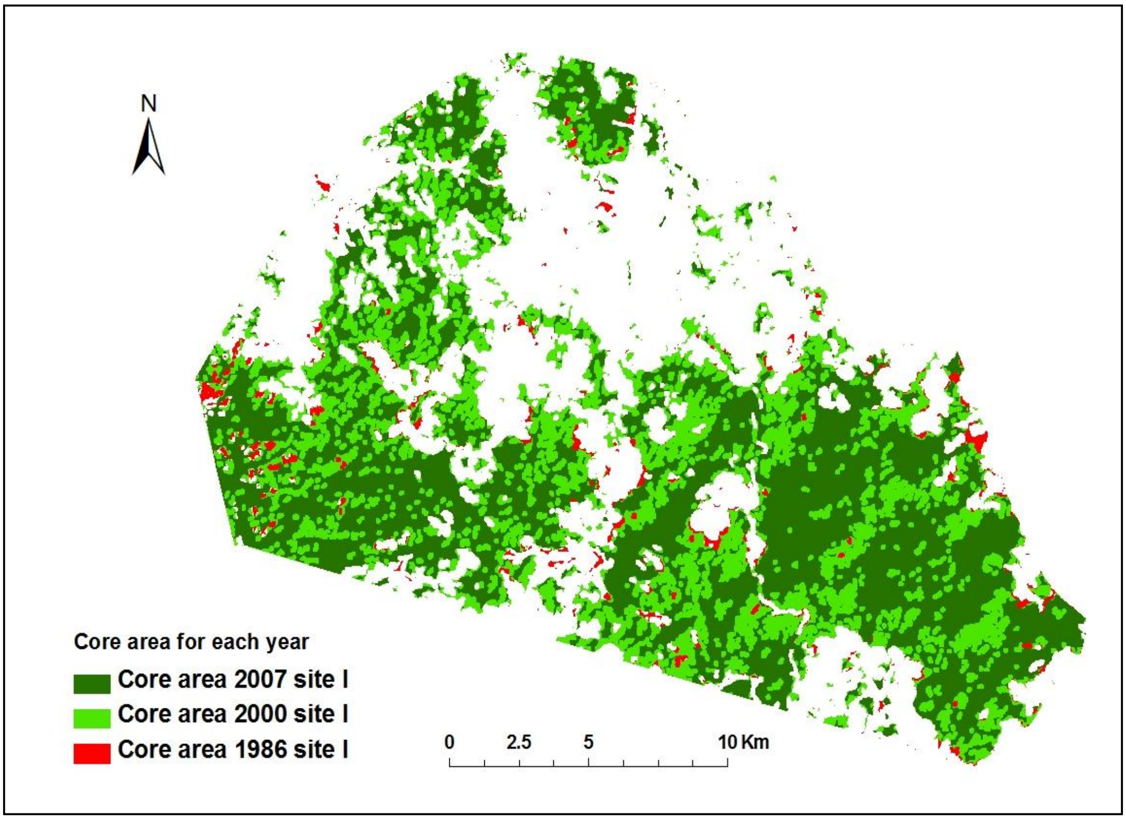
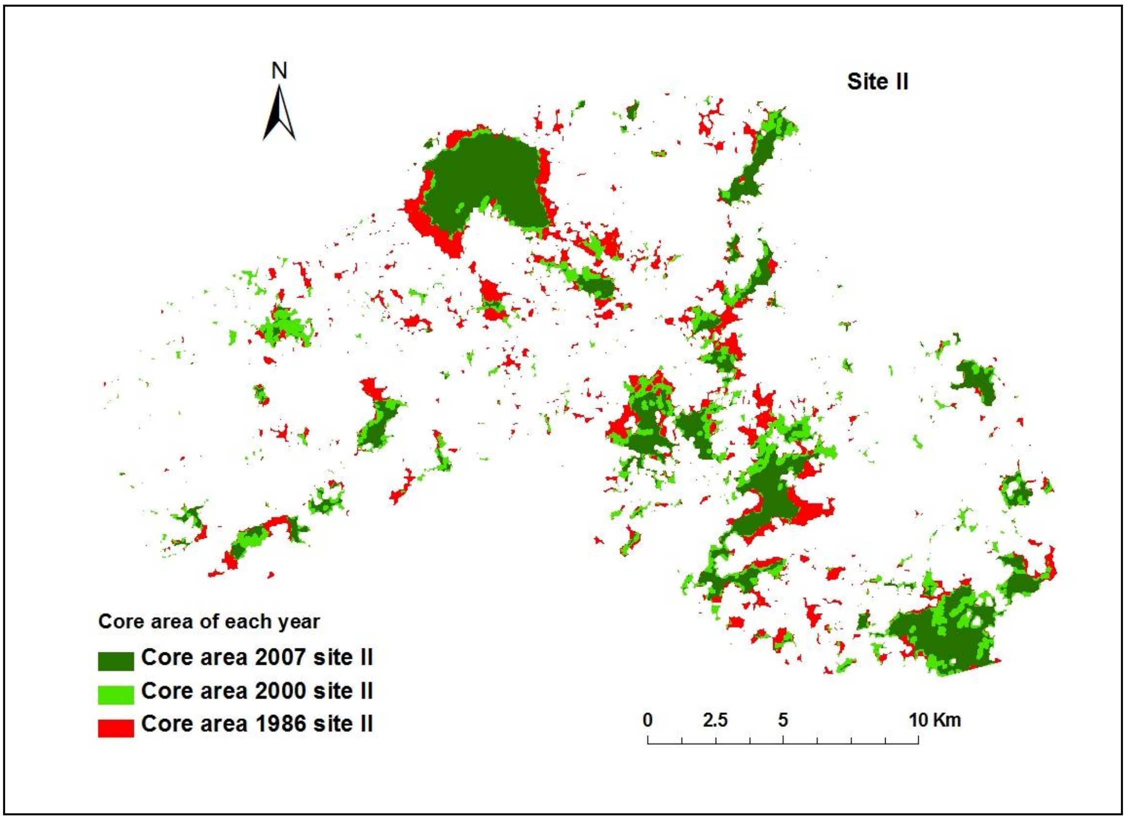


Figure 3.27 Selected sections of land-cover maps indicating the spatial process in natural forest loss in sites I, II and III from 1986 to 2007.


N
Core area for each year
Core area 2007 site I
Core area 2000 site I
Core area 1986 site I
0
2.5
5
10 Km


N
Site II
Core area of each year
Core area 2007 site II
Core area 2000 site II
Core area 1986 site II
0
2.5
5
10 Km

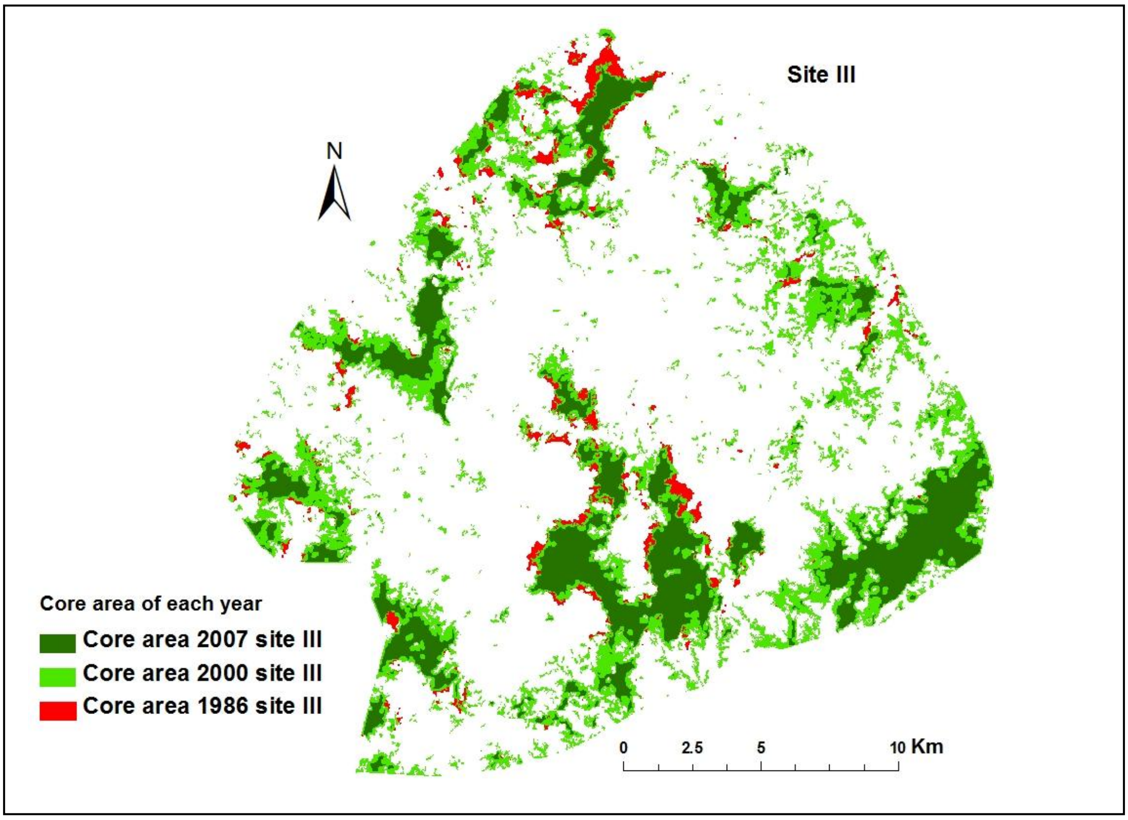


Figure 3.28 Diagrammatic illustration of changing core area at all three study sites in each year. Red shows core area present in 1986 that was lost in 2000 and in 2007. Light green shows core area present in 2000 that was lost in 2007. Dark green shows core area present in 2007.

CHAPTER 4

THE EFFECT OF HUMAN-INDUCED FRAGMENTATION ON FOREST FLORISTIC COMPOSITION, SPECIES RICHNESS AND STRUCTURE IN ILLUBABOR ZONE, ETHIOPIA

Abstract

Land-use expansion for crops, pasture and rural settlement in south-west Ethiopia has caused forest habitat fragmentation which is likely to lead to biodiversity decline due to habitat reduction and loss. Human disturbance may also directly alter woody species composition within remaining forest patches. I evaluated the importance of forest patches for the conservation of afromontane rainforest regional biodiversity in Illubabor Zone, Oromia Regional State, south-west Ethiopia, and the impact of fragmentation and disturbance on this biodiversity. I analysed woody species richness, composition and structure in 240 plots within 30 forest patches in three landscapes (sites I, Site II and III) with different land use intensities. Site I contained large patches of unfragmented forest (> 2000 ha) whereas sites II and III contained smaller forest patches (1 - 2000 ha).

All woody species (254 species) sampled were native to the area. Despite their recent reduction in area, the forest patches contained a diverse community of native plants, including: IUCN red-listed plant, nationally endemic species, afromontane endemics, national priority species for conservation and economically important species.

Species richness was significantly different amongst the three sites, with a notably steeper species/individual accumulation curve for Site I. However, tree species density and basal area in Site I was not different from sites II and III. Species composition and distribution patterns are influenced by disturbance and topography more than by patch area and shape. In conclusion, this study showed that a high portion of the remaining forest plant diversity in this landscape is located in fragmented patches which are important for regional biodiversity conservation.

4.1 Introduction

Land-use expansion due to human population growth has greatly transformed the land-cover of natural habitats, which has resulted in measurable changes to their composition and spatial pattern (Turner et al., 1998; Farina, 2007). Demand for more agricultural land, pasture and urban expansion has caused habitat fragmentation, which ultimately leads to a decline in biodiversity (CBD, 2001; MEA, 2005). Habitat fragmentation is a dynamic process in which the fragmented habitats are gradually reduced to smaller patches, decreasing connectivity between habitats (Turner et al., 2001, Echeverria et al., 2007). Reduction of forest patch size, and the accompanying spatial change caused by this fragmentation, decreases species richness (Laurance, 1998; Gergel and Turner, 2002), the density of species populations within these patches, and their meta-populations across the landscape (Farina, 2007). Following fragmentation, the patches could also be exposed to different types of human-induced disturbance, such as grazing, timber harvesting, forest clearance and firewood collection (Hamer et al., 1997). Such disturbance within a habitat could also have a profound effect on species richness, abundance and floristic composition, compounding the influence of reduction in patch size, and change in shape and edge density. However, little is known of the impact of these factors in south-west Ethiopia.

The south-west of Ethiopia was largely (40%) covered by afromontane rainforests in the 1970s (Reusing, 1998) of which less than 18% cover remained in 2000 (Reusing, 2000; Tadesse et al, 2002). This extensive forest depletion was caused by land-use change as a result of population growth from planned government resettlement programmes, and the fast conversion of forests to agricultural uses and human settlement (Scmmitt, 2006). Most large forests now exist along rivers and inaccessible rocky areas, while large numbers of patches of different size and shape exist as remnants in landscapes of different land-use intensities and patterns. Local communities have close relationships with these forest patches for their food and livelihood security. They contain important plant species such as coffee (*Coffea arabica*), cardamom (*Aframomum corrorima*) and wild pepper (*Piper capense*) (Teketay, 1999; Tadesse, 2003, Fayera and Denich, 2006). Furthermore, they provide people with wild honey, mushrooms and medicinal plants on which they depend to supplement their livelihoods (Desalegn and Yilma, 2008). However, these habitat patches have long been subjected to deforestation and fragmentation from land-use change, and degradation from grazing and over-exploitation of forest products (Reusing, 2000), and their maintenance is likely to be important for the ongoing provision of ecosystem goods and services.

Land-use practices around these fragmented forests, and over-harvesting of products within the forests, may alter the composition and distribution patterns of species. A detailed knowledge of these effects is important, in order to assess their conservation status and plan their management. There have been some descriptive studies undertaken of these forests, to gain information about species composition and structure (for example, Kumelachew and Tamrat; 2002; Tadesse, 2003, Fayera and Denich, 2006, Schmitt, 2006, Tadesse et al., 2008, Schmitt et al., 2010, Aerts et al., 2011). However, most of these studies focused on single large and unfragmented forest reserves. Currently, most of the fragmented forests situated in matrices of different land-uses in south-west Ethiopia have not been studied, and thus their conservation importance is little known. New research is needed, which uses methods of landscape ecology, to determine the relative importance of the loss of forest area, forest fragmentation and degradation of forest habitats within the landscape to species composition and conservation value.

The objectives of this study were to determine the impact of fragmentation of afromontane rainforests on plant species composition and to evaluate the importance of small forest patches embedded in agricultural land for the conservation of regional plant diversity. To accomplish these objectives I analysed species richness, species composition, diversity and forest structure in forest patches located in three landscapes with different levels of land-use intensity in Illuababor Zone, south-west Ethiopia. Specifically, I examined the contribution of small patches to regional plant species diversity; the relationship of species richness, species composition and forest structure to patch size, shape and edge density; differences in species richness and diversity among landscapes; and patterns of species composition in forest patches within and among landscapes.

4.2 Materials and Methods

4.2.1 Selection of sampled forests and their spatial characteristics

An exploratory survey identified the location of potentially suiTable forest patches in the three sites. From these, 30 forest patches were selected based on the following criteria: locally representative of natural forest in their full range of characteristics (naturalness, ≥ 50% canopy cover); sufficient accessibility for field work. Amongst the three landscapes defined in Chapter 3, this resulted in a sample of 8 patches in Site I, 12 patches in Site II and 10 patches in Site III.

The spatial characteristics of each sampled patch of size < 2000 hectare were derived from the 2007 land use map produced for land use study in chapter 3 (land-use and land-cover map 2007). This method was adopted from Altamirano et al. (2010). For each patch, the following spatial data were calculated: patch area, shape, fractal dimension (Mandelbrodt, 1982) and edge density using Patch Analyst (Arc GIS extension) (Table 4.1). The area of each forest patch was quantified from the polygon produced. The shape index (SI) of each patch was calculated using, $Si = \frac{Pi}{2\sqrt{\prod * A}}$, (Arroyo-Rodriguez and Mandujano, 2006) where P is the perimeter of the patch (m) and A is patch area (m^2). Its value for a circular shaped patch is 1 and values greater than 1 represent irregularity potentially indicating that the patch is more dominated by a range of edge-influenced microenvironments (Honnay et al., 1998). The edge density of each patch was obtained by dividing patch total edge length to area, (edge: area ratio). The slope and aspect of the patches were derived from a Digital Elevation Model (DEM) of 90 m resolution obtained from United State Geological Survey (USGS). The elevation of each patch was measured during field survey using a GPS (Garmin 60). All of the eight sampled forests in Site I were located in different parts of the same large continuous forests > 2000 ha in area, and none of these landscape ecology metrics were calculated for these samples (Table 4.1). The raw rainfall data over 23 years for the 24 meteorological stations within the study area were obtained from the Ethiopian Metrological Agency and, using the methods described in detail in Chapter 2, used to estimate mean annual rainfall for each sample/patch.

Table 4.1 The spatial and rainfall characteristics of forest patches in the Illubabor Administrative Zone of Ethiopia***:*** shape index, SI; fractal density, FD; edge density, ED; patch area, PA; altitude (above sea level); slope; aspect; mean annual rainfall (1986-2009).

Local name of patches	Site	SI	FD	ED	PA (ha)	Altitude (m)	Slope (%)	Aspect	Mean annual rainfall (mm)
Issya	III	4.9	1.4	162.6	109	2197	4.1	N	1928
Qotora1	III	8.4	1.4	62.9	2000	2179	12.9	N	1928
Qotora2	III	2.7	1.3	76.3	164	2312	13.3	N	1928
Koda	III	7.9	1.4	63.3	1958	2296	16.4	E	1780
Simber	III	4.9	1.4	111.8	249	2268	9.8	E	1780
Dike	III	5.6	1.4	78.0	656	2422	9.8	N	1811
Jireen	III	2.7	1.3	132.0	52	2356	9.4	E	1811
Busajireen	III	3.6	1.4	105.0	151	2156	4.9	NE	1811
Buyina	III	2.8	1.3	222.8	3	2237	7.3	W	1548
Jamia	III	2.7	1.3	58	2000	2197	3.6	S	1903
Tuluboka	II	4.7	1.3	96.2	304	2188	3.9	NE	1662
Sibu1	II	2.3	1.3	137.1	37	1942	2.9	N	1576
Sibu2	II	2.5	1.3	125.7	52	2198	3.3	N	1576
Markafa	II	3.2	3.2	152.4	55	1918	2.2	SW	1576
Yaroo	II	2.4	1.3	128.9	43	2141	3.3	N	1828
Lagachancho	II	2.1	1.5	110.0	85	1951	5.5	N	1576
Gobe	II	7.8	1.4	111.6	620	2005	5.4	E	1576
Malate	II	3.7	1.4	99.8	174	2044	6.4	W	1828
Tulumaka	II	2.9	1.3	81.2	166	2319	5.5	N	1828
Jorgo	II	3.1	1.3	30.7	1301	2192	5.7	N	1662
Qolowarabesa	II	1.2	1.3	526.0	0.6	1903	3.2	W	1576
Bongi	II	2.8	1.3	212.0	10	1878	5.7	W	1576
Gorezuria	I	-	-	-	>2000	1695	5.5	SW	1797
Metuzuria	I	-	-	-	>2000	1594	3.3	N	1548
Burusa	I	-	-	-	>2000	1629	3.3	N	1548
Baljo	I	-	-	-	>2000	1625	3.2	NE	1121
Uka	I	-	-	-	>2000	1635	3.7	SW	1121
Masha	I	-	-	-	>2000	1948	2.2	SW	2048
Leka	I	-	-	-	>2000	1853	3.5	SE	1898.
Gaba	I	-	-	-	>2000	1518	3.8	SW	1356

4.2.2 Vegetation data collection

A quantitative and qualitative field survey of the 30 forest patches was undertaken between January and November 2010. In order to obtain data on the plant species richness, species diversity, floristic composition and structure of each, a plot-based inventory was carried out. A single main plot of 60 x 100 m was established in each patch. This main plot was sub-divided into 15 sub-plots using a 20 x 20 m grid. Eight of these sub-plots, based on a systematic sampling regime, were used for vegetation inventory (see plot layout in Figure 2.10). All woody plants that were live, standing dead or fallen with ≥ 5 cm diameter at breast height (DBH) were recorded (Table 4.2). Their DBH was measured using a diameter tape. Trees with large buttresses were measured above these protrusions. All stems ≥ 5 cm in DBH of plants with multiple stems were measured separately. To get an equivalent total diameter for the whole plant of these multiple stems, the basal area of each stem was calculated; these were summed and then divided by π, square rooted and multiplied by two. The height of the every recorded plant was estimated using a clinometer. When difficult to identify the top part of an individual tree in a dense forest I took an estimated value for the height. In addition, all plants of woody species that were < 5 cm DBH but > 1m height (including understory shrubs and woody and semi-woody vines) were identified. Free standing plants < 5 cm DBH were counted (to give a density for each species) whereas for vine species their presence/absence was recorded. In each sub-plot, five sub-sub-plots each of 5 x 5 m were established, one at each corner (four corners) and one at the centre to record saplings and seedlings of woody species. In these sub-sub-plots all individuals of woody species < 5 cm DBH and > 1 m in height were identified and recorded as saplings. In addition, individuals of woody species of 0-1 m in height were identified, and recorded as seedlings.

Disturbance of the forest was evaluated by a number of criteria using a combination of quantitative and qualitative observations. To record tree cutting, all dead and live stumps occurring in the sampled sub-plots were recorded, (where possible) identified to species and sprouts were counted and recorded. Disturbance from grazing, non-timber forest product gathering and forest clearance for cultivation, was scored from a minimum of 0 (no disturbance) to a maximum of 4 (highly disturbed) in each sample sub-plot.

Plant species identification in the field was carried out with the help of a field key prepared for the Flora of Ethiopia (Edwards et al., 1995, 1997, 2000; Hedberg et al., 1989, 2003, 2006; Mesfin, 2004). Most of the common trees were easy to recognize and identify in the field. Voucher specimens of sterile plants and any others that were difficult to identify in the field were collected with standard information for identification using the reference specimens at the National Herbarium, Addis Ababa University. Classification of species by growth form was carried out in the field in all patches according to the criteria used in the Flora of Ethiopia

Table 4.2: Summary of site description information recorded from field data for each of 30 sampled forest patches in the Illubabor Administrative Zone of south-west Ethiopia.

Attribute	Description of variable generated for analyses
Disturbance	Visual assessment of grazing, logging, fuelwood collection, fire, and forest clearings rated from 0 (no disturbance) to 4 (highly disturbed) depending upon the perceived intensity of the disturbance
Altitude	Taken during field survey using GPS (Garmin 60)
Species richness and diversity of woody plants (trees, shrubs and vines)	Number of species, alpha diversity
Individual trees density	For all trees ≥ 5 cm DBH were counted
Species basal area	Calculated only for individuals ≥5 cm DBH
Species sapling density	Individuals < 5 cm DBH but > 1m height including individuals classified as shrubs or vines
Species seedling density	Individuals ≤ 1m height excluding individuals classified as shrubs or vines

4.3 Data Analyses

4.3.1 Species richness, diversity and structural composition

For these analyses data from all the sub-plots recorded in each patch were pooled to give a single total value for each patch. The mean species richness per patch was compared between the three sites using one-way ANOVA. A species-area curve (the accumulation of species as a function of the sample plot/area) was generated for all 30 patches using EstimateS (Colwell, 2010). In addition separate species-area curves were generated for each of the three sites for comparison. The number of observed species per patch and the non-parametric estimators Chao 1 and Jacknife-2 were used to estimate species richness (Magurran, 2004). Jacknife-2 was chosen because it calculates species richness and gives an estimate of absolute number of species in a community taking into account the occurrence of rare species (Magurran, 2004). Fisher's alpha diversity ($S = a*\ln(1+n/a)$, where S is the number of taxa, n is the number of individuals and a is Fisher's alpha) was calculated for each patch. Fisher's alpha is a good estimator of α-diversity because it is independent of the number of individual plants in a sample (Rosenzweig, 1995, Cayuela et al., 2006a) and assumes an underlying parametric model for the distribution of species' abundances. It is also useful as a measure of richness that is insensitive to sample size and its' adoption as the standard diversity index has been recommended by various authors (Rosenzweig, 1995).

The structure of a given forest is a product of forest dynamics and biophysical processes and a template for biodiversity and ecosystem function (Spies, 1998). Consequently, understanding forest structure can help to understand the history, function and future of a forest ecosystem (Spies, 1998). The structural composition of trees ($\geq$ 5cm DBH) recorded in each patch was described by their density (individual density/ha and basal area (m^2/ha)). The difference in tree density and basal area of trees between the three sites was tested using one-way ANOVA. The relationship of tree individual density and basal area with environmental (rainfall, altitude, aspect and slope), fragmentation (patch size, shape and edge density) and disturbance (cumulative score of disturbance from forest clearance, grazing and firewood collection) variables were tested using Pearson's correlation coefficient. All statistical analyses and regression analyses were performed using the QED Statistics 1.0 package (obtained from PISCES Conservation LTD, UK). QED Statistics is a Windows program that offers all the standard statistical methods used in science (Seaby and Henderson, 2007).

The importance value index of each tree species was calculated based on the sum of tree density (individuals/area), frequency (number of sub-plots in which each species was recorded/total number of sub-plots sampled) and dominance (basal area/total basal area of all species). This index provides an overall indicator of the abundance of plant species (Arro-Rodriguez and Mandujano, 2006). To investigate the past dynamics and regeneration status I carried out analyses of stand structure by classifying tree stems into different diameter size classes from (a) seedling and sapling to minimum diameter of 105 cm DBH, (b) from the lowest measured diameter (5 cm DBH) to a minimum diameter of the final size class of 60cm DBH and (c) from the lowest measured diameter (5 cm DBH) to minimum diameter of the final size class of 105 cm DBH to see the full range of size class distribution. The range between these minimum and maximum value for (a) and (c) was divided into 13 equal size classes with a 10 cm DBH interval, with a 13th class of trees > 105 cm DBH and for (b) the minimum and maximum value was divided into 11 equal size classes with a 5 cm DBH interval, with eleventh class of trees > 60 cm DBH.This diversity of approaches was adopted to minimise the risk of visual misinterpretation of distributions due to arbitrary decisions about the selection of size class intervals.

4.3.2 Community composition and fragmentation, disturbance and environmental gradient

Multivariate data analyses were employed to examine the similarities among and within the three sites (I, II and III). The process involved the production of a similarity matrix (Bray-Curtis similarity) and subsequent ANOSIM using the Community Analysis Software Package (Seaby and Henderson, 2007). Analysis of similarities (ANOSIM) has been widely used for testing spatial differences in community composition (Clarke, 1993; Chapman and Underwood, 1999; Clarke and Warwick, 2001). It uses the test statistic (R) based on the differences between the average of all the rank dissimilarities between species and between sites and the average of all the rank dissimilarities between species within the sites. ANOSIM generates a value of R which is scaled to lie between -1 and +1, a value of zero indicates no difference among a set of samples while R values > 0 indicate that species are more dissimilar between sites than within sites

(Clarke and Warwick, 2001). Negative R values indicate that dissimilarities within sites are greater than dissimilarities between the sites (Clarke and Warwick, 1994). In this study, R values less than 0.5 are interpreted to indicate little difference and values greater than 0.5 to indicate a substantial difference. To test for significance, the ranked similarity between sites was compared with the similarity that would be generated by random chance. The samples were randomly assigned to sites 1000 times and R was calculated for each permutation. A difference between sites was considered to be significant if the P-value was below 1% (0.01). Then, similarity percentage (SIMPER) analysis was used to determine which species accounted for the greatest percentage of similarity or dissimilarity between sites and within sites (Clarke and Warwick, 1994). SIMPER computes the percentage contribution of each species to the similarity or dissimilarity between all pairs of patches between sites and the percentage contribution of each species to the similarities between all pairs of patches within the site.

The relationships between the species composition of the 30 patches was then examined using Non-metric Multi-Dimentional Scaling (NMDS) ordination. Non-Metric Multi Dimensional Scaling expresses the similarity between sites in a reduced number of dimension by reducing a complex interrelationship to a simple Figure and provides a visual representation of how well a model accounts for variation in species composition between sites (Seaby and Henderson, 2007; Keleher and Rader, 2008). It also calculates the stress (residual modelling error) as a rank function instead of absolute dissimilarity values (Jongman et al., 1995). The analysis was run using the Bray-Curtis similarity matrix based on the presence/absence of species. The Bray-Curtis index gives less weight to outliers and is the recommended distance measure for NMDS (Southwood and Henderson, 2000). In NMDS two-dimensional ordination spaces, sample plots are represented by a point and the distance between points show the degree of dissimilarity or similarity between the samples (Hahs and Mcdonnell, 2007).

A second ordination technique, Canonical Correspondence Analysis (CCA), was used to relate the variation in species composition to environmental, fragmentation and disturbance variables to determine whether they correlated with community composition. This method examines direct vegetation change along known environmental gradients, e.g. topography, and indirect environmental causes of vegetation patterns by arranging species according to their floristic similarity (Ter Braak, 1987; McCune and Mefford, 1999, Ter Braak and Smilauer, 2002). It allows ordering species and plots along axes according to their relationships (Ter Braak, 1987; McCune and Mefford, 1999). The axes represent gradients in environmental, fragmentation and disturbance variables, and therefore provide a dimensional view of the distribution pattern of sites and species in relation to environmental variables (Tadesse, 2003). Relationships of the individual species to the environmental, fragmentation and disturbance gradients were then further tested by regression.

4.4 Results

4.4.1 Overall species composition, floristic pattern and structure of forest patches

In the total 9.6 ha (0.32 ha/patch x 30) of sampled forest across the 30 patches in all three sites, 254 species of woody plants belonging to 179 genera and 62 families were recorded. Of these, 42.6% of species were classified as trees, 33.5% as shrubs, and 23.6% as vines. All species were angiosperms with the exception of *Afrocarpus falcatus* (Gymnospermae) and a tree fern *Cyathea manniana* (Pteridophyta). The angiosperms were all dicot except *Phoenix reclinata* (a palm). A list of all the woody plant species recorded is given in Appendix 4.6).

Among the recorded species, trees were the most frequent life form (represented by 109 species), followed by shrubs (85 species) and vines (60 species). The species richness per patch ranged from 17 to 50 for trees; 4-33 for shrubs and 5-20 for vines (Figure 4.1). The most species-rich genus of trees was *Ficus* represented by 11 species. The most species-rich genus of shrubs was *Vernonia* also represented by 11 species and of vines were *Hippocratea, Ipomoea* and *Clematis* with three species each. The species richness at family level was high with 16 families represented by five or more species. These were Asteraceae (20 species), Rubiaceae (19), Euphorbiaceae (16), Moraceae (14), Fabaceae (13), Celasteraceae (10), Asclepiadaceae (9), Malvaceae (8), Sapindaceae (7), Lamiaceae, Cucurbitaceae (6) each, Acanthaceae, Oleaceae, Ulmaceae, Verbenaceae, Vitaceae (5) each, jointly accounting for 63.2% of the total species.

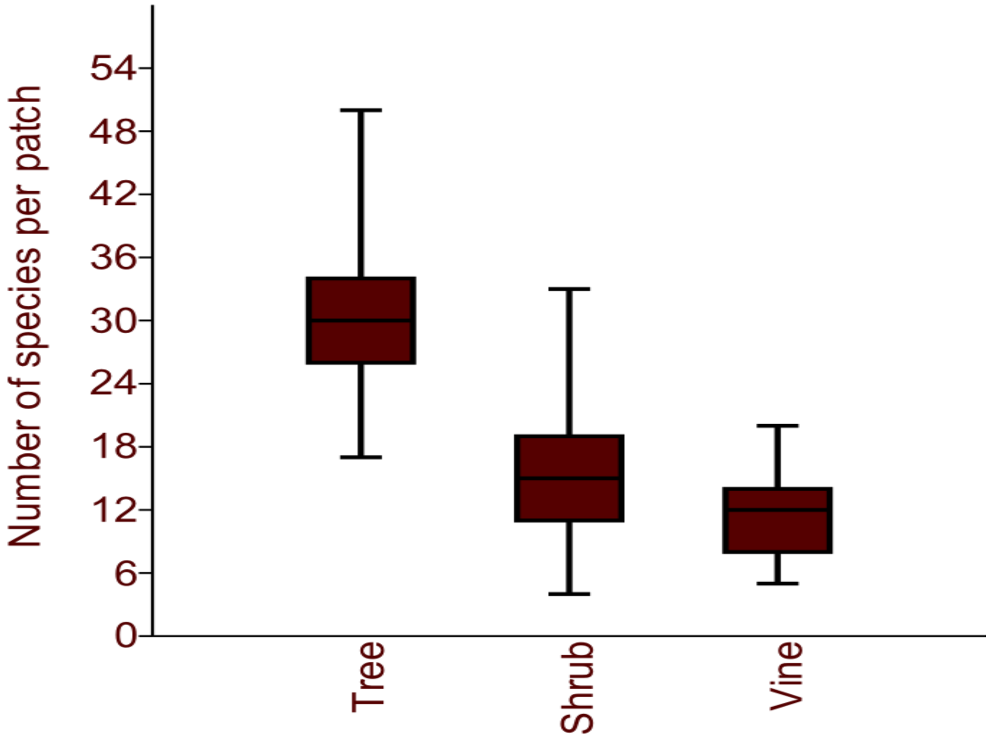


Figure 4.1 Number of woody species by life form per patch in 30 forest patches in the Illubabor Administrative zone of south-west Ethiopia. Results are shown as a box plot: the horizontal line through each box depicts the median; the box depicts the central 50th percentile of data, and the two whiskers indicate the highest and lowest values.

Ten families with > 100 individual plants of ≥ 5cm DBH had a combined density of 467.4 individuals ha^{-1}. Rubiaceae was the most abundant family with 105.7 individuals ha^{-1}, (largely comprising understorey trees); its most abundant species were *Coffea arabica, Oxyanthus speciosus* and *Psychotria orophila*. Second most abundant was Myrtaceae with 81.6 individuals ha^{-1} predominantly comprising large canopy trees, such as *Syzygium guineense* with 77.6 individuals ha^{-1}. Myrtaceae were especially abundant in the high altitude forest patches without extensive coffee management in sites II and

III. The third most abundant family was Fabaceae (its most abundant species were *Milletia ferruginea* with 49.2 individuals ha^{-1} and the canopy tree *Albizia gummifera* with 26.3 individuals ha^{-1}). Fabaceae was the dominant family of the species kept as shade or for their value in increasing soil fertility (pers. comm. with local farmers) in coffee managed forest in most of the Site I. The fourth most abundant family was Euphorbiaceae, with its most abundant species being *Croton macrostachyus* (28.0 individuals ha^{-1}) and *Macaranga capensis* (26.8 individuals ha^{-1}); it was concentrated in small patches or in heavily disturbed large patches/continuous forest. Myrsinaceae showed a similar distribution amongst patches (its most abundant species was *Maesa lanceolata* with 22.2 individuals ha^{-1}). The next three most abundant families were mainly represented by understory tree species: Oleaceae (with *Chionanthus mildbraedii* having 32.5 individuals ha^{-1}) followed by Rutaceae (with *Vepris dainellii* having 32.4 ha^{-1}) and Melianthaceae (with *Bersama abyssinica* having 31.7 individuals ha^{-1}). Two other abundant canopy tree species in rarer families were *Pouteria adolfi-friederic* in the Sapotaceae with 23.9 individuals ha^{-1} and *Prunus africana* in the Rosaceae with 10.9 individuals ha^{-1}.

The 32 afromontane endemic and near endemic species and those endemics to Ethiopia are distributed amongst 20 different families (Table 4.3). These species are of high local conservation concern in the future even though the majority of these species may not be considered to be globally threatened.

Table 4.3 Distribution types of the recognize conservation-important species collected in sample plots**.** Eth/e, Ethiopian endemics; AfrM/e, Afromontane endemics; AfrM/n-e, afromontane near endemics; Sub-AfrM/n-e, Afromontane sub-endemics. Sources: Friis, 1992; Tadesse, 2003; Schmitt, 2006.

Species	Family	Distribution	Growth form
Rhus glutinosa	Anacardaceae	Eth/e	Tree
Tiliacora troupinii	Menispermaceae	Eth/e	Climber
Phyllanthus immuensis	Euphorbiaceae	Eth/e	Shrub
Milletia ferruginea	Fabaceae	AfrM/e/Eth/e	Tree
Vepris dainelli	Rutaceae	AfrM/n-e /Eth/e	Tree
Coffea arabica	Rubiaceae	AfrM/e	Tree
Canthium oligocarpum	Rubiaceae	AfrM/e	Tree
Olea capensis	Oleaceae	AfrM/e	Tree
Bersama abyssinica	Melianthaceae	AfrM/e	Tree
Ritchiea albersii	Anacardiaceae	AfrM/e	Tree
Olea welwitschii	Oleaceae	AfrM/e	Tree
Galineria saxifraga	Rubiaceae	AfrM/e	Tree
Psychotria orophila	Rubiaceae	AfrM/e	Tree
Ilex mitis	Aquifoliaceae	AfrM/e	Tree
Macaranga capensis	Euphorbiaceae	AfrM/e	Tree
Brucea antidysenterica	Simarubaceae	AfrM/e	Shrub
Cassipourea malosana	Rhizophoraceae	AfrM/n-e	Tree
Prunus africana	Rosaceae	AfrM/n-e	Tree
Polyscias fulva	Araliaceae	AfrM/n-e	Tree
Apodytes dimidata	Icacinaceae	AfrM/n-e	Tree
Canthium giordanii	Rubiaceae	AfrM/n-e	Tree
Rhus quartiniana	Anacardaceae	AfrM/n-e	Tree
Senna petersiana	Fabaceae	AfrM/n-e	Tree
Pittosporum viridiflorum	Pittosporaceae	AfrM/n-e	Tree
Oxyanthus speciosus	Rubiaceae	AfrM/n-e	Tree
Podocarpus facatus	Podocarpaceae	AfrM/n-e	Tree
Teclea noblis	Rutaceae	AfrM/n-e	Tree
Trichilia dregeana	Meliaceae	Sub-AfrM/n-e	Tree
Eugenia bukobensis	Myrtaceae	Sub-AfrM/n-e	Tree
Elaeodendron buchananii	Celasteraceae	Sub-AfrM/n-e	Tree
Albizia grandibracteata	Fabaceae	Sub-AfrM/n-e	Tree
Dracaena steudneri	Dracaenaceae	Sub-AfrM/n-e	Tree

4.4.2 Species richness and diversity

The mean (±1.96*SE) woody species richness per patch varied significantly amongst the three sites (ANOVA, F = 8.14, $p < 0.05$): at Site I it was 70.8±9.2, at Site II it was 50.9±6.3 and at Site III it was 54.3±4.9 (Figure 4.2). There was a large range of Fisher’s alpha diversity (Whittaker, 1972) values amongst the forest patches. Within, the lower altitude/larger forest block, Site I mean diversity was 37.0±1.28 with a large range amongst patches of 27.4-68.68. For both the upper altitude sites, II and III, the mean was notably lower (21.9±1.82 and 21.7±1.80 respectively) with a much narrower range amongst patches, 20 of the patches had diversity values between 18.8 and 24.2 with only one patch in each site having a higher value (41.4 in Site II and 37.7 in Site III). The mean alpha diversity of Site I was significantly different from Site II (ANOVA, $F = 12.9$, $p < 0.05$) and III (ANOVA, $F = 10.49$, $p < 0.05$). In contrast, alpha diversity was not significantly different between sites II and III (ANOVA, $F = 0.64$, $p > 0.05$, Appendix 4.5).

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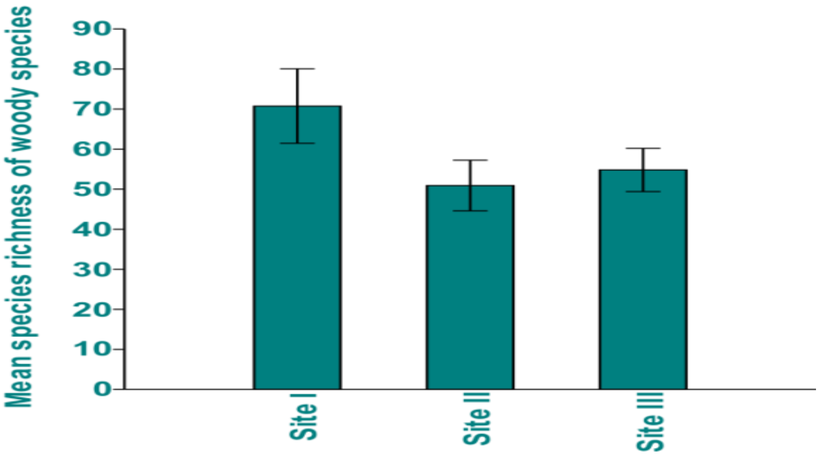


Figure 4.2 Species richness of wood plants (mean±SE*1.96) of forest patches in three sites in the Illubabor Administrative zone of western Ethiopia.

The species accumulation curve for all 240 sub-plots pooled together (one value per sub-plot) had completed the steep section of the curve after ca. 80 sub-plots (3.2 ha), however even after the accumulation of species in all 30 patches (240 sub-plots, 9.6 ha) the curve was still showing a steady increase indicating that the sample was still a long-way below the total species richness of the whole regional species pool (Figure 4.3a). The tight 95% confidence intervals around the mean curve give confidence in its representativeness across the three study landscapes. Separate species area curves for the sub-plots within each of these three landscapes again show a clear distinction between the lower altitude/larger forest block SiteI and the other two sites (Figure 4.3b). There was a much more rapid initial accumulation of species in Site I, however after about 20 sub-plots (1 ha) the gradient of the curve of Site I had reduced to less than that of sites II and III which continued to accumulate species at a slightly higher rate. None of the three curves had approached a plateau by the end of the number of sub-plots sampled. The curves for sites II and III remained very similar throughout.

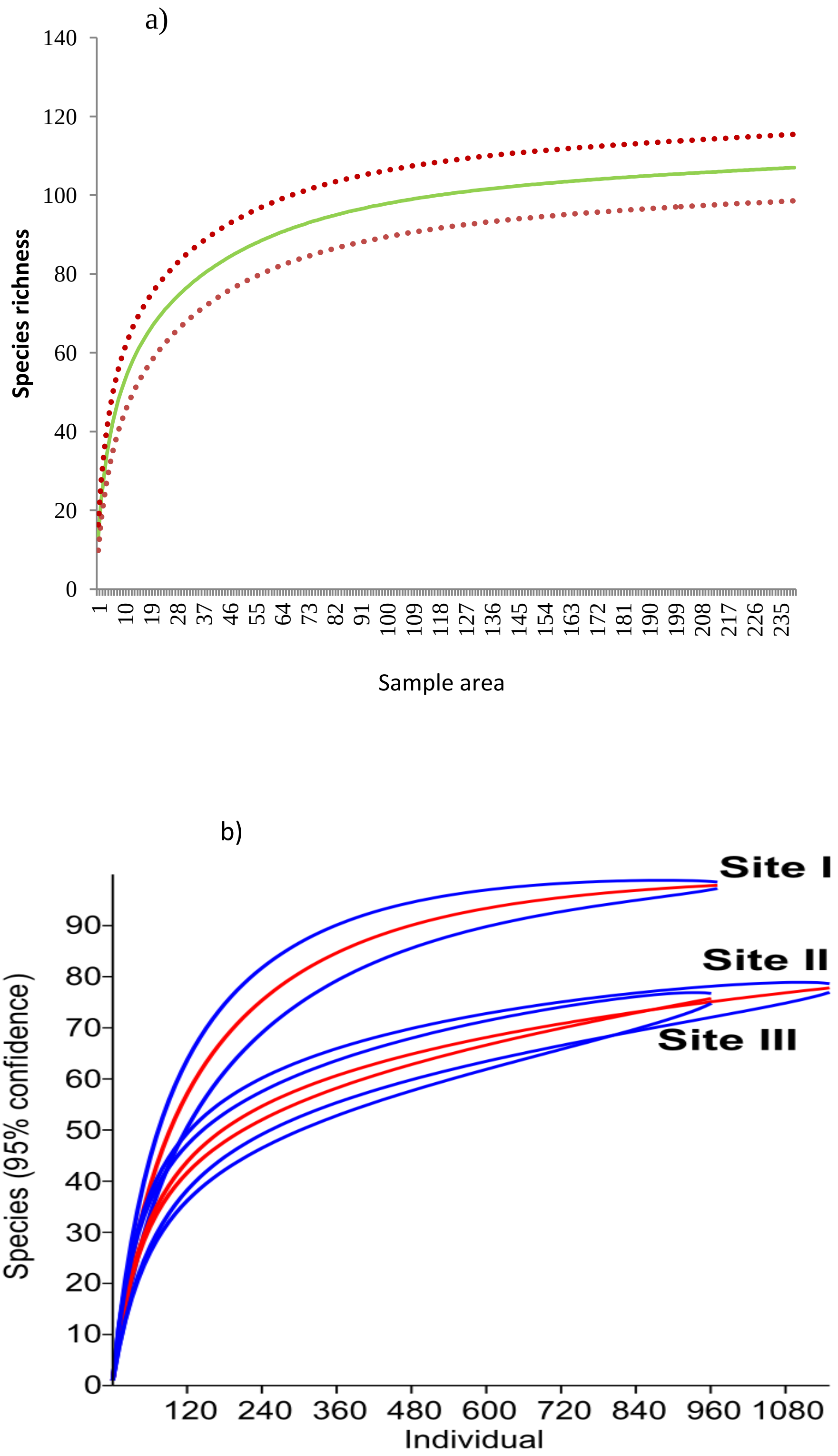
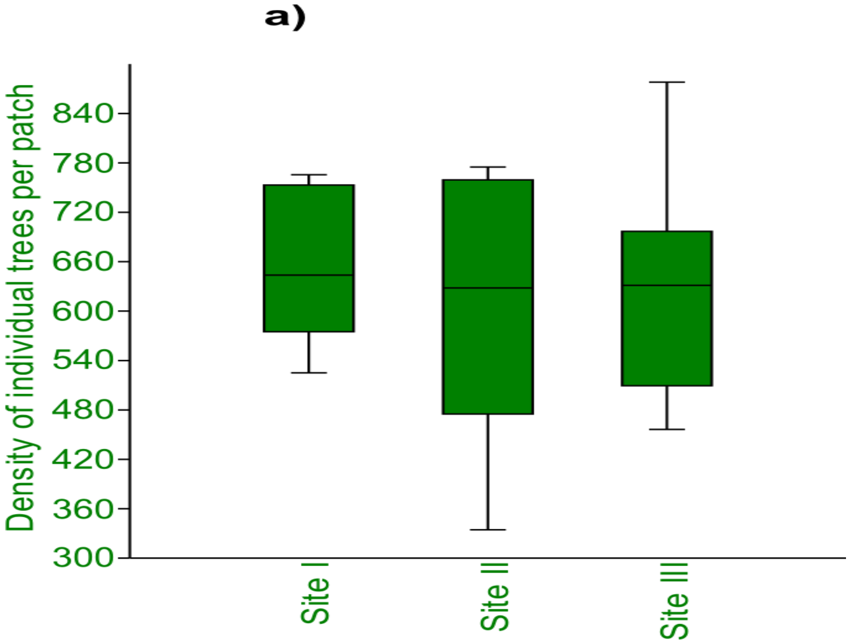


Figure 4.3 Area- and individual-based woody species accumulation curves for 30 forest patches in the Illubabor Administrative Zone of south-west Ethiopia (a) area-based curve for all three sites pooled, (b) individual-based curves for each site. The mean curve is shown as a solid line with the upper and lower 95% confidence intervals as dotted lines in (a) and solid lines coloured blue (b).

4.4.3 Stand composition and structure

A total of 7446 tree individuals ≥ 5cm DBH was measured and identified in the 240 plots across the three sites. When quantified separately, 1600 individuals in Site I, 2660 individuals in Site II and 2306 individuals in Site III were measured. Trees down to 5.0 cm DBH were found in all three sites. The maximum DBH in Site I was 365.6 cm (a *Ficus sur* tree in Gore Zuria), in Site II 550.8 cm (a *Syzygium guineense* tree in Tuluboka) and in Site III 434.5 cm (a *Prunus africana* tree in Jireen). The mean±SE density of individual trees ≥ 5cm DBH in Site I was 662±31 ha^{-1} with a range from 525 ha^{-1} at Baljo to 766 ha^{-1} at Gore-Zuria (Figure 4.4a, Table 4.4). In Site II the mean tree density was slightly lower at 607±42 ha^{-1} ranging from 334 ha^{-1} at Qollo to 775 ha^{-1} at Bongi (Figure 4.4a). At Site III mean tree density was intermediate at 624±39 ha^{-1} with a range from 456 ha^{-1} at Qotora2 to 878 ha^{-1} at Dike (Figure 4.4a, Table 4.4). The difference among the three sites was not significant ($p > 0.05$). Likewise, the mean total tree basal areas were similar (and not significantly different) amongst the three sites: in I it was 94.2±4.7 m^2 ha^{-1} (range 74.0-110.9 m^2 ha^{-1}), in II 99.2±3.4 m^2 ha^{-1} (range 68.0-113.8 m^2 ha^{-1}), and III 89.8±3.7 m^2 ha^{-1} (range 68.4-101.9 m^2 ha^{-1}) (Table 4.4, Figure 4.4b).



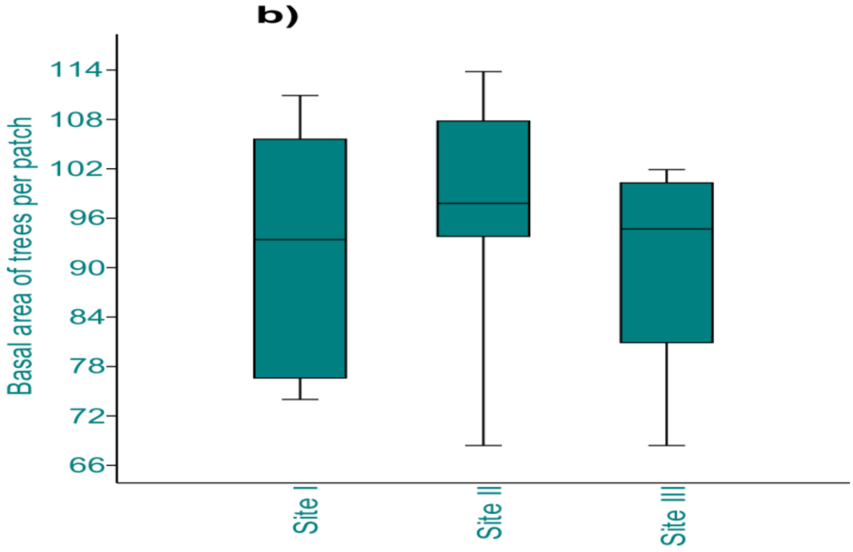


Figure 4.4 Box plot diagram depicting the distribution of individual tree (a) density and (b) basal area per forest patch in each of three sites in the Illubabor Administrative zone of south-west Ethiopia. The horizontal line through each box depicts the median; the box depicts the central 50th percentile of data, and the ends of the whiskers show the full range of values.

The basal area per patch was positively correlated with disturbance ($r = 0.405$, $p < 0.05$). However, the basal area per patch was also heavily influenced by the inclusion within the sample plots of very large trees of a limited number of dominant species (Table 4.5). There was a noTable distinction between Site I (with three dominant species, of which one *Celtis africana*, was rare in the other two sites) and sites II and III (which were more similar).

Table 4.4 Tree species richness, basal area, stem density and height at sites I, II and III. Species richness is shown as the cumulative total for all eight sub-plots sampled in each patch, basal area and density are shown as the mean (± SE) for the eight sub-plots, and mean tree height

Site I

Patch name	Species richness	Mean basal area (m^2/ha)	Mean stem density (no./ha)	Mean Height (m)
Gorezuria	23	93.4±2.5	765.6±20.4	12.5
Burusa	33	74.0±1.4	631.3±18.0	12
Baljo	34	96.25±1.9	525±7.3	15.5
Gaba	34	92.8±1.8	740.6±8.0	17.7
Uka	35	110.9±1.4	575±31.8	13.4
Masha	44	104.4±0.9	753.1±7.8	16.5
Metuzuria	48	76.6±1.2	643.7±8.0	8
Leka	50	105.6±1.2	662.5±5.7	16

Site II

Patch name	Species richness	Mean basal area (m^2/ha)	Mean stem density (no.//ha)	Mean height (m)
Qolowarabesa	17	113.8±3.0	334.4±7.9	10.5
Lagchancho	20	102.5±2.2	768.8±15.8	13.6
Markafa	26	97.5±3.5	418.8±5.5	16.8
Yaroo	27	113.1±2.8	593.8±8.1	15.5
Sibu1	28	102.8±2.1	475±7.6	15.6
Malate	28	90.3±1.6	650±9.5	15.6
Bongi	28	68.4±1.9	775±10.9	8.9
Tulumako	29	97.8±1.9	740.6±1.7	15
Sibu2	32	105.3±1.7	512.5±3.5	14.5
Gobe	32	96.9±1.9	759.4±9.4	14.5
Tuluboka	35	93.8±1.7	628.1±6.4	12.4
Jorgo	36	107.8±1.4	631.3±4.3	16.8

Site III

Patch name	Species richness	Mean basal area (m^2/ha)	Mean stem density (no./ha)	Mean height (m)
Qotora1	18	100.3±3.4	509.4±7.1	15.6
Busajreen	20	80.9±2.1	668.8±8.8	14.7
Jireen	24	84.4±2.1	565.6±8.9	16.6
Issya	25	92.5±1.9	503.1±8.4	16.5
Buyina	26	68.4±1.5	631.3±11.4	10
Simbir	29	75.9±1.2	734.4±12.8	12.6
Dike	30	101.9±1.6	878.1±7.8	16.7
Koda	32	97.5±1.4	696.9±6.9	15.5
Qotora2	34	101.6±1.3	456.3±2.5	13.7
Jamia	38	94.7±1.2	596.9±5.9	14.5

Table 4.5 Dominant species in terms of proportion of total basal area in forest patches in each of three sites in the Illubabor Administrative zone of south-west Ethiopia. All species contributing ≥ 4% of basal area are shown.

	Proportion of total basal area (%)		
Species	Site I	Site II	Site III
Milletia ferruginea	10.9	4.9	4
Celtis africana	10.2	-	-
Albizia gummifera	8.5	7.8	4.1
Syzygium guineense	-	19.6	16.8
Trilipsium madagascariense	-	8.9	-
Pouteria adolfi-fredericii	-	5.6	13.6
Macaranga capensis	-	5.3	6
Croton macrostachyus	-	4.9	5.7
Prunus africana	-	4.2	10.3
Afrocarpus falcatus	-	-	5

For an overall assessment of species abundance, relative density and relative frequency were added to relative dominance (basal area) to give an “importance value” (Table 4.6). The most abundant tree species in Site I, e.g. *Milletia ferruginea*, were pioneer species indicative of a high level of forest disturbance. In this site (and connected with the disturbance) the most abundant species in the lower storey (< 10 m height) was *Coffea arabica* (IVI = 7.8%, Table 4.6), which had the highest relative density and relative frequency of all species in this site (Appendix 4.1). Most of the species that are common understorey trees in sites II and III, such as *Chionanthus mildbraedii*, *Vepris dainellii* and *Oxyanthus speciosus*, were either absent or rare in Site I (Table 4.6).

With regard to frequency, a few species were widely distributed amongst the patches. In Site I, two species, *Milletia ferruginea* (73%) and *Coffea arabica* (84%), occurred in more than half of the plots (one plot per patch) (Appendix 4.1). Sites II and III had more high frequency species: five in Site II, *Syzygium guineense* (85%), *Croton macrostachyus* (68%), *Chionanthus mildbraedii* (56%), *Macaranga capensis* (55%) and *Bersama abyssinica* (55%) (Appendix 4.2), and five in Site III, *Syzygium guineense* (92%), *Croton macrostachyus* (66%), *Bersama abyssinica* (61%), *Allophylus abyssinicus* (58%) and *Pouteria adolfi-frederici* (52%) (Appendix 4.3).

Table 4.6 Tree species with importance values ≥ 2% in forest patches in each of three sites in the Illubabor Administrative zone of south-west Ethiopia**.** The importance values were calculated for all trees ≥ 5cm DBH and the tree species are divided into two groups (upper and middle storey) according to the height of their tallest individuals.

	Importance value (IVI)	
Site I	Site II	Site III
Upper storey (15-30 m height)	**Upper storey** (15-45 m height)	**Upper storey** (15-40 m height)
Albizia gummifera (IVI 4.7)	*Syzygium guineense* (IVI 13.4)	*Syzygium guineense* (IVI 13.2)
Celtis africana (IVI 3.9)	*Albizia gummifera* ((IVI 4.9)	*Pouteria adolfi-frederici* (IVI7.8)
	Pouteria adolfi-frederici (IVI 3.8)	*Prunus africana* (IVI5.2)
	Elaeodendron buchananii (IVI 2.5)	*Albizia gummifera* (IVI3.8)
		Afrocarpus falcatus (IVI2.7)
Middle storey (10-15 m height)	**Middle storey** (10-15 m height)	**Middle storey** (10-15 m height)
Milletia ferruginea (IVI 8.5)	*Croton macrostachyus* (IVI4.9)	*Croton macrostachyus* (IVI5.1)
Croton macrostachyus (IVI 3.5)	*Milletia ferruginea* (IVI4.9)	*Macaranga capensis* (IVI5.1)
Maesa lanceolata (IVI 3.4)	*Macaranga capensis* (IVI4.8)	*Bersama abyssinica* (IVI4.4)
	Bersama abysinica (IVI 3.0)	*Milletia ferruginea* (IVI4.0)
		Allophylus abyssinicus (IVI3.2)
		Apodytes dimidata (IVI3.0)
		Canthium oligocarpum (IVI2.9)
Understory (< 10m height)	**Understory** (< 10m height)	**Understory** (< 10m height)
Coffea arabica (IVI 7.8)	*Vepri dainellii* (IVI3.3)	*Vepris dainellii* (IVI3.2)
	Oyxanthus speciosus (IVI3.1)	*Oyxanthus speciosus* (IVI2.9)
	Chionanthus mildbraedii (IVI4.3)	*Chionanthus mildbraedii* (IVI2.0)

The patches in each site vary in density and composition of understory shrub and liana species. Using a density threshold of > 5 individuals per hectare, abundance in the understorey was shared amongst a greater number of species in Site I (29) than Site II (18) or Site III (20) (Table 4.7).

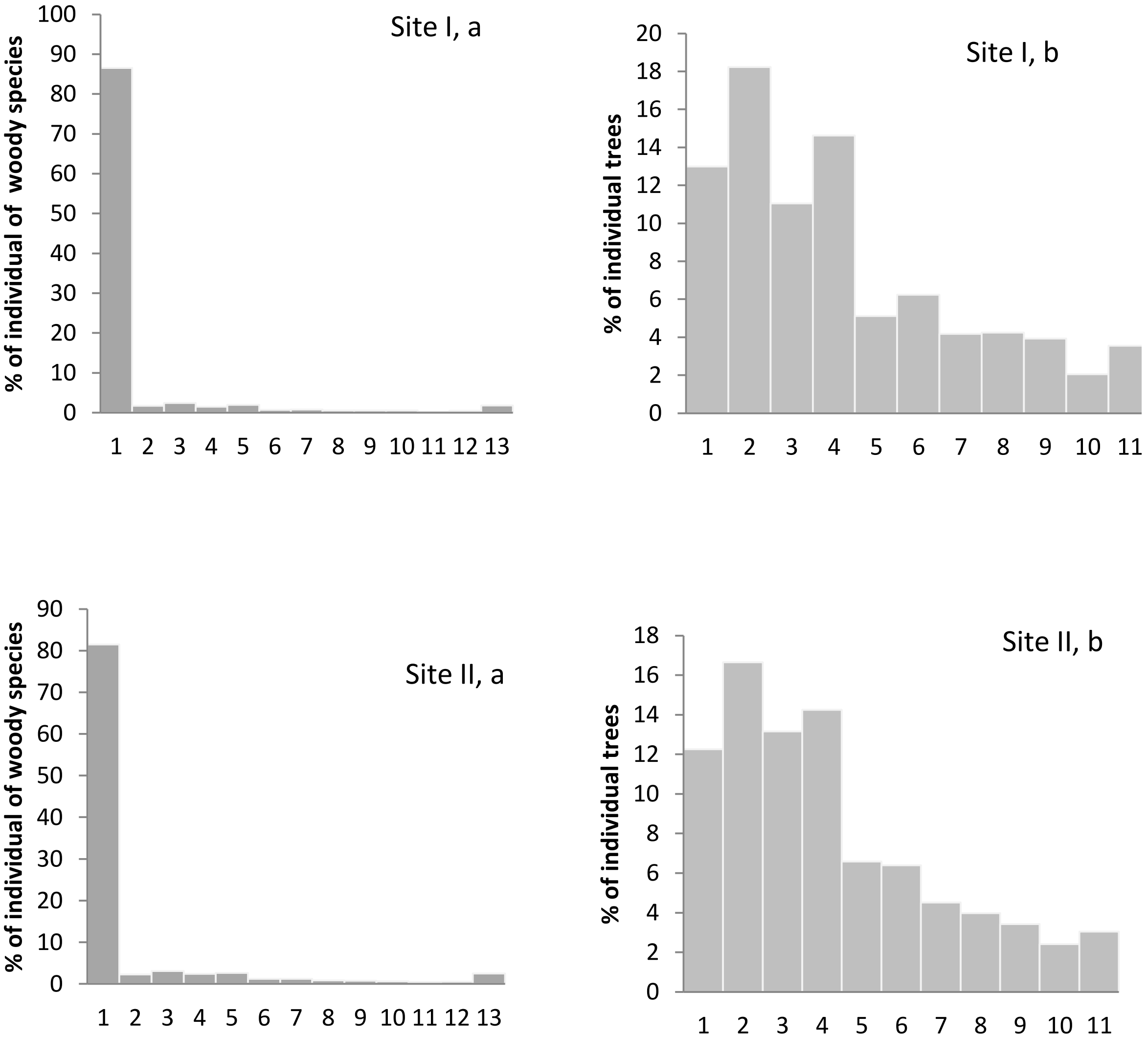
Table 4.7 Density (individuals/ha) of the most abundant shrub and liane species (with > 5 individuals/ha) in the understorey (< 10 m height) in forest patches in each of three sites in the Illubabor Administrative zone of south-west Ethiopia.

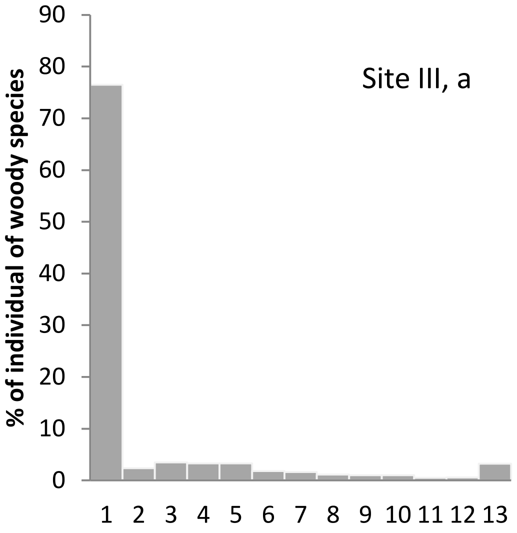
Species	Site I	Site II	Site III
Justcia schimperiana	134.4	35.4	37.5
Maytenus gracilipes	131.3	49.5	135.3
Clausena anisata	128.9	9.4	83.1
Erythrococcaa trichogyne	60.5	21.6	57.8
Rytigina neglecta	50.8	20.1	126.6
Vernonia glabra	47.3	-	-
Phyllanthus ovalifolius	35.9	10.9	-
Cyathula uncinulata	33.2	25.0	129.1
Pavetta abyssinica	26.6	-	7.2
Pavetta oliveriana	26.6	13.8	46.6
Rubus apetalus	25.8	10.7	28.4
Vernonia auriculifera	22.3	-	-
Premna schimperi	21.5	-	-
Aframomum corrorima	16	43.8	-
Hibiscus micranthus	15.2	-	5.9
Hibiscus ludwigii	13.7	-	-
Vernonia hochstetteri	13.3	-	13.4
Vernonia urticifolia	12.5	23.1	-
Microglosum pyrifolia	11.3	-	-
Rubus steudneri	10.2	-	7.8
Acalypha psilostachya	9.8	-	-
Acalypha racemosa	9.8	8.6	-
Pentas lanceolata	9.4	-	-
Paulina pinnata	7.4	-	-
Acalypha ornata	7	-	-
Solanium schimperiana	7	6.3	-
Acanthopale pubescens	5.9	45.6	121.6
Ocimum suave	5.5	-	-
Acanthus eminens	-	32.6	88.8
Senna petersiana	-	15.4	10.0
Solanacieo mannii	-	11.7	11.6
Solanacieo gigas	10.9	-	33.8
Embelia schimperi	-	5.2	5.9
Discopodium penninervium	-	-	11.6
Pychnostachys abyssinica	-	-	9.1

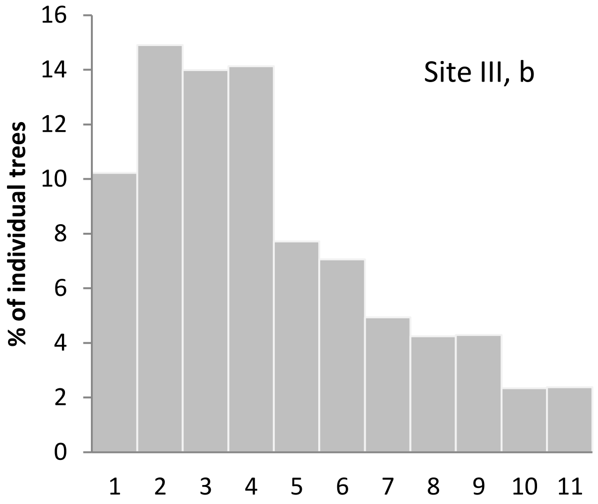
The size-class distributions of all woody species were examined in three different forms. The first form included all sampled individuals of woody plant species (including those < 5 cm DBH). The second form included all sampled individuals of woody species ≥ 5 cm DBH. The first and second forms both used 5 cm size-class intervals. The third form included all individuals of woody species ≥ 5 cm using 10 cm size-class intervals to show the full range of sizes. The all-individuals diameter size-class distributions with 5 cm interval showed a strikingly similar pattern across the three study sites (Figure 4.5a & b). There was consistently a very high density of individuals in the smallest size class (> 0 m height and < 5 cm DBH), which accounted for > 68% of the individuals recorded in each site (Figure 4.5a). In all three sites there was a large reduction in density from this < 5 cm DBH class to the subsequent 5-9.9 cm DBH class: 98.4% for Site I, 96.8% for Site II and 95.4% for Site III. Above 5 cm DBH, it is noTable that in all three sites there is a general decline in density with size, yet density is lower in the 5-9.9 cm than the subsequent 10-14.9 cm class, there is an uneven trend across the three classes 10-24.9 cm and then a steep decline to the 25-29.9 cm class, after which there is a more consistent stepped decline in density (Figure 4.5b). This may be attributed to the human-caused disturbance (e.g. tree cutting) being concentrated in trees up to 30 cm DBH, with dynamics of trees > 30 cm DBH being more a result of natural population processes. The size class distribution of trees ≥ 5cm DBH with 10 cm size-class intervals showed similar pattern for Site I and III (Figure 4.5c) but indicated a lower density in the smallest size class (5-14.9 cm DBH) in Site III.

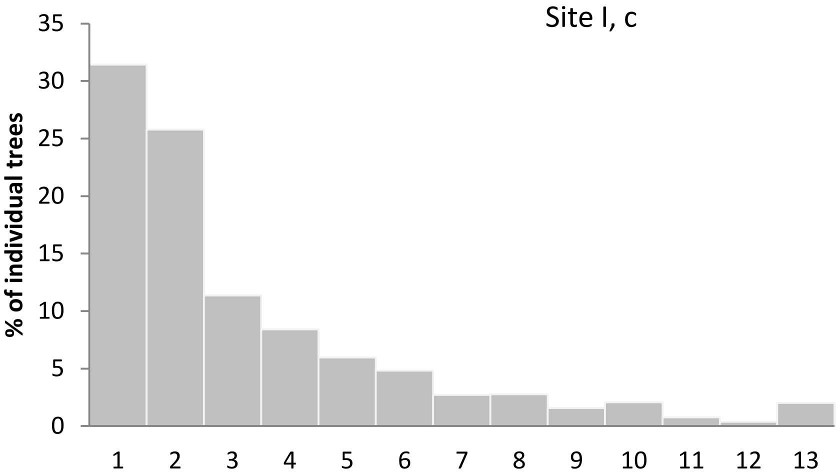
4.4.4 Regeneration status of tree species in all patches at each site

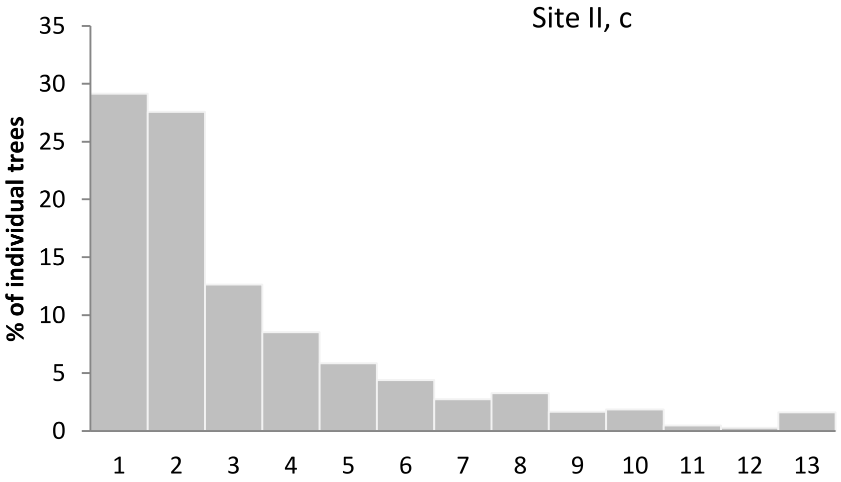
A total of 22,423 seedlings and saplings (> 0 m < 1m tall for seedlings and > 1m but < 5 cm DBH for saplings) of tree species were recorded in all plots in all sites. The seedling and sapling size class represents a majority of the population of all tree species together (76.7%) (Figure 4.5a). The density of seedlings and saplings was greater in Site I (9302 in the sample area of 25,600 m^2), than Site II (6284/38,400 m^2) and Site III (6837/32,000 m^2). The composition and density of tree species in the sapling and seedling size class varied greatly amongst the three sites (Appendix 4.4). *Coffea arabica,* a crop species, had the highest density in Site I (1483 individuals < 1 m height/25,600 m^2). Second-fourth in rank were the pioneer species *Albizia gummifera* (778/25,600 m^2), *Milletia ferruginea* (718/25,600 m^2) and *Maesa lanceolata* (388/25,600 m^2). The former two species are intentionally retained and even sometimes transplanted into areas managed for coffee cultivation due to their value in increasing soil fertility and coffee shade (pers.com with local farmers). However, local farmers stated no benefits for *Maesa lanceolata* and its abundance may simply be an indicator of human disturbance as it usually regenerates in forest gaps and open areas (personal observation). The species with highest density in Sites II and III combined was *Oxyanthus speciosus* (561/70,400 m^2) (Appendix 4.4). Second-fourth in ranking were *Pouteria adolfi-friederici* (531/70,400 m^2), *Chionanthus mildbraedii* (487/70,400 m^2) and *Bersama abyssinica* (336/70,400 m^2).




Site III, a
% of individual of woody species
0
10
20
30
40
50
60
70
80
90
1 2 3 4 5 6 7 8 9 10 11 12 13


Site III, b
% of individual trees
0
2
4
6
8
10
12
14
16
1 2 3 4 5 6 7 8 9 10 11


Site I, c
% of individual trees
0
5
10
15
20
25
30
35
1 2 3 4 5 6 7 8 9 10 11 12 13


Site II, c
% of individual trees
0
5
10
15
20
25
30
35
1 2 3 4 5 6 7 8 9 10 11 12 13

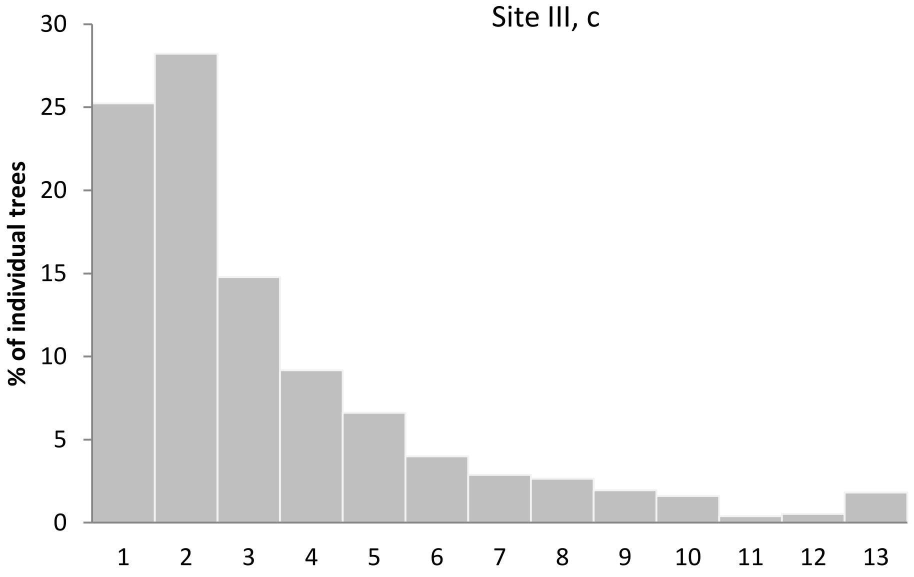


Figure: 4.5 All individual woody plant species size class distributions (for all species) of forest patches in each of three sites in the Illubabor Administrative zone of south-west Ethiopia. Density of stems in each size class is shown as the percentage of all individuals in the sample plots. The distributions are shown for two size groups: (a) for all individuals of woody species < 5 and ≥ 5 cm DBH, (b) and (c) for all individuals of woody species ≥ 5 cm DBH. Size class intervals (DBH) are 5 cm for a and b and 10 cm for (c) with the minimum of the first interval being 0 cm DBH in (a) and 5 cm DBH in (b) and (c).

4.4.5 Compositional similarity

Analysis of similarity (ANOSIM) showed that similarity in woody plant species composition amongst patches within each site was greater than similarity amongst sites, as the Global R value is positive (+0.543, p = 0.001, Table 4.8). Pairwise tests showed that patches in Site I had a distinct species composition from those in Site II (R = +0.87, p = 0.001) and those in Site III (R = +0.83, p = 0.001), but patches in sites II and III were not significantly different (R = +0.12, p = 0.037, at the 1% level i.e. p > 0.01)). Similarity percentage (SIMPER) analysis showed that the average dissimilarity in community composition between Site I and Site II was 65.6% and this dissimilarity was partially attribuTable to the high abundance in patches of 11 species. The average dissimilarity in community composition between Site I and Site III was 66.4%, and was attributed to the high abundance of nine species. The dissimilarity between Site II and Site III was least, 48%, and. was accounted for by the ten times higher abundance of

Pouteria adolfi-fredrici in Site III, probably due to the high rate of logging in Site II (pers. obs.). Other species contributing to dissimilarity between sites II and III were *Nuxia congesta*, *Teclea noblis*, *Flacourtia indica* and *Ekbergia capensis*, which were four times more abundant in Site II. *Apodytes dimidata*, *Chionanthus mildbraedii*, *Albizia schimperiana*, *Lepidotrichilia volkense*, *Prunus africana* had twice the abundance in Site II.

Table 4.8 Analysis of similarity (ANOSIM) of woody species composition for pairwise comparisons of patches amongst and within the three study sites in the Illubabor Zone of south-west Ethiopia.

	Permutation	P Value	R
ANOSIM for all three sites (Global R)	1000	0.001*	0.543
Pairwise tests			
Site I (n = 8) vs Site II (n = 12)	1000	0.001*	0.875
Site I (n = 8) vs Site III (n = 10)	1000	0.001*	0.837
Site II (n = 12) vs Site III (n = 10)	1000	0.037$^{n.s}$	0.118

* indicate significance at 0.01, $^{n.s}$ not significant at 0.01

Two dimensional non-metric multi-dimensional scaling of woody plant composition in, firstly, all 30 sampled patches and, secondly, each of the 8 sub-plots per patch, also clearly separated patches in Site I from those in sites II and III on the first axis (Figure 4.6a). However, this distinction was clearer at the whole patch (Figure 4.6a) than sub-plot (Figure 4.6b) level. Axis 2 predominantly separated patches within each site (especially for Site I). Site II and Site III show strong overlap. The Site III axis 1 outlier patch, Jireen, and site 2 axis 2 outlier patch, Qollo were noTable for their different species composition (Figure 4.6a). Plot distribution amongst the sites also showed variation in species composition. In the ordination space the plots of Site I are clustered on the right side while almost plots from Site II and III overlapped (Figure 4.6b).

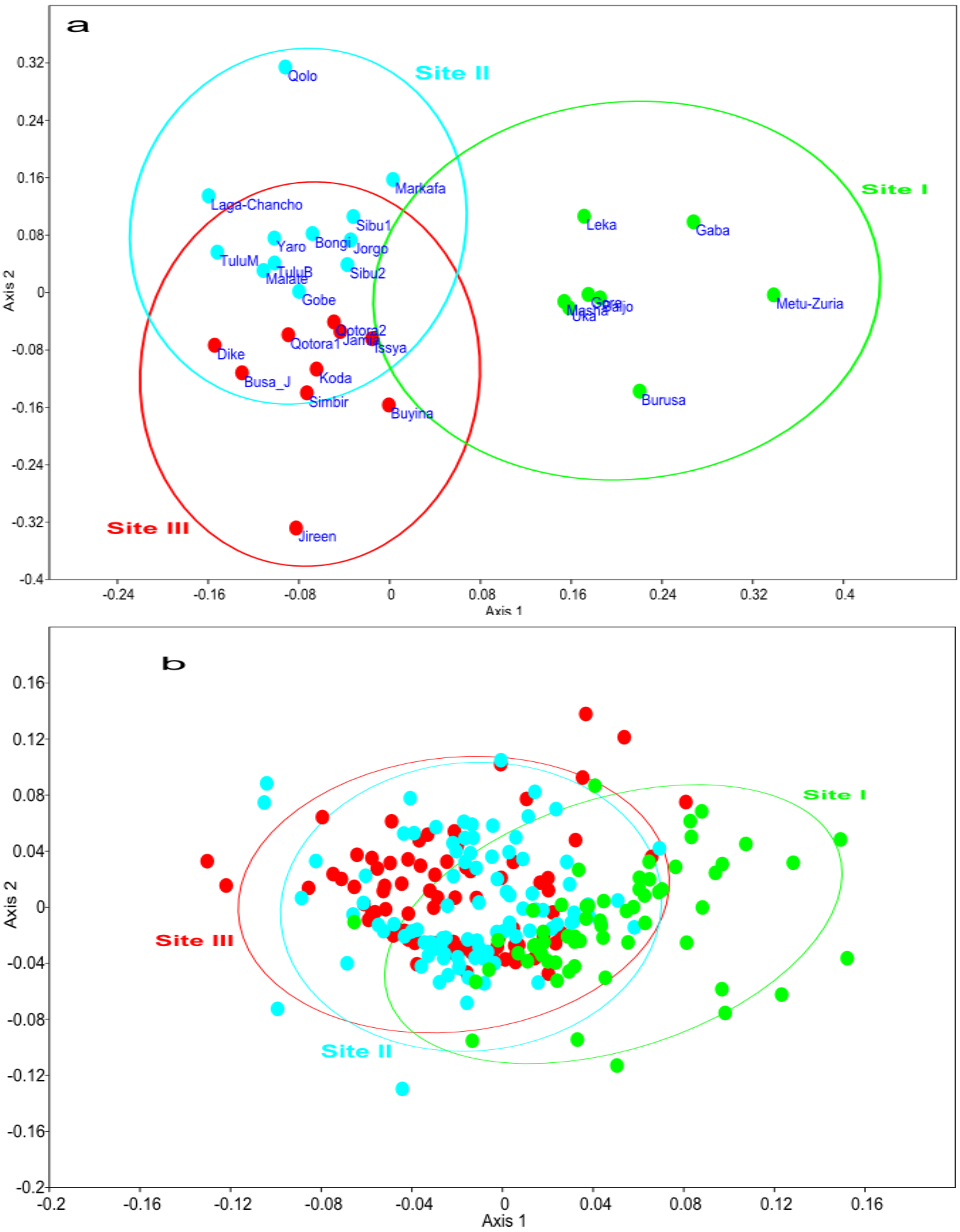


Figure 4.6 Non-metric multi-dimensional scaling Axis 1 vs Axis 2 – 2D model-sampled using a Bray-Curtis similarity matrix for: (a) individual patches, (b) individual sub-plots within the patches at 95% concentration ellipse level for the three study sites in the Illubabor Zone of south-west Ethiopia. The basic NMDS algorithm calculates the similarity or dissimilarity between sites or sub-plots and assign to each site or sub-plot a set of coordinate in ordination space.

4.4.6 Effect of fragmentation, disturbance and environmental gradients on woody species community composition

Canonical Correspondence Analysis (CCA) ordination showed the distribution of the 8 20 x 20 m sub-plots within each patch and species against vectors of disturbance, patch shape, edge density, altitude, rainfall and slope in ordination space (Figure 4.7). Other environmental and fragmentation variables such as patch aspect, patch area and fractal dimension were discarded after testing multicollinearity since they were highly co-correlated. The first CCA axis explain just 14% of the variance in species composition, while the first three CCA axes together explain only 25%. However, the correlation of species/explanatory variables scores are high, 0.93 for the first axis, 0.94 for the second axis and 0.89 for the third axis (Table 4.9) indicating that most of the variation in species composition can be accounted for by the variables linked to these three axes. Monte Carlo analysis (McCune and Mefford, 1999) with 1000 replicates showed that the amount of variability explained by the tested environmental, fragmentation and disturbance variables was significant ($p < 0.01$) for axes 1, 2 and 3 (Table 4.10).

Table 4.9 Variance explained by the first three axes of a Canonical Correspondence Analysis of forest patches in the Illubabor Administrative Zone of south-west Ethiopia.

Axis		1	2	3	Total variance
Eigen value		0.265	0.121	0.089	1.89
% variance explained		13.991	6.424	4.701	
Cumulative % variance explained		13.991	20.415	25.116	
Multiple correlation species/environment scores		0.932	0.946	0.890	
Total Eigen value	0.628				

Table 4.10 Result of Monte Carlo test (with 1000 runs) for the significance of linear correlation between the environmental variables and species matrices for each of the first three axes of the Canonical Correspondence Analysis.

Axis	1	2	3
Eigenvalues	0.265	0.121	0.089
Mean	0.122	0.080	0.064
Maximum	0.266	0.119	0.093
Minimum	0.072	0.056	0.046
Probability at 5% level	0.002**	0.001**	0.002**

** significant at $p < 0.01$

To best use the CCA ordination to investigate the association between forest species composition and the tested environmental variables, axes 1 and 2, and axes 1 and 3, were plotted (Figure 4.7a &b). The CCA ordination showed the same pattern as the NMDS with axis 1 clearly distinguishing patches in Site I from those in sites II and III (which showed great overlap). The environmental variables most strongly linked with the first axis of the CCA are altitude and patch shape (negative, associated with sites II and III) versus disturbance (positive, associated with Site I) (Figure 4.7a&b). The only environmental variable strongly linked to the second axis of the CCA is edge density (Figure 4.7a). Disturbance showed the strongest association with axis 3.

In the CCA ordination, the distribution of 21 species was significantly correlated with altitude (four positively and 17 negatively) (Table 4.11). *Coffea areabica* and *Trichilia emetica* were most strongly negatively correlated with altitude. Three species were significantly positively correlated with disturbance and four negatively. It is noTable that this CCA-based analysis distinguishes species associations with the different environmental variables: of the seven species significantly correlated with disturbance, only one (*Apodytes dimidiata*) showed a significant strong correlation with altitude (positive). Three species were significantly positively correlated with edge density and six negatively. Of these none were significantly correlated with disturbance, and only one with altitude (*Ficus vallis-choudea*, weakly negative with edge density and negative with altitude). The six species negatively correlated with edge density were all observed to be concentrated in the forest understorey where there was dense shade (except for *Albizia grandbracteata*). Eight species were significantly positively correlated with patch shape index and seven negatively. More of these species also showed significant correlations with other environmental variables: *Allophylus abyssinicus* was positive for patch shape and altitude, *Olea capensis* and *Trichilia dregeana* were positive for patch shape and negative for disturbance, while *Ficus vallis-choudae*, *Trichlia emitica* and *Vanguria apiculata* were negative for shape index and for altitude, and *Albizia grandbracteata* was negative for patch shape and edge density.

Table 4.11 Coefficients of correlation between species and environmental variables in the CCA ordination of forest patches in the Illubabor Administrative Zone of south-west Ethiopia. All correlations with a p value < 0.05 are shown resulting in correlations with four environmental variables: altitude, disturbance, edge density and patch shape.

Species	r-value	p-value	Species	r-value	p-value
Correlation with altitude			**Correlated with edge density**		
Trichilia emtica	-0.78	0.00007	*Dalbergia lactea*	0.64	0.0001
Ehertia cymosa	-0.63	0.0001	*Rhus glutinosa*	0.59	0.0005
Vanguria apiculata	-0.63	0.0002	*Allophylus macrobotrys*	0.5	0.004
Allophylus abyssinicus	0.62	0.0003	*Galineria saxifraga*	-0.49	0.006
Albizia grandibracteata	-0.61	0.0003	*Chionanthus mildbraedii*	-0.43	0.02
Apodytes dimidata	0.6	0.0004	*Dracaena fragrans*	-0.42	0.02
Coffea arabica	-0.68	0.0005	*Ficus vallis-choudae*	-0.38	0.03
Bridelia micrantha	-0.57	0.001	*Oxyanthus speciosus*	-0.37	0.03
Celts toka	-0.55	0.001	*Albizia grandbracteata*	-0.39	0.03
Filicium decipiens	-0.56	0.001			
Alangium chinense	-0.62	0.002	**Correlated with patch shape**		
Antiaris toxicaria	-0.44	0.002	*Trichilia dregeana*	0.66	0.0008
Sarcocephalus latifolius	-0.5	0.004	*Trichlia emitica*	-0.57	0.001
Croton macrostachyus	0.44	0.01	*Vanguria apiculata*	-0.43	0.001
Ficus vallis-choudae	-0.46	0.01	*Albizia grandbracteata*	-0.55	0.002
Garcinia buchananii	-0.43	0.01	*Cassipurea malosana*	0.5	0.004
Dracaena afromomum	0.44	0.02	*Ficus vallis-choudae*	-0.48	0.007
Olea welwitschii	-0.42	0.02	*Olea capensis*	0.46	0.009
Allophylus macrobotrys	-0.39	0.03	*Macaranga capensis*	0.45	0.01
Argomuellera macrophylla	-0.4	0.03	*Diospyros abyssinica*	-0.44	0.01
Ficus mucuso	-0.39	0.03	*Entada abyssinica*	0.39	0.03
			Afrocarpus falcatus	0.39	0.03
Correlation with disturbance			*Pappea capensis*	-0.4	0.03
Olea capensis	-0.56	0.0007	*Allophylus abyssinicus*	0.38	0.04
Elaeodendron buchananii	-0.55	0.001	*Pouteria adolfi-friederici*	0.39	0.04
Apodytes dimidata	-0.53	0.003	*Eugenia bukobensis*	-0.37	0.04
Trichilia dregeana	-0.58	0.007			
Albizia gummifera	0.45	0.01			
Cordia africana	0.45	0.01			
Anthocleista schweinfurthii	0.35	0.03			

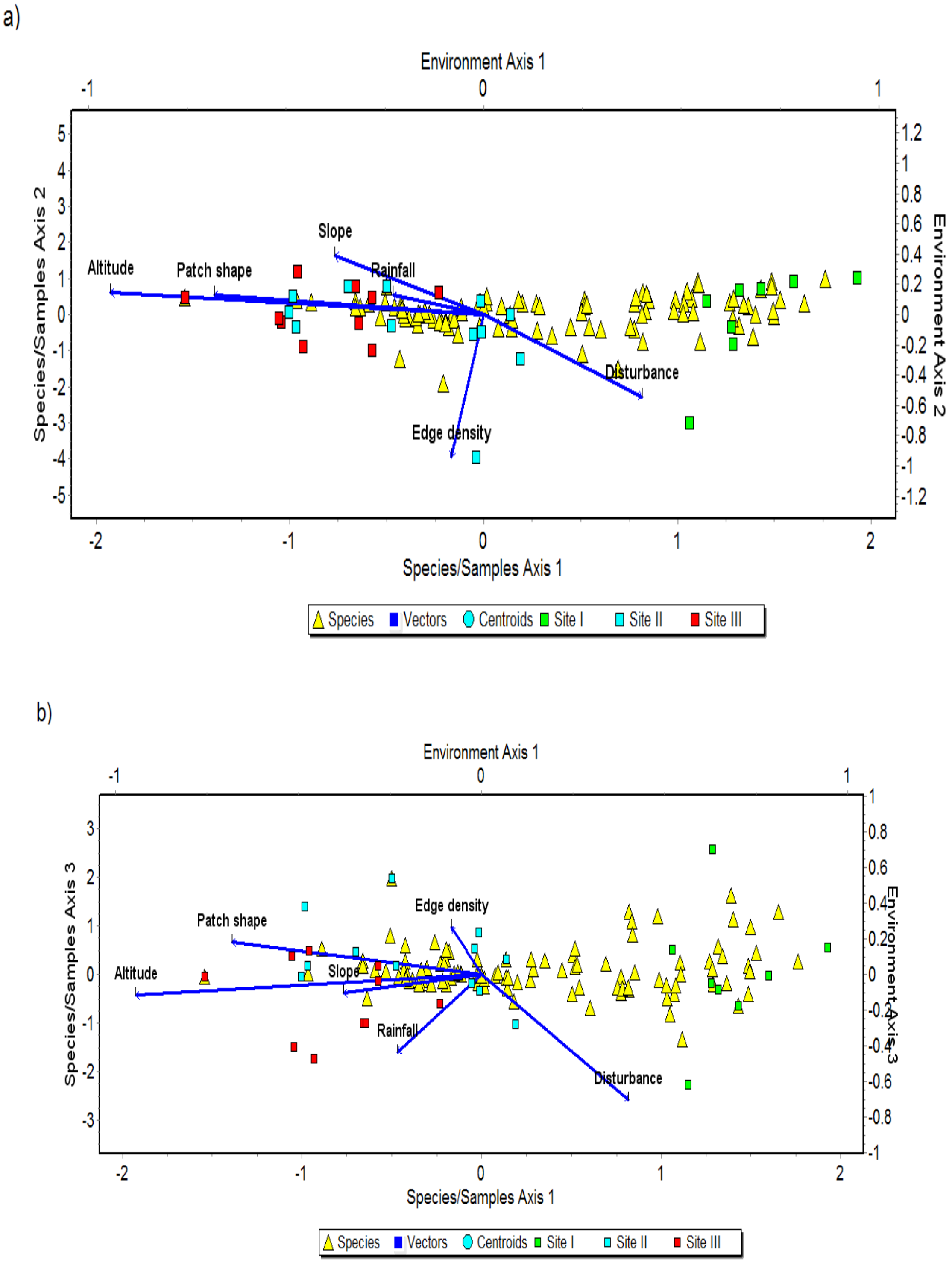


Figure 4.7 CCA ordination biplots of species and forest patches with vectors for environmental variables showing (a) axes 1 and 2; (b) axes 1 and 3. Location of patches in sites I, II and III of the studied landscape in the Illubabor Administrative Zone of south-west Ethiopia are distinguished. The woody species with frequency distribution > five patches or 40 sub-plots were used for the ordination.

4.4.7 Effect of fragmentation, disturbance and environmental gradients on woody plant species richness, diversity and abundance

Patch area and shape had negligible influence on woody plant species richness (r = -0.12, p = 0.58; r = 0.003, p = 0.98 respectively), basal area (r = 0.26, P = 0.21; r = 0.05, p = 0.81) and individual tree density (r = 0.10, p = 0.63; r = 0.25, p = 0.24). Likewise altitude, slope and rainfall had no influence on basal area (r = -0.14, p = 0.5, r = -0.02, p = 0.9), r = 0.05, p = 0.8 respectively), density (r = 0.05, p = 0.7, r = 0.19, p = 0.3, r = 0.22, p = 029 respectively) and species richness (r = -0.14, p = 0.5, r = -0.04, p = 0.8, r = -0.20, p = 0.3 respectively). However, the relationship between patch shape and individual tree density was positive even though it was on the margin of significance (r = 0.25, p = 0.05). Patch area and individual tree density were not significantly correlated (r = 0.10, p = 0.63). Individual tree density did show a clear decrease with edge density of patches (r = -0.498, p = 0.01). Disturbance intensity within patches (the sum of scores for cut trees, forest clearance, firewood collection and grazing) was significantly negatively correlated with tree basal area (r = -0.40, p = 0.04) and tree height (r = -0.55, p = 0.005). Basal area and height of trees were not correlated with any of the other environmental variables, i.e. rainfall, slope, altitude, patch area or patch shape.

There is no significant correlation between Fisher’s alpha diversity of woody species and fragmentation metrics: fractal dimension (r = - 0.0002, p = 0.99), patch size (r = -0.15, p = 0.46), patch shape (r = -0.28, p = 0.15) and other environmental variables: rainfall (r = -0.25, p = 0.23) or slope angle (r = -0.35, p = 0.09). However, Fisher’s alpha diversity did increase significantly with decreasing altitude (r = -0.58, p = 002) and with increasing edge density (r = 0.45, p = 0.02).

4.5 Discussion

4.5.1 Overall species richness and diversity

Tropical forests are considered to be one of the most species-diverse ecosystems in the world (Kumar et al., 2006). However, fragmentation of these forests due to deforestation is occurring faster than their recovery even if disturbance is ceased. Consequently, in many tropical regions forests are restricted to small and isolated patches (Turner and Corlett, 1996). Where large forest areas persist they can still support a high proportion of the original forest biodiversity, including the larger biota, however recent research has shown that a substantial number of forest species can also persist for decades in fragmented forests (Lindenmayer and Fischer, 2006). Forest patches can also play an important role in resilience at the landscape scale: in regions with little forest remaining, as in the case of Ethiopia, patches can be the source from which natural vegetation re-establishes (Turner and Corlett, 1996).

In Ethiopia, deforestation and forest degradation have a long history and only a few fragmented afromontane forests have remained intact in the south-west and south-east part of the country. Recent studies of these forests reported that they contain a high diversity of woody species (Friis, 1992; Abayneh, 1998; Kumelachew and Tamrat, 2002; Tadesse, 2003; Schmitt, 2006). Although it is difficult to do direct comparison due to variation in forest size, sampling method, number of sampled plots and objective of the study (Tadesse, 2003), it is worth mentioning (for comparison) some results of previous studies in large reserved forests in south-west Ethiopia in the vicinity of the current study area within similar landscapes and environments. Hence, in Masha-Anderecha afromontane forest located in Shekicho Zone 107 species of woody plants were recorded in a total of 96 plots (Kumelachew and Bekele, 2003), while in Bonga forest, in Kaffa Zone 158 species were recorded in 85 plots (Schmitt, 2006); in Illubabor Zone itself in Yayu forest 92 species were recorded in 50 plots (Tadesse, 2003) and in Sese forest 133 species were recorded in 75 plots (Shiferaw, 2010). In the present study, I recorded a much higher richness than each of these previous studies, though with a larger sample plot number: 254 woody species, distributed among 179 genera and 62 families, in 22 fragmented patches and 8 different sites of the same large continuous reserve forests in 240 plots (Appendix 4.6). Recorded species richness is clearly a product of the diversity of habitat conditions which it contains. Within the present study, since the total sample sizes were different between the three sampled sites, it is difficult to compare them as a means of assessing the impact of past and contemporary forest management or land-use intensity on species richness. Differences in overall recorded species richness between them could be caused either by an actual difference in landscape species richness or just an artificial sample difference because of unequal sample size. Therefore, in addition to comparing total species richness and average species density per plot, the species accumulation curves were used to compare species richness of the three sites rather than absolute number of species. The individual accumulation curves for each of the three sites shown in Figure 4.3b allow comparison of species richness at a number of different same sample sizes. Therefore, the individual accumulation curve showed the higher species richness of Site I as compared to Site II and III

The majority of population of species recorded were found in the (larger number of sampled) smaller patches in sites II and III rather than being confined to the larger forest areas in Site I, suggesting that the small patches at higher altitude are extremely valuable for maintaining regional plant biodiversity. The forest patches contained economically important species such as *coffea arabica*, *Aframomum korarima* and *Piper capense*, which also contribute delivery of provisioning services to the value of their biodiversity. The patches also contained a large number of forest-habitat specialist species of conservation interest such as *Chionanthus mildbraedii*, *Galineria saxfraga*, *Oxyanthus speciosus* and *Dracaena fragrans*, species designated as being of national conservation priority, e.g. *Cordia africana* and *Afrocarpus falcatus*, and a high population density of *Prunus africana*, a species listed as vulnerable in the IUCN Red Book (IUCN, 2010; Tadesse, 2003).

4.5.2 Species type and structural composition

Change in forest structure and composition is a typical effect of human influences (Thompson et al., 2009). These human influences are more conspicuous in small fragmented forest patches than unfragmented forests (Hill and Curran, 2005) since the unfragmented forests are relatively resilient (Thompson et al., 2009; Malhi et al., 2009). The current study revealed that human-induced fragmentation and within-habitat disturbance shaped the woody plant community structure and composition of the forest patches. Here, I sampled patches in three contrasting sites differing in the dependency of local livelihoods on forest resources and found a significant difference in their species composition (Table 4.8). In Site I pioneer species, e.g. *Albizia gummifera*, *Milletia ferruginea*, *Croton macrostachyus* and *Maesa lanceolata*, accounted for most of the stem density and basal area (Appendix 4.1). These forests are under continuous human disturbance associated with coffee management, which benefits pioneer species as evidenced by their abundance in human-created canopy gaps. The species composition was more or less similar between sites II and III, where the forest area predominantly comprised small fragmented forest patches: a mixture of wide-habitat-range/generalist (e.g. *Prunus africana*, *Pouteria adolfi-friederici*, *Syzygium guineense*), forest-habitat (e.g. *Macaranga capensis*, pioneer (e.g. *Croton macrostachyus*) and many forest understory (e.g. *Chionanthus mildbraedii*, *Vepris dainellii*, *Galineria saxfraga* and *Oxyanthus specieous*) species (Appendix 4.2 & 4.3). These differences in species composition among the three sites can be attributed to altitudinal variation, within-habitat disturbance from timber extraction and firewood collection, forest clearance for cultivation, and forest thinning for coffee management, as well as fragmentation. The sampling design restricts the potential to separate the influence of these factors on variation amongst sites because they are confounded, however (using variation amongst patches within sites) their correlation with species composition is analysed in chapter 5.

In contrast to the differences in species composition between the three sites the mean stem density and basal area of trees were very similar between them with no significant differences (Figure 4.4). In all three sites, when shown in 5 cm intervals (but not in 10 cm ones) the density of trees in the smallest DBH size class is less than the second class

(Figure 4.5). This low density in the smallest size class of trees in all three study sites is in agreement with other studies carried out in unfragmented forests of Ethiopia (Kumelachew and Tamirat, 2002; Tadesse, 2003). Observations of cut stems in the sample plots indicate that this may be due to selective felling of trees in the 5-10 cm DBH size range. This may pose a threat to the rate of establishment of younger trees to replace older canopy trees when they are harvested or die, which may eventually lead to an opening of the forest canopy. Cut stumps resulting from illegal timber extraction of large diameter trees were observed in all three sites, contributing to the low density of large trees retained as coffee shade.

4.5.3 Fragmentation versus environmental effects on species composition

Many studies have shown that the area of habitat patches has an effect on plant species richness and diversity (e.g. Hill and Curran, 2001; Godefroid and Koeda, 2003; Cagnolo et. al., 2006; Fischer and Lindenmayer, 2007). In contrast, in the present study, regression analyses indicated that patch area (as an indicator of fragmentation) was a poor predictor of woody plant species richness and diversity. This was also the case in some previous studies in the tropics, e.g. Cayuela et al. (2006a) found no effect of fragmentation per se (defined as the reduction in patch area and connectivity) on plant species richness and diversity. In his review of many empirical studies, Fahrig (2003) pointed out that the effects of fragmentation on species richness and diversity were ambiguous and as likely to be positive as negative. Cayuela et al. (2006c) mentioned that these inconsistent results are based on studies that differ in spatial and temporal scale, and targeted species, with different response variable such as presence/absence, abundance and alternative diversity indices, in sites differing in environmental variability and history of human use. In my study, patch area was correlated with tree community physical structure indicated by tree basal area and density. Other studies have also found a significant positive correlation between patch size and tree stem basal area and density (e.g. Tripathi et al., 2010).

Patch shape was expected to be another major variable influencing the number and composition of species in habitat patches (Fahrig, 2003) but, supporting other findings (Hill and Curran, 2003), I did not find significant effects of this variable on species richness and diversity. This result, however, should be considered carefully, since the sampling location within a patch and size of the patch can determine the influence of patch shape on species composition (Graham and Blake, 2001). In this study, I sampled the interior of all patches and compared the influence of shape between patches also differing in size. The influence of patch shape may be more significant where the sampling is also done near the edge of patches since patch shape determines the ratio of edge to interior habitat (Graham and Blake, 2001); results of this are reported in chapter 5.

The shape of patches could be more influential over species composition than species richness and diversity. For example, Hill and Curran (2003, 2005) found that irregular-shaped patches have different species composition than do more regular (circular) patches. Irregular patches are mostly dominated by light-demanding pioneer species and animal-dispersed species (Hill and Curran, 2003). In the present study, Pearson correlation analyses showed that the presence of forest-margin or light-demanding species was associated with irregular patch shape indicating that it can influence species composition, even in the sampled forest interior habitat (Table 4.11). However, as with all of these correlation results, the confounding of the variables (in this case between irregular patch shape and altitude (Sites II and III), must be taken into account.

Edge density of habitat patches has also been found to explain the distribution, abundance and richness of species (McGarigal and Marks, 1995). Many studies indicated that forest fragmentation results in the increased susceptibility of forest remnants to the edge effect which may result in change of species composition (Kupfer et al., 2006). In my study, as for patch shape, the abundance of forest-margin or light demanding species in the sampled interior forest of small patches was correlated with patch edge density. The lack of this effect for species associated with larger patches such as *Chionanthus mildbraedii*, *Oxyanthus speciosus* and *Vepris daniellii* may be accounted for by the purposeful location of the sample plots in the patch interior habitat away from the edge. The results from sample plots located in both edge and interior habitat presented in chapter 5 are more suited to testing edge effects.

Analysis of environmental and fragmentation metrics was made in order to determine the factors contributing to the variation of woody species composition. Amongst the three sites this variation appeared to reflect the significant influence of altitude more than indicators of fragmentation such as patch area and shape in the Canonical Correspondence Analyses (CCA) of species and plots (Figure 4.7). Altitude corresponded with the transition from mixed transitional evergreen forest in Site I to afromontane forest of sites II and III as indicated by the presence of a mixture of species from lowland and highland afromontane forests in Site I (Friis, 1992). These species include *Vanguria apiculata, Pouteria altissma, Manilkaria butugi, Deinbollia kilimandscharica, Morus mesozygia and Hallea rubrostipulata* with a lowland affinity and *Allophylus abyssibicus, Milletia ferrugeana Bersama abyssinica, Maesa lanceolata* and *Croton macrostachyus* with a broader range but a particular affinity with highland forests. These transitional evergreen forests are typically located within an altitudinal range of 1300-1900 m a.s.l. (Friis, 1992) – the altitude of the forests studied in Site I was 1594-1695 m. Some species are restricted in their distribution to within this altitudinal range, for instance the abundance of *Vanguria apiculata, Pouteria altissma, Manilkaria butugi, Deinbollia kilimandscharica* is significantly higher in Site I located below 1900 m.a.s.l. As well as the broad-range species that were also abundant in Site I (listed above), the tree community of sites II and III was characterised by the abundance of species much rarer or even absent from lowland forests, e.g. *Syzygium guineense, Pouteria adolfi-friederic, Afrocarpus falcatus, Elaeodendron buchananii* and *Prunus africana*. These results support the distribution range of species recorded in the flora of Ethiopia (Hedberg and Edwards, 1989-2006; Edwards et al., 1995-2000).

4.5.4 Within-habitat disturbance

Although fragmentation of forest area is currently the most widely recognised cause of biodiversity loss within landscape ecology, it is important not to neglect the potentially critical role of human disturbance within remnant forest patches. Disturbance from selective tree logging, cutting of understorey tree stems, firewood cutting, and browsing by livestock were all observed to be factors impacting the studied fragmented patches with potential implications for species richness, composition and structure (as reported by Gao et al., 2009, Cayuela et al., 2006d). In the present study, plot species richness was negatively correlated with an index combining the intensity of these different types of disturbance. Disturbance of canopy trees, e.g. through felling for timber, in general creates gaps providing a rich mosaic of microhabitats capable of supporting a diversity of plant species (Denslow, 1987; Schnitzer and Carson, 2001; Wright, 2002). On the other hand disturbance can negatively affect populations of species dependent on intact forest habitats (Teketay, 1992). In the present study, I observed that disturbance affected understory trees. These trees such as *Chionanthus mildbraedii., Vepris daniellii., Macaranga capensis* and *Olea capensis subsp. macrocarpa*, were gradually thinned, and in some areas totally removed, and this resulted in increasing convergence of the composition of tree stands with dominance by the same unexploited and disturbance-benefiting species of *Albizia gummifera, Milletia ferruginea, Croton macrostachyus* and *Maesa lanceolata* in Site I.

The analyses reported in chapter 3, supported by information from local people, showed that for three decades, especially after the government's planned resettlement program in 1984, the forests in Illubabor Zone have experienced varying degrees of disturbance, such as human-induced fragmentation through clear-cutting of forests for settlement and crop agriculture (including in some places tea plantations) and introduction of intensive management of natural forest converting it to forest managed for coffee production, utilization of the forests for grazing and production of timber and non-timber products (Tadesse et al., 2001; Mengistu, 2005; Fayera and Dinch, 2006). Forests are cut down in order to convert it to cropland, then after a few cycles of crops the land is allowed to or actively, converted to grazing land. Grazing is also allowed in forests to such an intensity that there is very few tree regeneration in some sub plots; combined with other ongoing disturbance this leads to a gradual transition from forest to grazing land. Their current natural status is greatly diminished also by gradual extraction of large canopy and understory tree species (Tadesse, 2003). In the present study, almost all the sampled forest patches in Site I had undergone thinning of large- and medium-sized trees. In these forests only some canopy species were likely to be preserved including some legume species such as *Albizia gummifera*, *Milletia ferrugina* and *Albizia schinperiana* with high recorded importance value indices in Site I (Appendix 4.3), which are considered to be important for promoting soil fertility (per. com. from local community elders). While this preservation of some tree species by farmers for coffee shade and soil fertility has potential benefits for biodiversity and the provision of ecosystem services, it should not be taken as achieving the conservation of the full biodiversity of these afromontane forests, which is clearly being threatened by gradual fragmentation, selective timber extracted and establishment of homogenous stands of low tree species diversity. Due to disturbance, the interiors of forests managed for coffee production in Site I, as well as some of the fragmented forest patches in sites II and III, were dominated by pioneer tree species (e.g. *Milletia ferruginea, Macaranga capense, Croton macrostachyus* and *Maesa lanceolata*) as well as a range of vine species (e.g. *Gouania longispicata*, *Sericostachys scandens*, *Urerea hypselodendron, Stephania abyssinica, Dioscorea bulbifera*, *Peponium vogelii, Scutia myrtina, Lagenaria abyssinica* and *Combretum paniculatum*) that are able to invade forest gaps. Shrubs that are typical of more open woodland or other disturbed woody habitats (e.g. *Vernonia spp*., *Hibisus spp.*, *Cyathula sp*. and *Rubus steudneri*) were also notably more abundant in these forests. An empirical study using an individual tree-based gap model of neotropical montane forest following human disturbance found that the recovery rate is very low and it may take several hundred years to re-establish a canopy dominated by shade-tolerant tree species (Golicher and Newton 2007). In fragmented forest subjected to recurrent disturbance and gradual thinning, as in the case of the afromontane forest in the present study, there is therefore a risk of species becoming locally extinct if the rate of disturbance exceeds the rate of forest habitat and population recovery.

4.5.5 Implication for biodiversity conservation

The high rate of loss of forest area across Illubabor Zone was established in chapter 3. The present chapter reports the process and impacts of fragmentation and degradation of the remaining forests. It also reports the high woody plant diversity and conservation importance of the remaining forest fragments. In a landscape ecology context, conservation of the remaining forest patches would also allow them to act as stepping stones through inter-patch movement of plant propagules (seeds and pollen) that increase landscape connectivity (Lindenmayer and Fischer, 2006; Arroyo-Rodriguez et al., 2008). This will be important to enable the landscape-scale survival of metapopulations of rare and endangered species, even if individual patches cannot support a viable population in isolation (Hanski, 1991). In a functionally connected landscape the total area of forest in remaining patches will be critical for the conservation of biodiversity (Arroyo-Rodriguez et al., 2008). Diversity in environmental conditions amongst forest patches (providing complementary habitats/resource) will also be important for maximising the beta component of landscape-scale biodiversity (Dunning et al., 1992; Arroyo-Rodriguez et al., 2008).

The results of the present study indicate that forest patches distributed in a landscape with an agricultural-dominated matrix contained many species of high conservation importance, and were (on balance) more important for biodiversity conservation than the limited number of large preserved forest in this part of south-west Ethiopia. In addition to their value for biodiversity conservation and ecological processes, the small forest patches distributed across the landscape of Illubabor Zone are also important in providing food provisioning ecosystem services as they are the major sources of wild spices, such as *Aframomun corrorima* and *Piper capense*, and a major cash crop, *Coffea arabica*, on which people depend as an important component of their livelihoods.

As the forest patches in Illubabor Zone continue to shrink in size and become increasingly degraded due to surrounding farm encroachment and intensive disturbance within patches, their capacity to provide sufficient habitat for viable metapopulations of many species (Arroyo-Rodriguez et al., 2008) and their capacity to deliver ecosystem services to local people will decline. There may also be a delayed decline in conservation and economic value of these small patches because extinction may occur one or more generations after fragmentation (Tilman et al., 2002). Should further fragmentation (and intensification of agricultural land use in the matrix) increase the effective isolation of remaining forest patches this will be an important factor (combined with habitat degradation and loss of habitat area) limiting tree recruitment and establishment within the patches (Arroyo-Rodriguez et al., 2008).

In conclusion, to conserve forest plant biodiversity in Illubabor Zone, it is critical to address all of these issues: retaining patches, minimising their isolation, preventing loss of their area and minimising further habitat degradation. If resources permit an active restoration programme then it would be valuable to target recovery from degradation within existing patches and re-establishment of forest in small patches distributed across the landscape of Illubabor Zone. In addition to preserving and protecting large unfragmented forests, in highly deforested and fragmented regions, such as Ethiopia, biodiversity conservation will only be possible by preserving and restoring small remnants concurrently (Hill & Curran 2003; Arroyo-Rodr´ıguez & Mandujano 2006). Therefore, under current circumstances, rather than focus on the creation of one single biosphere reserve of forest that is already deteriorated in its natural condition, a wider landscape-scale planning approach is recommended to rescue plant biodiversity from local extinctions (da Silva & Tabarelli 2000, Arroyo-Rodriguez et al., 2008). The local communities in this study area are effectively protecting some forest patches, e.g. Dike, Koda, Simbir, Qotora 1 & 2, Jireen and Sibu 1, and if their management efforts are coordinated, they may be able to conserve forest patches at a landscape scale, which could have a greater impact on conservation of the biodiversity of afromontane forests of south-west Ethiopia.

CHAPTER 5

RESPONSE TO FRAGMENTATION OF PLANT SPECIES DIFFERING IN LIFE-FORM AND HABITAT REQUIREMENT

Abstract

Fragmentation alters both the extent and spatial configuration of habitat patches and reduces living space for species. It also reduces connectivity of patches and increases the length of edge between patches and the surrounding matrix ultimately increasing the edge effect on microclimate and physical disturbance at the boundaries of patches. The objective of this chapter is to determine the impact of fragmentation-linked edge effects on plants of different functional characteristics in patches of afromontane forest in south-west Ethiopia. The abundance of different plant functional types (for woody species based on their shade tolerance and for epiphyte and fern species based on their shade tolerance and morphological characteristics) was assessed and the net impact on plant community composition and structure was determined.

Woody (tree, shrub and vine) and non-woody (vascular epiphyte, geophytic fern and other angiosperm herbs) species were sampled in 18 patches within a matrix of intensive land use. All plants were sampled in 16 plots of 20 x 20 m per patch (8 in interior and 8 in edge habitats) – total sample area 11.52 ha. Total species richness in the interior habitat was 394 (113 herbs, 90 trees, 62 shrubs, 50 epiphytes, 46 vines and 30 ferns) and in the edge habitat 413 (110 herbs, 95 trees, 83 shrubs, 58 vines, 26 ferns and 28 epiphytes). Most of species were common in both habitats

Tree species composition varied significantly between edge and interior habitats. Light-demanding species were more abundant (with higher frequency amongst plots) in the edge habitat and forest-habitat/forest-specialist species in the interior. Disturbance and altitude accounted for more of the variability in species composition between the two habitat types than did patch area, shape or edge density. Stem density of trees did not differ significantly between edge and interior habitats but basal area was significantly

higher in the interior. The geophytic angiosperm herbaceous species community showed differences between edge and interior due to higher ruderal species in the edge habitat. The species richness of both vascular epiphytes and geophytic ferns was significantly higher in the interior habitat than the edge and the species composition differed. The habitat specificity and functional group of epiphyte and geophytic fern species, and available host tree size for epiphytes, had a significant influence on species distribution between edge and interior habitats. Epiphyte species with herbaceous leaves and stems were more abundant under deep shade in the forest interior, whereas species with fleshy and leathery leaves, pseudobulbs and succulent or woody stems were distributed more evenly between edge and interior habitats, though still having greater abundance in the interior.

5. 1 Introduction

Habitat fragmentation is a major characteristic of habitat loss caused by anthropogenic disturbance which may be an important mechanism in its effects on biodiversity. It has been a major focus of landscape ecological research worldwide (e.g. Turner et al., 2001; Echeverria, 2006; Ewers and Didham, 2006) and is considered to be a major factor in functional and structural landscape pattern change and biodiversity loss in the tropics (Hill and Curran, 2001). Fragmentation alters both the extent and spatial configuration of habitat patches and reduces living space for plant and animal species that require large and connected patches (Fahrig, 2003; Baldwin and Bradfield, 2007). It reduces functional connectivity by isolating remaining patches from one another (Forman, 1995) and increases the length of boundaries or edges between patches and the surrounding matrix (Forman, 1995). The boundaries created between patches and the surrounding matrices are likely to be subject to edge effects (Forman and Godron, 1986; Forman, 1995; Franklin, 2001; Lindenmayer and Fischer, 2006). These changes in the spatial arrangement of habitats could have negative effects on species richness and community composition within the remnant patches (Saunders et al., 1991; Forman, 1995; Laurance et al., 1998; Hill and Curran, 2001; Harper et al., 2005; Echeverria, 2007) and alter ecological processes such as movements of propagules, e.g. spores, pollen and seeds from parent individuals in the metapopulation (Hamilton, 1999; Turner et al., 2003).

The fragmentation of habitats, however, does not affect all species in the same way. Species that require large and continuous habitats often react negatively to the effect of fragmentation (Bennett and Saunders, 2010). The effects of fragmentation on such species can be a direct result of changes in habitat area (Lindenmayer and Franklin, 2002). For some species, impacts of habitat fragmentation are mainly due to changes in the spatial arrangement of suiTable habitat across the landscape and more frequent interaction with altered microclimate and other factors associated with human land use at the periphery of fragmented patches (Turner et al., 2001). This could indicate that the ecological consequences of forest fragmentation may depend on a combination of parameters in addition to area of the patches (Echeverria et al., 2006), such as adjacent land use (matrix quality), spatial configuration of the patch including shape, edge and connectivity (Forman, 1995; Lindenmayer and Franklin, 2002) and within-patch habitat quality (e.g. linked to disturbance type and intensity) (Lindenmayer and Fischer, 2006).

Most studies of fragmentation effects thus far are largely derived from the theory of island biogeography (MacArthur and Wilson, 1967; Lovei et al., 2006) which is based on the idea that the surrounding habitat, within which fragments are situated, is unsuiTable for fragment habitat-dependent species (Benedick et al., 2006). Island biogeography theory took into account only patch size and isolation or connectivity, but not variation in habitat quality, or other attributes of the patch or surrounding matrix (Farina, 2006). Thus, the theory has been used to explain the loss of species as a function of the loss of patch area resulting from land use/cover change (Godefroid and Koedam, 2003; Lovei et al., 2006). On the other hand, there is increasing evidence that habitat patches usually exist in a complex landscape matrix, and dynamics within a patch are also affected by matrix composition (Andren, 1994; Wiens, 1994; Hobbs, 2001; Cook et al., 2002; Lindenmayer and Franklin, 2002; Helm et al., 2006), change to their shape (usually represented as edge-to-area ratio), connectivity and within-habitat quality or disturbance (Lindenmayer and Fischer, 2006). Therefore, in addition to patch area, other patch characteristics such as matrix quality, geometry of the patch (shape, edge density, fractal dimension) and within-patch anthropogenic disturbance should be considered when assessing the impact of fragmentation on species richness, composition and diversity (Metzger, 2000; Fahriga, 2003; Godeforid and Koedam, 2003; Helm et al., 2006).

The quality of the matrix in terms of vegetation composition could play a major role in ecological process within patches and on which species are present (Lindenmayer and Fischer, 2006). It can contribute to the edge habitat dynamics, including population recovery of species after disturbance (Lamberson et al., 1994), the exchange of individuals between patches (Gaublomme et al., 2008) and occupancy rates of the patches (Bradford et al., 2003). The shape of fragmented patches represented by edge-to-area ratio is an indicator of the influence of edge effects within the patch (Hill and Curran, 2003). Changes in patch shape could increase the amount of edge habitat, resulting in loss of habitat area for species dependent on patch interior conditions. Edge habitats have distinct microclimates, with abrupt changes in the amount of solar radiation, soil nutrients and moisture, and generally have high microhabitat diversity (Murica, 1995). For many plant species these habitats are a physical arena within which population dynamics, ecological processes and evolution occur (Murphy and Lovett-Doust, 2004). As many studies indicate, persistence of plant species over the long term in a heavily anthropogenically-influenced landscape requires coping with these changes to their habitat. Many plant life-history characteristics, including growth form, shade-tolerance and clonal propagation, may have significance in this context (Murphy and Lovett-Doust, 2004). For example, the resprouting of some tree stumps after disturbance from tree cutting in the edge indicates that local populations may persist in the medium-term even though a patch has become unsuiTable for regeneration from seed.

Fragmentation can affect species in different ways and at different geographical scales (Forman, 1995). Some species are very sensitive to fragmentation, while others are more tolerant (Haila, 2002; Hill and Curran, 2005). This differential susceptibility to habitat fragmentation is expected to depend on species' functional characteristics related to plant growth form, habitat requirement and resistance/resilience to disturbance, but only rarely have measurements been made of the difference between plant functional types in their tolerance of habitat fragmentation. Some studies have indicated that habitat fragmentation most severely affects plant species with low frequency of occurrence and high habitat specificity (Hill and Curran, 2001). There is a hypothesis that shade-tolerant trees and shrubs, and moisture-dependent lower plants like ferns and epiphytes, could be more sensitive to fragmentation and its accompanying edge effects. On the other hand, many established forest interior trees, after exposure to the environment of edge habitat, show resilience to the effects through their vegetative regeneration capacity and seed traits. There is a major debate in plant ecology about the extent to which ineviTable functional trade-offs mean that species traits which confer a competitive advantage in within-forest habitat conditions, necessarily disadvantage these species in more disturbed conditions (e.g. Wright et al., 2004; Easdale et al., 2007; Easdale and Healey, 2009). This indicates that community composition at the patch edge following fragmentation may depend on the actual capacity to tolerate disturbance of the species present and the diversity of genotypes within their population (Mayfield et al., 2010) as well as the potential for invasion by more disturbance-benefiting species.

These landscape ecology issues are an important applied consideration because the use of well-evidenced and ecologically-realistic information of responses of species will improve our ability to make wise management decisions. Since the forest fragments in the study area, the Illubabor Zone of south-west Ethiopia, are exposed to different intensities of land use, they provide a good opportunity to test the response of plants of different functional types to forest habitat fragmentation. The objective of this study was to evaluate how the response of plant species, and thus community composition, to fragmentation of afromontane forest habitat in south-west Ethiopia is linked to species functional type. It is hypothesised that the species of different functional types respond to fragmentation in different ways, with shade-tolerant tree species and herbaceous species lacking drought-tolerant morphological traits being more sensitive.

The approach taken was, within a landscape where there is a rapid rate of fragmentation of forest habitat patches, to compare the population abundance of species of different functional types in patch edge and interior habitats, and to determine the impact of these species-level responses on overall community structure and composition. The analysis also assessed other factors known to affect species diversity and composition such as rainfall, topography and habitat disturbance. The focus was on woody plants and indicator groups of herbaceous plants (angiosperm epiphytes and geophytic ferns). Specific tests were made of the relationships between the abundance of plant species of different functional type (life form, morphology and shade tolerance) and patch attributes (area, shape and edge density) and landscape matrix variables (proportion of different land-use and cover types).

5.2 Materials and methods

5.2.1 Selection and stratification of study habitat patches

From the 30 forest habitat patches sampled in the whole project, 18 were selected for this particular study. They were selected to fulfill the following criteria: area greater than 10 ha as calculated from the 2007 land-use and land-cover map; embedded within a matrix of agricultural land; and predominantly composed of highland rainforest species. The habitat within these 18 patches was stratified into two zones of contrasting environmental conditions reflecting the impacts of fragmentation, the edge and interior. For the present study the boundary between edge and interior habitat was defined by a distance of 50 m from the margin of the patch inward.

5.2.2 Vegetation sampling

A single main plot of 60 x 100 m was established in a position within the interior habitat of each fragment subjectively chosen on the basis of representativeness in species composition. Each main plot was sub-divided into 15 sub-plots using a 20 x 20 m grid. Eight of these sub-plots, based on a systematic sampling regime (see Figure 2.10 for plot layout) were used for vegetation inventory and are hereafter referred to as “plots”. In the edge zone of each patch, eight independent 20 x 20 m sample plots were established in a systematic design with even spacing between plots around the edge of the patch, with the nearest edge of each plot being at approximately 10 m from the outer edge of the patch. To get equal distance between plots around the edge, the circumference of the patch was calculated using GIS tools and the total divided into eight (number of plots) to get equal distance between the plots. Then, a single coordinate point for each edge plot was taken from the land use map and entered into a GPS for navigation to each plot.

All woody live, standing dead and fallen individuals ≥ 5 cm diameter at breast height (DBH) in each 20 x 20 m sample plot were measured using diameter tapes, and identified to species (this was achieved for all live individuals and for a majority of those that were dead). Trees with large buttresses were measured above these protrusions. The DBH of all stems ≥ 5 cm DBH of trees with multiple stems were measured separately. To get an equivalent total diameter for the whole individual the basal area of each stem was calculated, summed and divided by π, square rooted and multiplied by two. The height of each individual tree was estimated using a clinometer. When it was difficult to sight the top of individual trees in dense forest, the height was estimated by observation. Density of shrubs (woody plants < 5 cm DBH with multiple stems arising at the base) and presence/absence of all woody or semi-woody vines were collected in each 20 x 20 m sample plot. In addition, five nested 5 x 5 m sub-subplots were established within each 20 x 20 m sample plot, one at each corner and one at the centre, to record saplings (individuals < 5 cm DBH and > 1 m in height) and seedlings (individuals ≤ 1 m in height) of tree species, which were identified and counted. For angiosperm herbaceous species presence/absence was recorded in sub-sub plots used for seedling and sapling.

Within each 20 x 20 m sample plot the species of each vascular epiphytic plant growing on tree stems ≥ 10 cm DBH, and the DBH of its host tree, were recorded. Similarly all individuals of geophytic fern were identified to species and recorded. Epiphytes and ferns growing in dense clumps (often from the same stem) were recorded as single individuals following the method of Barthlott et al. (2001). Identification to the species level was made where possible in the field using field guides such as Demissew et al. (2004) and the keys in the Flora or Ethiopia and Eritrea (Volume 1). For those epiphytes located on upper trunks and difficult to identify from the ground, binoculars were used and where necessary samples were collected by local tree climbers under my direction with the aid of the binoculars (Lowan and Atwood, 2002). Identification of non-orchid species was made using Flora of Ethiopia and Eritrea. The majority of unidentified morphospecies were those found only in a sterile condition. Voucher specimens were collected for all unidentified morphospecies and taken to the National Herbarium, Addis Ababa University, where they were identified by matching with determined herbarium specimens.

5.2.3 Functional groups

Species were classified according to life form, habitat requirement and morphology (Appendix 5.5 – 5.8). There were six life forms: tree, shrub, vine, geophytic angiosperm herb, vascular epiphyte and geophytic fern. This life form classification was based on combined information from personal field observation and the Flora of Ethiopia and Eritrea (V1-V7). Three categories of habitat requirement were recognised: under deep shade/forest, widely distributed amongst habitats and forest-margin. Tree species were classified into shade/forest-habitat, wide-habitat/generalist and forest-margin on the basis of the description of which habitats they are present in given in the Flora of Ethiopia and Eretria (1989-2009); Bekel et al., 1993; Fichtl and Admasu, 1994). Epiphyte species were classified into forest-habitat/forest specialist and wide-habitat species. They classification was also informed by information about habitat association given by Sebsebe et al. (2004) and in the Flora of East Africa (Vol.3). I combined information from these two sources to obtain the most parsimonious classification for each species. For geophytic fern species two different categories of habitat-association were used: deeply shaded/moist forest and forest-margin (Flora of Ethiopia Vol. 1; White et al., 2001; Roux, 2001; Roux, 2003; Sebsebe et al., 2004; Archer and Victor, 2005; Burrows, 1990). In addition, two separate classifications of epiphyte species was made according to their morphology: (a) two categories of leaf texture (leathery/succulent or (thin) herbaceous) and (b) two categories of stem type (woody/pseudobulbus or herbaceous stem) using information from the same literature sources cited above. Fern species were also classified morphologically according to (a) two categories of rhizome orientation (creeping or erect); (b) two categories of frond arrangement (spaced or tufted) (my own personal field observation; Burrows, 1990). Shrub and vine species were classified into forest-habitat and forest-margin/wide-habitat species. Both forest-margin and wide-habitat shrub and vine species were pooled together and considered as one functional group since it was difficult to separate the two groups into independent functional groups by the available literature.

5.2.4 Environmental data, fragmentation indices and landscape descriptors

The cumulative effect of disturbance from (a) grazing, (b) gathering of non-timber forest products and (c) forest clearance for cultivation in each 20 x 20 m sample plot was subjectively evaluated in five categories from minimum 0 to maximum 4. The disturbance score of each interior habitat 20 x 20 m plot in single patch was summed and divided by eight to get mean disturbance score for single patch. The disturbance score for the edge habitat of each patch was calculated in the same way. For each patch altitude was recorded using a GPS (Garmin 60) located at the centre of the patch during the field survey. Rainfall data for the years 1986-2009 were obtained for the 24 monitored meteorological stations in the study area from the Ethiopian Metrological Agency. From these mean annual rainfall was estimated for each patch using the methods described in chapter 2. The boundary of each patch was determined in a GIS-based land-use/land-cover map generated from a Spot 2007 image (as described in Chapter 3). Fragmentation indices (patch size, shape, fractal dimension and edge density) for each patch were then extracted from this map. To estimate land cover in the matrix surrounding each patch, the proportion of forest land, farm and settlement land, grazing land and wooded grassland) within 100 m distance outwards from the patch edge was calculated using Patch Analyst (Arc GIS 9.2 extension) (Elkie et al., 1999).

5.2.5 Data analyses

5.2.5.1 Overall species composition

Assessment of the total species composition recorded across the 18 patches in the edge and interior habitats was based on a total of 144 sample plots (5.76 ha) in each habitat. The species in each were classified by life form (tree, shrub, vine, geophytic angiosperm herb, vascular epiphyte and geophytic fern).

5.2.5.2 Species composition and diversity in edge and interior

Differences in patch composition (based on individual species) between edge and interior habitats were tested using the ANOSIM randomization test (Clarke 1993). This technique is based on a Bray-Curtis dissimilarity matrix. Hence, a vegetation data matrix using presence/absence, comprising species X plots was constructed separately for trees, shrubs, vines, geophytic angiosperm herbs, vascular epiphytes and geophytic ferns both for the interior and for the edge habitats. The Similarity Percentage Analysis (SIMPER) described by Clarke (1993) and available in CAP 4 (Community Analysis Package 4) (Seaby and Henderson, 2007) was used to identify discriminating species. It calculates the average Bray-Curtis dissimilarity between pairs of inter-group samples (i.e. all sample plots in edge habitat against all sample plots of interior habitats). The Bray-Curtis dissimilarity measure incorporates the average dissimilarity between sample plots in edge and interior habitats and expresses this in terms of the average contribution from each species (Clarke, 1993). Then, the average abundance of shrub, vine and herb calculated using SIMER analysis were used to test the difference between edge and interior for selected functional groups (forest-habitat and forest-margin/wide-habitat species).

5.2.5.3 Structural composition of woody species between edge and interior habitats

In each patch, individual tree density (number of individual trees/plot area) was calculated for trees, saplings and seedlings. Likewise, basal area (m^2/ha) was calculated for trees ≥ 5 cm DBH. The importance value of each species in edge and interior habitats was calculated from their relative basal area, relative density and relative frequency (amongst the 20 x 20 m sample plots). The significance of differences in individual tree density and basal area between edge and interior habitats was tested for all species combined using two-way ANOVA. The model for the ANOVA treated each patch as a unit of replication (thus the data for the eight 20 x 20 m sample plots in each habitat type were combined to a single value for each patch). The importance value of selected tree functional groups based on their habitat characteristics also tested between edge and interior.

5.2.5.4 The effect of fragmentation, disturbance and environmental variables

As a test of fragmentation effects (Kupfer and Malanson, 1993) the influence on species composition of patch area, shape, edge density, disturbance, altitude, rainfall, slope, aspect and proportion of land-uses and land-covers in the surrounding matrix within 100 m of each patch was assessed using Canonical Correspondence Analysis (CCA). Before applying CCA, a multicollinearity test was conducted and highly correlated variables (which were patch area, land-use and land-cover proportion and slope aspect derived from the digital elevation model) were discarded. To explore the significance of the association of the above independent variables with the dependent vegetation variables Pearson correlation analysis was performed. The dependent variables were interior-habitat species richness (woody species), edge-habitat species richness (woody species), interior-habitat individual density (trees ≥ 5 cm DBH), edge-habitat individual density (trees ≥ 5 cm DBH), interior-habitat tree basal area and edge-habitat tree basal area within each forest patch. All statistical analyses were performed using QED Statistics 1.0 (obtained from PISCES Conservation Ltd, UK). QED Statistics is a Windows program that offers all the standard statistical methods used in science (Seaby and Henderson, 2007). The correlation of some individual species to each environmental variables also tested using Pearson correlation.

5.2.5.5 Vascular epiphytes and geophytic ferns

The significance of differences in species richness and individual density of vascular epiphytes and geophytic ferns between patch-edge and patch-interior habitats was tested using ANOVA. As above, the model for the ANOVA treated each patch as a unit of replication (thus the data for the eight 20 x 20 m sample plots in each habitat type were combined to a single value for each patch).The diameter of each epiphyte’s host tree, the overall density of host trees and the density of epiphytes on these host trees were used as variables to test for differences between patch-edge and patch-interior habitats using same model of ANOVA. The relationship between host tree diameter and epiphyte density was tested using Pearson correlation separately for the patch-edge and patch-interior habitats in order to compare the results between edge and interior for each patch. The density (number of individuals of each species) of geophytic ferns was compared between the two habitats.

The species rank-abundance for both vascular epiphytes and geophytic ferns was performed using density of each species in all 144 plots. Variation amongst different vascular epiphyte and geophytic fern species in their density between patch edge and interior habitats was assessed in terms of their functional/ecological type (habitat association and morphology). The species for epiphytes were classified according to habitat association (forest-habitat, wide-habitat) and morphology (woody and pseudobulbous stem, herbaceous stem, leathery and succulent stem and herbaceous leaf). The species of fern were classified by habitat association (forest-habitat, forest-margin species) and morphology (creeping rhizome, erect rhizome, spaced frond, tufted frond). The density of individual plants of all the species of one type/functional group were lumped together into a single pool and a single score calculated in interior as well as edge habitat (i.e. the unit of replication for this test was individual species).

5.3. Results

5.3.1 Overall species composition

A total of 465 species belonging to 305 genera and 98 families were recorded in the sample plots across the 18 forest fragments. Geophytic angiosperm herbs were the dominant growth form (133 species, 28.6%), followed by trees (104 species, 22.4%), shrubs (94 species, 20.2%) vines (64 species, 13.8%), vascular epiphytes (50 species, 10.8%) and geophytic ferns (30 species, 6.5%). Of these, 30 species (6.5%) were endemic to Ethiopia (11 herbs, 10 shrubs, 4 trees, 3 vines and 1 epiphyte). Only three of the recorded species, *Eucalyptus camldulensis*, *Eucalyptus globulus* and *Cupressus lucitanica*, recorded in the edge of two fragments (Jireen, Bussa-Jireen), were non-native (for the list of all plant species, see Appendix 5.18).

5.3.2 Structural composition of trees of patch interior versus edge habitat

The interior habitats and edge habitats showed considerable variation amongst patches in their structural composition (density, basal area and population size class structure). Individual tree density ranged from 418.8 ha^{-1} to 878.1 ha^{-1} in the interior and from 350 ha^{-1} to 725 ha^{-1} in the edge habitats (Appendix 5.3, Figure 5.1). Therefore, the mean tree density did not differ significantly between interior (621.7 ± 127.0 ha^{-1}) and edge (559.4 ± 94.8 ha^{-1}) habitats ($t = 1.69$, $DF = 17$, $p > 0.05$).

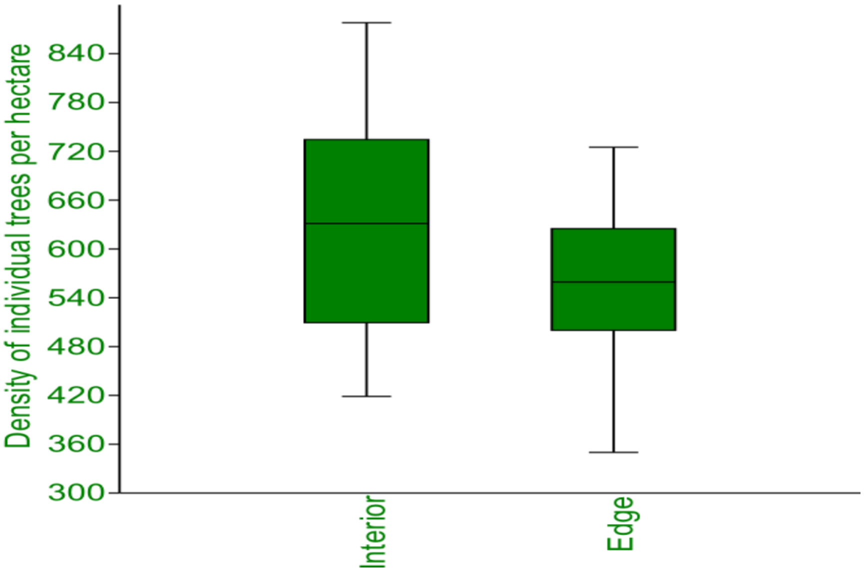


Figure 5.1 Box plot depicting individual tree density (trees ha^{-1}) in edge and interior habitats in 18 forest patches in the Illubabor Zone of south-west Ethiopia. The horizontal line through each box depicts the median; the box depicts the central 50th percentile of data and the ends of the whiskers show the smallest and largest values.

Tree basal area was significantly higher in patch interior habitat (96.8±9.4 m^{2} ha^{-1} with a range of 75.9- 113.1 m^{2} ha^{-1} amongst the patches) than edge habitat (77.2±15.3 m^{2} ha^{-1} with a range of 39.7-101.7 m^{2} ha^{-1}) (Figure 5.2, Appendix 5.3, $t=4.61$, $DF=17$, $p < 0.05$). Trees in the interior had slightly larger average DBH (15.16 cm) than those located in the edge habitat (14.85 cm).

There are 201 (3.4% of the total individual trees recorded) and 336 (4.8%)) big individual trees > 100 cm DBH in edge and interior habitat respectively. This indicates that large trees were selectively extracted from edge habitat, while a few original canopy trees of species such as *Syzygium guineense*, *Pouteria adolfi-friederici* and *Prunus africana* still persisted. The total basal area in interior and edge habitats was greatly influenced by these large trees in the plots. When individual species are considered, the highest proportion of basal area in the interior habitat was accounted for by widely-distributed species: *Syzygium guineense* (13%), *Pouteria adolfi-fredericii* (8.2%), *Prunus africana* (6.3%) and *Millettia ferruginea* (5.7%) and forest-margin species: *Albizia gummifera* (6.6%) (Appendix 5.2). The forest-margin species are more common in gaps created by disturbance.

The highest proportion of basal area for edge habitat was also accounted for by *Syzygium guineense* (11.7%), here followed by other widely-distributed species such as *Croton macrostachyus* (8.4%), and the forest-margin species *Maesa lanceolata* (6.9%) and the forest-habitat species *Macaranga capensis* (7.7%). In the edge habitat *Maesa lanceolata* was noTable for having a high density of trees rather than large individual trees.

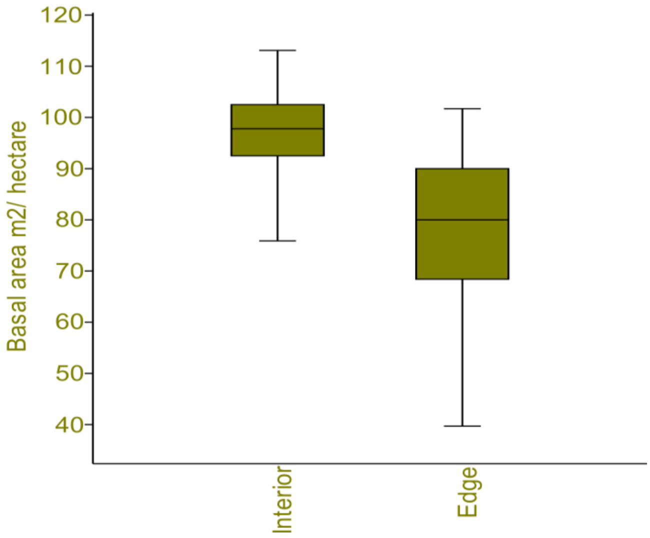


Figure 5.2 Box plot depicting stand basal area (m^2 ha^{-1}) in edge and interior habitats in 18 forest patches in the Illubabor Zone of south-west Ethiopia. The horizontal line through each box depicts the median; the box depicts the central 50th percentile of data and the ends of the whiskers show the smallest and largest values.

Overall abundance, as indicated by the Importance Value Index, showed a similar pattern amongst species to basal area (Appendix 5.1 & 5.2). In the patch-interior habitat the most abundant species were forest-margin and widely-distributed that dominated the canopy (Appendix 5.2). However, the next sets of species in abundance rank were in the forest-habitat specialist group and were typically found below the main canopy (Table 5.1). The most abundant tree species in the edge habitat (at ≥ 10% IVI threshold) were indicated under Appendix 5.1 and 5.2.

The trees of edge and of interior habitat of the 18 patches were grouped into 11 size classes with 10 cm intervals. The edge habitat is characterized by a small number of individuals in the first class (5-14.9 cm DBH) and a large number of individuals in the second class (15-24.9 cm DBH) (Figure 5.3a). Then, the second class decreased sharply to the third class and the third to fourth. After that there was a gradual decline of numbers towards the larger size classes (> 105 cm DBH). In contrast, the first size class (5-14.9 cm DBH) in the interior habitat is characterised by a large number of individuals (Figure 5.3b). There is a sharp decrease in numbers from the second class (15-24.9 cm DBH) to the third class (25-34.9 cm DBH). Then from the third class there is a gradual decrease toward the larger classer. It is only > 105 cm DBH that the interior habitat has a higher density of individuals than the edge, which is the sole important cause of the higher basal area in the interior habitat than the edge.

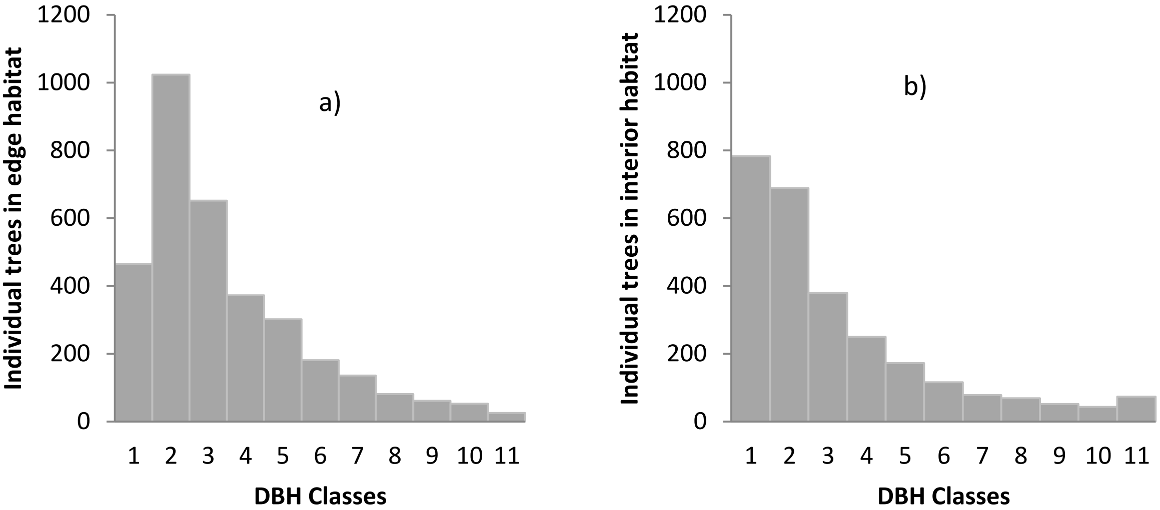


Figure 5.3 Individual size class distributions of all tree species in (a) edge and (b) interior habitat of 18 forest patches (5.76 ha interior and 5.76 ha edge). Absolute density of stems in each size class is shown for all individuals in the sample plots. Size class intervals are 10 cm: 1 = 5-14.9, 2 = 15-24.9, 3 = 25-34.9, 4 = 35-44.9, 5 = 45-54.9, 6 = 55-64.9, 7 = 65-74.9, 8 = 75-84.9, 9 = 85-94.9, 10 = 95-104.9 and 11 = > 105 cm.

Canopy height and stand vertical structure varied between interior (Figure 5.4a) and edge (Figure 5.4b) habitats. The number of individual trees > 22 m height is > two times larger in interior than edge (Figure 5.4b). The average maximum tree height per patch was also significantly higher in the interior (15.01 ±1.4 m) than the edge (11.2 ±1.5 m) habitat (F = 9.72, df = 17, $p < 0.005$). Across all patches absolute maximum height in the interior habitats was 45 m, whereas in the edge it was 25 m.

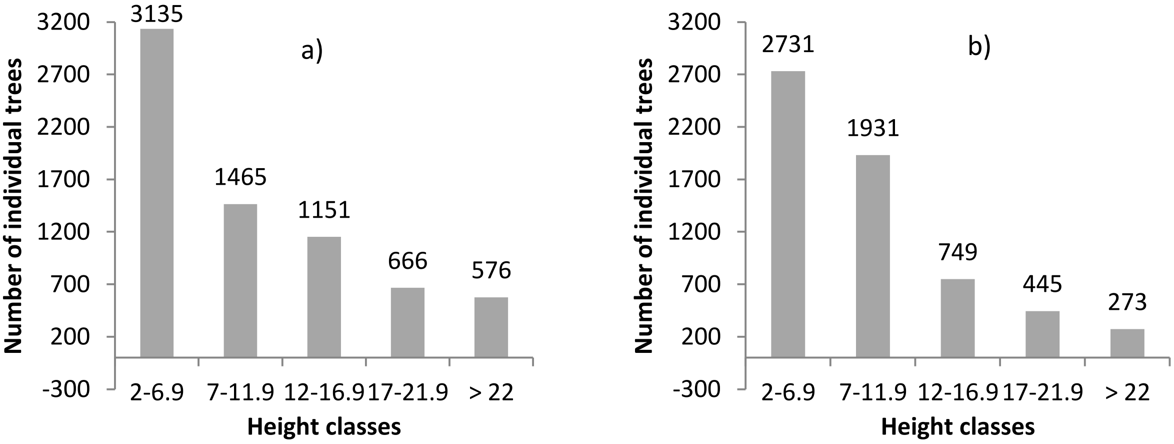


Figure 5.4 Height classes (in m) of tree species in (a) interior and (b) edge habitat of 18 forest patches (5.76 ha interior and 5.76 ha edge). Numbers shown above bars are the number of individual trees per class.

5.3.3 Species richness

The range of edge and interior habitat tree species richness amongst the 18 patches was 18-36 and 18-45 for interior and edge respectively (Appendix 5.3). The mean (±1.96*SE) species richness for the interior habitat is 28.0±2.4 and edge 31.4±3.3; the difference was on the borderline of significance (t = 2.119, DF = 17, P = 0.05,Figure, 5.5a). Likewise, the range of total woody species richness (sum of tree, shrub and vine species) was 39-65 for the interior habitat and 47-90 for the edge. The mean was significantly lower for the interior (52±3.5) than for the edge (69.3±5.9) (t= -5.4, DF=17, $p < 0.05$, Figure 5.5b)

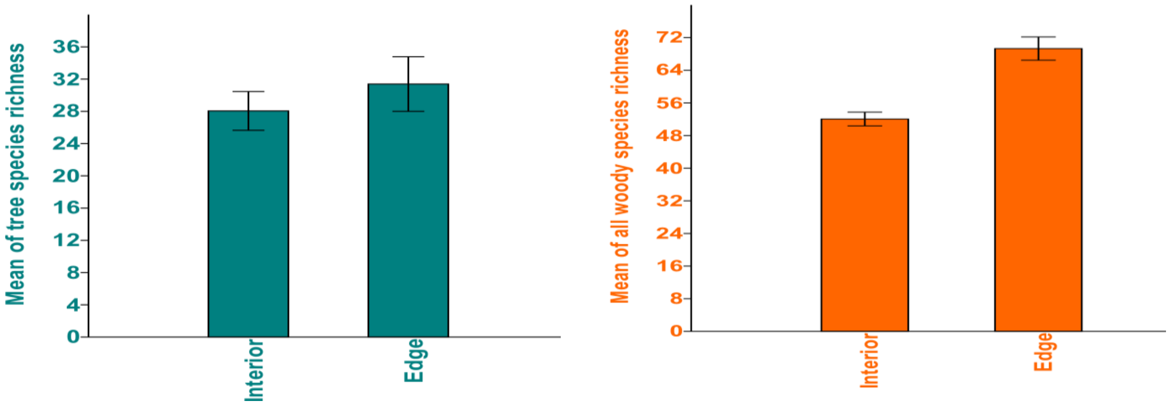


Figure 5.5 Mean (± one confidence interval, 1.96*SE) species richness of (a) tree species and (b) all woody species in interior and edge habitat of 18 patches.

5.3.4 Woody and herbaceous species composition similarity between edge and interior habitats

The species composition of trees, shrubs, vines and geophytic angiosperm herbs were each significantly different between patch edge and interior habitats (Table 5.1).

Table 5.1 Non-parametric analyses of similarity (ANOSIM) for differences in the rank similarities of species for two habitat types (edge and interior) grouped by species presence/absence and composition for trees, shrubs, vines and herbs. ANOSIMs were calculated based on Bray–Curtis similarity.

Pairwise comparison interior X edge	No. of permutations	R	P Value	Significance level (1 %)
Tree	1000	0.41	0.001	0.01
Shrub	1000	0.18	0.001	0.01
Vines	1000	0.24	0.001	0.01
Heb	1000	0.42	0.001	0.01

The habitat associations of the individual species are shown in Appendix 5.8 and 5.9 for woody species and herbs respectively. In addition, the species with importance value (IVI) ≥ 1.5% in either habitat (edge or interior) are shown in Table 5.2 and the difference in abundance between these two habitats for each of the three species functional (distribution) groups was tested using ANOVA. The tests indicate that tree species in the forest-habitat specialist group had higher abundance in the interior than the edge habitat (ANOVA, F= 10.22, P < 0.05, Figure 5.6a) whereas the forest-margin specialist group species were more abundant in edge habitat (F= 9.17, p < 0.05) (Figure 5.6b) but the species in the wide-distribution group showed no difference between the two habitats (F=0.27, P > 0.05, Figure 5.6c).

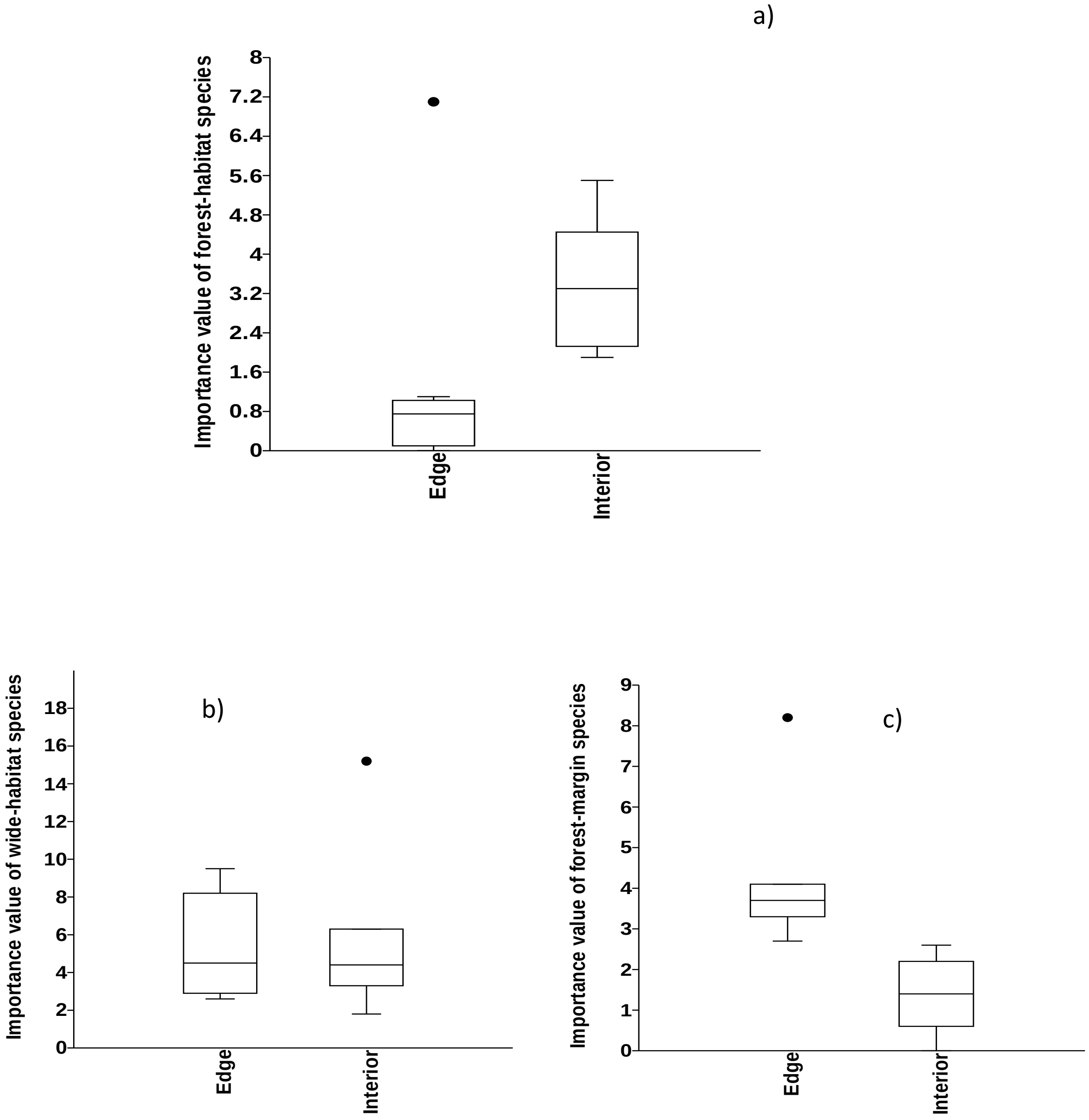


Figure 5.6 The abundance (IVI)) of tree species (restricted to those with an IVI > 1.5%) of three different distribution functional groups in edge and interior habitats: (a) forest-specialist species (b) wide-distribution species and (c) forest-margin species.

Table 5.2 Three components and total (IVI) abundance in edge and interior habitat of all tree species with an importance value > 1.5% in either habitat, or both.

Functional (distribution) groups	Relative dominance		Relative density		Relative frequency		IVI	
Forest-habitat species	Edge	Interior	Edge	Interior	Edge	Interior	Edge	Interior
Macaranga capense	7.7	6.6	7.9	5.4	5.7	4.5	7.1	5.5
Vepris dainelli	0.3	5.3	1.4	3.6	0.6	2.9	0.8	4.0
Chionanthus mildbraedii	0.2	5.1	0.8	3.7	1.1	5.1	0.7	4.6
Oxyanthus speciosus	0	3.1	0	0.6	0	3.6	0	2.5
Canthium oligocaprpa	0.8	2.5	1.1	1.9	1.4	1.4	1.1	1.9
Cassipurea malosana	0.5	2.0	0.3	1.8	0.5	2.3	0.4	2.0
Dracaena fragrans	0	1.1	0	5.8	0	3.8	0	3.6
Galinera saxifraga	0.4	1.0	1.0	3.6	1.0	4.4	0.8	3.0
Mean (±SE)							1.4±0.8	3.4±0.5
Wide-distribution species								
Syzygium guineense	11.7	21.6	10.5	16.7	6.3	7.4	9.5	15.2
Croton macrostachyus	8.4	10.9	9.5	4.4	6.9	3.7	8.2	6.3
Bersama abyssinica	2.1	5.9	5.9	1.8	5.4	2.1	4.5	3.3
Prunus africana	4.8	1.3	1.8	5	2.5	3.6	3.0	3.3
Allophylus abyssinicus	2.1	4.2	2.5	5.5	3.2	3.4	2.6	4.4
Pouteria adolfi-friederici	4.5	3.3	2.3	0.9	2.1	1.3	2.9	1.8
Mean (±SE)							5.1±1.2	5.7±1.9
Forest-margin species								
Maesa lanceolata	6.9	0.9	11.7	0.6	6.0	0.2	8.2	0.6
Rhus quartinana	2.3	0	3.6	0	3.2	0	2.7	0
Nuxia congesta	4.2	0.4	3.6	1.7	3.2	1.9	3.7	1.4
Albizia gummifera	4.2	2.1	2.6	2.6	3.1	3.1	3.3	2.6
Milletia ferrugina	4.2	2.5	4.4	1.3	3.7	2.8	4.1	2.2
Albizia schimperiana	3.4	0.4	2.4	0.8	4.3	1.5	3.4	0.9
Mean (±SE)							4.2±0.03	1.3±0.02

As stated above ANOSIM also showed a significant difference in shrub species composition between edge and interior habitats (Table 5.1). The habitat association of shrub, vines and herbs calculated by SIMPER (Clark, 1993) and species with ≥ 0.5 average abundance value in either habitat were also tested using ANOVA. The composition of shrub species in the forest-margin/wide-habitat functional (distribution) group showed a significant difference between edge than interior habitats (ANOVA, F = 35.29, $p < 0.05$, Figure 5.7a). In contrast, for the forest- specialist species there was no significant difference (ANOVA, F=1.26, $p > 0.05$). During the field work it was observed that the number of shrubs in the margin group was far higher than in the forest-specialist group.

There was also a significant different between edge and interior habitats in the composition of vine species (ANOSIM, Table 5.1). Like shrubs, vine species in the forest-margin functional (distribution) group were more abundant in edge habitat (ANOVA, $F = 38.40$, $p < 0.05$, Figure 5.7b, Appendix 5.10), while forest-specialist species did not differ significantly between the two habitat types ($F = 5.24$, $p > 0.05$, Figure 5.7b, Appendix 5.11). Most of the forest-margin group vine species were observed to be present in the surrounding matrix and to be invading the patch from its edge after disturbance. At the edge of some forest fragments (such as Sibu1, Sibu2, Tulumako, Lagachancho and Gobe), they formed impenetrable thick entanglements intermingled with other woody species and appeared to restrict establishment and growth of tree seedlings (though they may also have provided some protection against browsing livestock). In almost all sampled patches, the forest-margin group lianas *Sericostachys scandens*, *Oncinotis tenuiloba* and *Gouania longispicata* were observed to form dense growth up the trunk of canopy trees and might have restricted rates of tree growth due to their weight on the tree crowns.

The composition of angiosperm geophytic herbaceous species is also different between the two habitat types (ANOSIM, Table 5.1, Appendix, 5. 12). This could be due to the presence of more ruderal species in edge than in interior habitats. However, the average abundance of both forest-specialist and forest-margin functional (distribution) groups did not differ significantly between edge and interior habitats ($F = 7.3$, $p > 0.05$; $F = 6.0$, $p > 0.05$ respectively, Figure 5.7c)

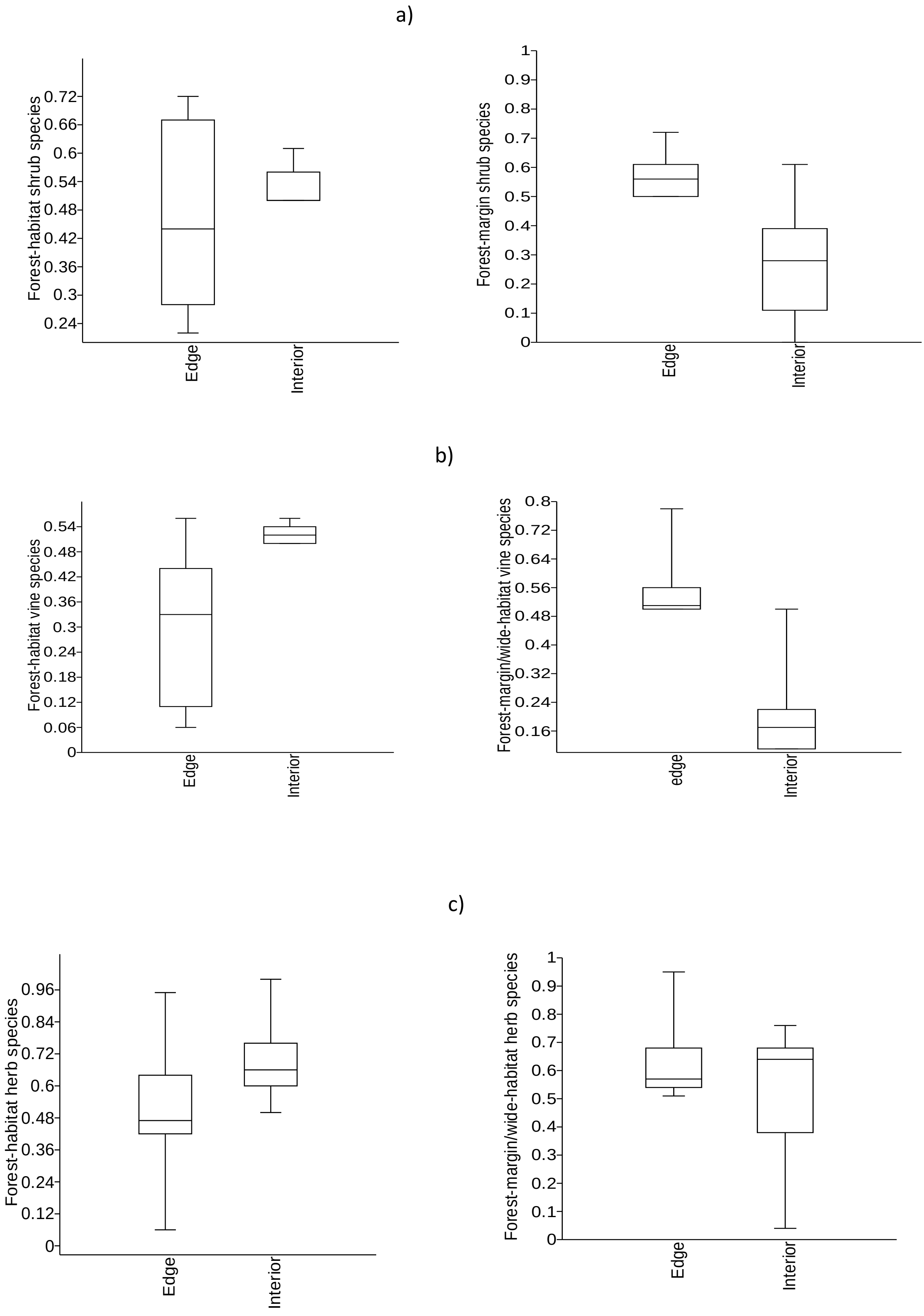


Figure 5.7 The difference in average abundance (for species with ≥ 0.50 average abundance as calculated by SIMPER) in edge and interior habitats of (a) shrub, (b) vine and (c) geophytic angiosperm herb species in two different functional (distribution) groups. Results are shown as a box plot: the horizontal line through each box depicts the median, the box depicts the central 50th percentile of data, and the two whiskers indicate the highest and lowest values.

5.3.5 Environmental, fragmentation and disturbance impacts on structure and species composition in patch edge and interior habitats

Tree individual density in both habitat types was not associated with any fragmentation variables (patch size r = 0.14, p = 0.57 and r = 0.04, p = 0.86; patch shape r = 0.27, p = 0.28 and r = 0.06, p = 0.80; edge density r = 0.37, p = 0.12 and r = 0.06, p = 0.8 for interior and edge respectively) or environmental variable (cumulative disturbance r= 0.26, p= 0.28 and r= 0.31, p = 0.20; rainfall r = 0.04, p = 0.87 and r = 0.11, p = 0.65; altitude r = 0.28, p = 0.27 and r = 0.25, p = 0.30 for interior and edge respectively). Likewise tree basal area and mean tree height in both habitats were not associated with fragmentation variables i.e. patch size, patch shape, edge density and environmental variables rainfall, altitude and slope. However, in the patch edge, but not in the patch interior, they were both significantly lower with greater cumulative disturbance (basal area r = -0.86, $p < 0.05$; height: r = -0.77, $p < 0.05$).

5.3.6 Ordination

Correlations amongst the environmental variables were tested using Pearson correlation analyses (Table 5.3). Rainfall and slope angle both increase with altitude, whilst patch interior disturbance (due to clear cutting, grazing and firewood collection) decreases with altitude. This disturbance was associated with coffee management which decreased with altitude and was rarely observed above 2000 m. Slope angle is positively associated with patch shape but negatively associated with patch edge density, interior disturbance and especially edge disturbance. Interior disturbance and edge disturbance are positively correlated. However, whilst edge disturbance is negatively correlated with patch shape, interior disturbance is not related to it. Similarly while interior disturbance is strongly positively correlated with edge density, edge disturbance is only weakly (and non-significantly) correlated with it.

Table 5.3 Preseason correlation coefficients (r) for relationships amongst environmental and disturbance variables for 18 forest patches in Illubabor Zone south-west Ethiopia.

	Altitude	Rainfall	Patch shape	Edge density	Interior disturbance	Edge disturbance
Altitude						
Rainfall	0.60*					
Patch shape	0.14	0.18				
Edge density	-0.41	-0.20	-0.32			
Interior disturbance	-0.55*	-0.23	-0.12	0.67*		
Edge disturbance	-0.28	-0.35	-0.56*	0.35	0.51*	
Slope	0.55*	0.51*	0.56*	-0.56*	-0.49*	-0.71*

* P < 0.05

The Canonical Correspondence Analysis (CCA) model fitted to the tree species x patches data against the environmental variables listed in Table 30 showed two axes to be significantly associated with the species distribution in a Monte Carlo test ($P < 0.05$, Figure 5.8, Table 5.4). However, total variation in species distribution explained by the first two axes was only 26% and even by the first four axes was just 38%; the total inertia was 1.5 (Table 5.5). The first axis (Eigenvalue 0.22) mainly represented altitude, rainfall and patch shape (which were very closely linked) versus edge disturbance (Table 5.6). Axis 2 (Eigenvalue 0.16) was most strongly linked with edge density and interior disturbance. No environmental variables were strongly linked with axes 3 or 4. The axes scores for individual plant species are shown in Appendix 5.4

Pooled mean score (mean±SE) of selected species functional groups (based on their habitat preference) is indicated on both axes (Appendix 5.17). Forest-specialist functional group species are scored (0.47±0.19 and 0.91±0.29 on axis 1 and axis 2 respectively). The distribution of these species along these axes showed no significant difference ($t = -1.077$, $p > 0.05$). However, the mean of these species is higher in axis two. Species in the wide-habitat and forest-margin functional groups scored 0.57±0.21 and 0.55±0.19, and 0.70±0.09 and 0.69±0.23 on axis 1 and axis 2 respectively. Likewise the two functional groups are not significantly difference along the two axes' gradients ($t = 1.254$, $p = 0.05$ and $t = 0.06$, $p > 0.05$ respectively).

Table 5.4 Results of 1000 Monte Carlo tests of the significance of the axes of a CCA of tree species in 18 forest patches in Illubabor Zone south-west Ethiopia.

Axis	**Axis 1**	**Axis 2**	**Axis 3**
Actual Eigenvalues	0.2196	0.1566	0.0926
Eigenvalue results from simulation			
Mean	0.1578	0.1156	0.0925
Maximum	0.2412	0.1624	0.1297
Minimum	0.0960	0.0827	0.0713
Probability	0.0070	0.0030	0.4705

Table 5.5 Variance and Eigenvalues of a CCA of tree species in 18 forest patches in Illubabor Zone south-west Ethiopia.

		Axis1	**Axis2**	**Axis3**	**Axis4**
Total variance (inertia) in species data	1.5				
Eigen value		0.220	0.157	0.093	0.087
% variance explained		15.083	10.755	6.360	5.944
Cumulative % variance		15.083	25.838	32.198	38.142
Multiple correlation species/environment scores		0.947	0.969	0.971	0.919
Sum of canonical Eigen values	0.718				

Table 5.6 Axis scores for environmental variables tested in a CCA of tree species in 18 forest patches in Illubabor Zone south-west Ethiopia.

Environmental variables	**Axis 1**	**Axis 2**	**Axis 3**	**Axis 4**
Altitude	0.297	-0.216	0.012	-0.038
Rainfall	0.232	-0.171	-0.064	0.163
Patch shape	0.169	-0.184	-0.024	0.061
Edge density	0.002	0.336	-0.070	0.050
Interior disturbance	-0.074	0.269	0.101	0.166
Edge disturbance	-0.296	0.100	0.127	0.028

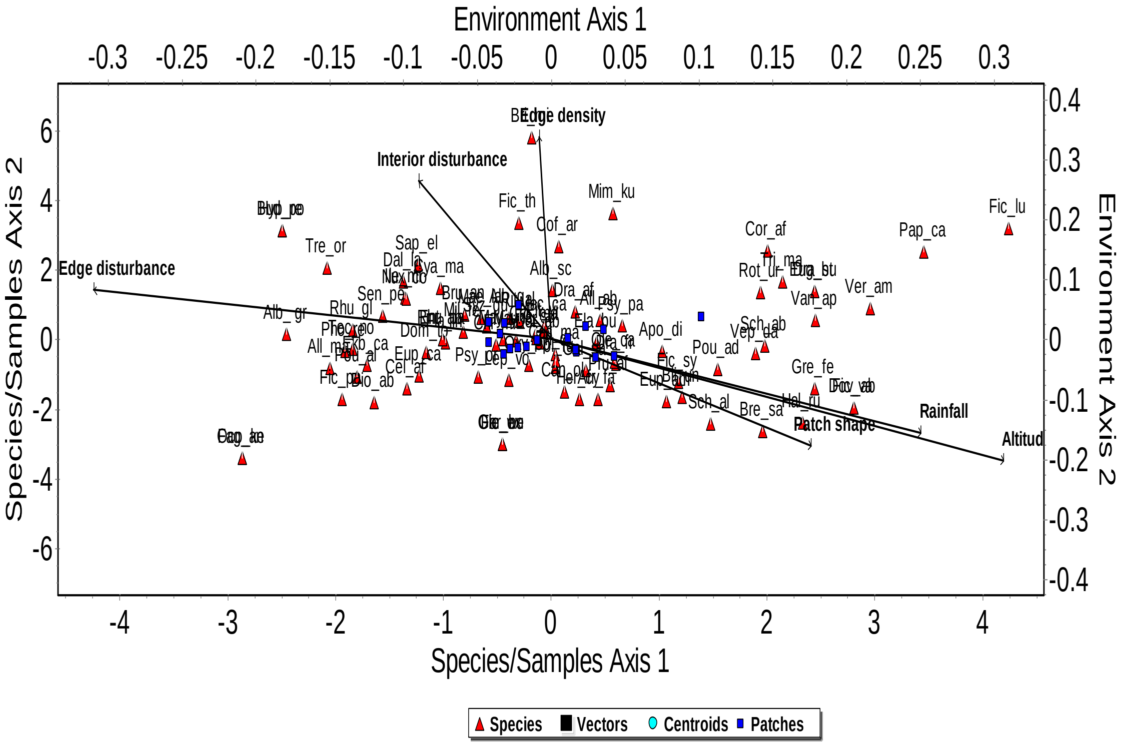


Figure 5.8 First and second axes of a CCA of all woody species (red) in 18 forest patches (blue) and six environmental variables in the Illubabor Zone of south-west Ethiopia. Species abbreviations are given in Appendix 5.5

Inferred ranking of the species along the environmental variables tested in the CCA enables us to identify species that are particularly positively or negatively correlated with each chosen environmental variable (Henderson and Seaby, 2007). The position of each species is projected onto the line formed by the environmental vector. The closer the point (species) to the line of the environmental vector, the higher its correlation with that environmental variable. The plots for edge disturbance and altitude (the two variables most strongly negatively and positively associated with axis 1 respectively) are shown in Figures 5.9 and 5.10. The arrow shows the direction of increasing magnitude for these environmental variables. Hence the proportion of species closely associated with high (> 2 axis score), medium (> 1 but < 2 axis score) and low (< 1 but > 0) values of altitude in a positive direction on axis 1 and high (> -2 axis score), medium (> -1 but < -2) and low (< -1 but > 0) value of edge disturbance in a negative direction on axis 1 was quantified. Twelve (13.8%) of species scored a high value, 11.5% a medium value and 19.5% a low value of altitude (Figure 5.9).

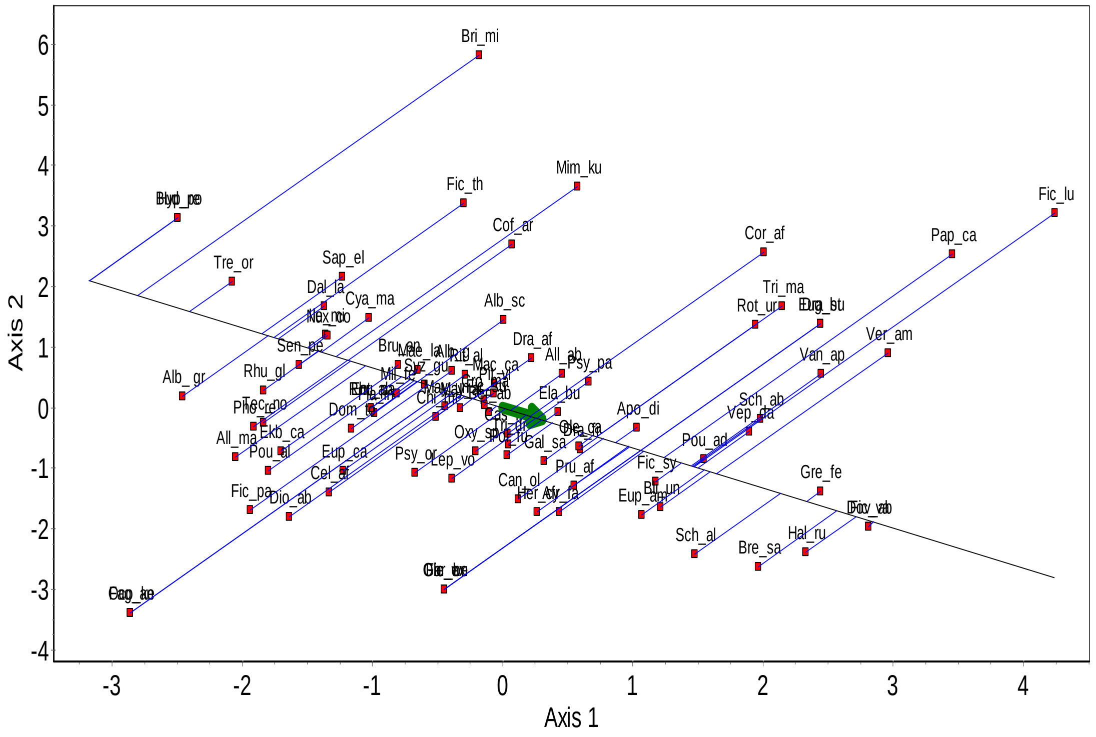


Figure 5.9 CCA biplot showing the inferred ranking of species along the vector of altitude for 18 forest patches in Illubabor Zone south-west Ethiopia. Species abbreviations are given in Appendix 5.5

Likewise, seven (8.1%), 18 (20.7%) and 23 (26.4%) of species scored high, medium and low values respectively of edge disturbance in a negative direction on CCA axis 1 (Figure 5.10). The three species placed at the extreme end of edge disturbance are *Albizia grandibracteata*, *Rhus glutinosa* and *Nuxia congesta*. These species were as all in the forest-margin functional (distribution) group. Similarly, 11 (12.6%), nine (10.3%) and 21 (24.1%) species scored high, medium and low values respectively of edge density in a positive direction on CCA axis 2, and eight (9.2%), 17 (19.5%) and 21 (24.1%) species scored high, medium and low values respectively of edge density in a negative direction (Figure 5.11, Appendix 5.14). *Bridelia micrantha* is placed at the extreme end of a positive association with high edge density while *Mimusops kummel*, *Ficus thonningii* and *Coffea arabica* were less strongly positively associated with high edge density (Figure 5.11).

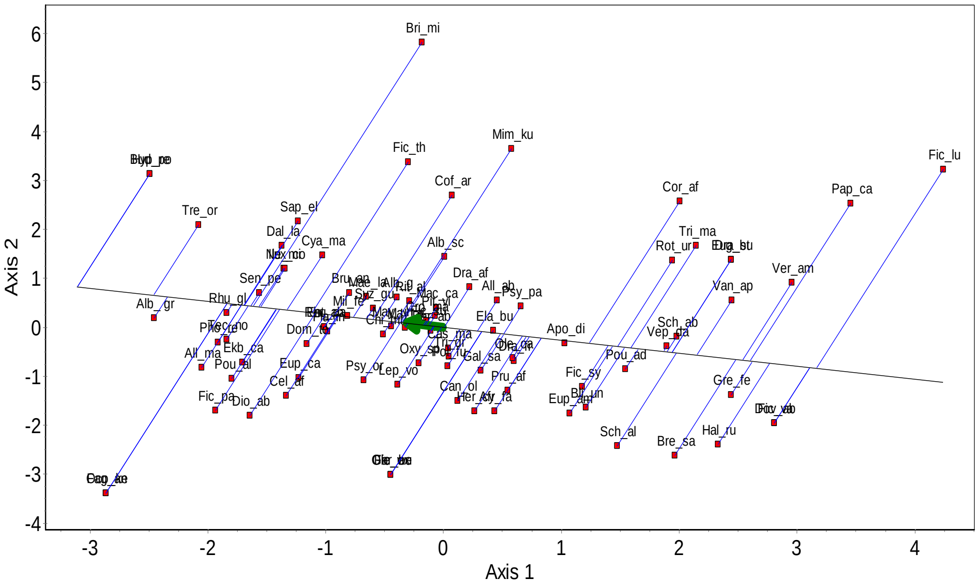


Figure 5.10 CCA biplot showing the inferred ranking of species along the vector of edge disturbance for 18 forest patches in Illubabor Zone south-west Ethiopia. Species abbreviations are given in Appendix 5.5

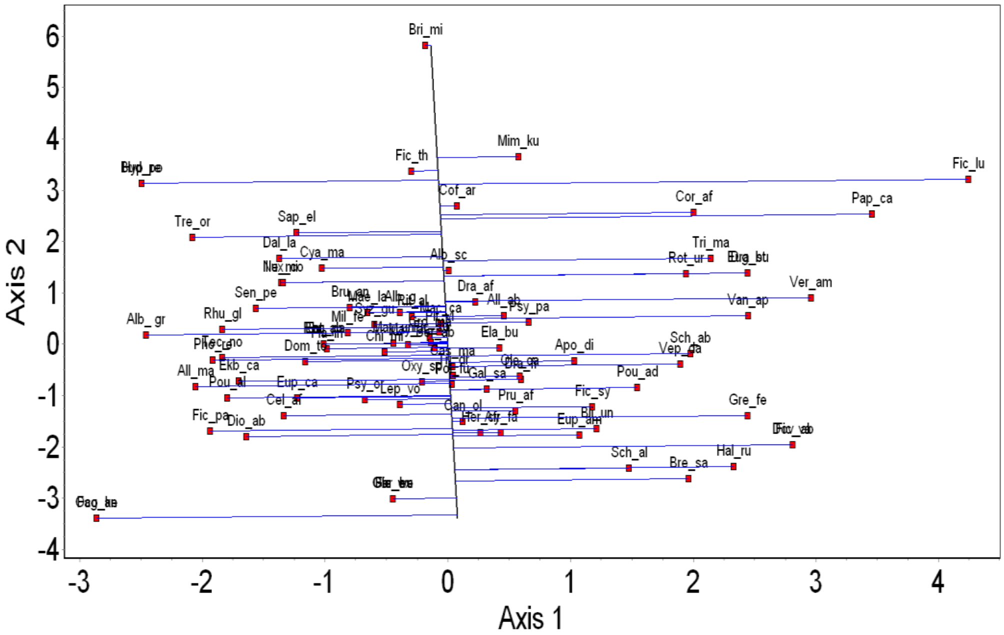


Figure 5.11 CCA biplot showing the inferred ranking of species along the vector of edge density for 18 forest patches in Illubabor Zone south-west Ethiopia. Species abbreviations are given in Appendix 5.5

5.4 Patterns of epiphyte and fern distribution and their response to fragmentation

5.4.1 Vascular epiphyte species richness, abundance and patterns of distribution in forest patch interior and edge habitats

A total of 50 vascular epiphyte species were recorded on 995 trees in the interior and 28 species on 293 trees in the edge habitat. Of these, 57% were orchids, 35% were ferns and 8% were in other groups of vascular plants (Appendix 5.6). Total vascular epiphyte species richness per patch was significantly greater in the interior habitat (mean±1.96*SE, 28.9±1.8; range 22-36) than the edge habitat (mean ± 1.96*SE, 13.6±1.4; range 7-19) (t = 8.957, DF 17, $p < 0.05$, Figure 5.12, Appendix 5.13).

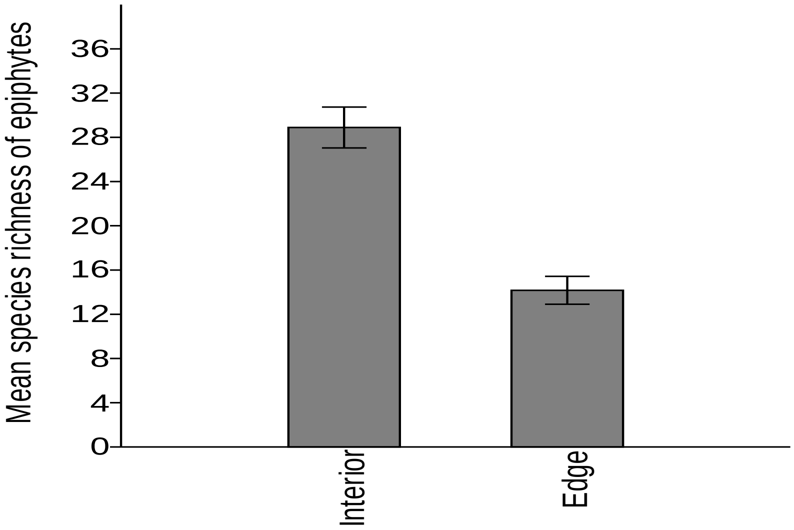


Figure 5.12 The mean (± confidence interval, 1.96*SE) species richness of epiphytes in interior and edge habitat of 18 forest patches in Illubabor Zone south-west Ethiopia.

The most frequent species of vascular epiphyte in the interior habitat was *Peperomia tetraphylla* which was recorded in 123 (85.4%) of the interior plots and in all 18 forest patches. This species belongs to the leathery leaf and succulent stem functional (morphological) groups. Five other species also occurred in ≥ 50% of the interior habitat plots, whereas no species occurred in ≥ 50% of the edge habitat plots (Appendix 5.6,

Figure 5.13). Twenty six species were noTable for having a frequency more than 5% greater in the interior than the edge plots, whereas only two species had a frequency more than 5% greater in the edge than the interior plots.

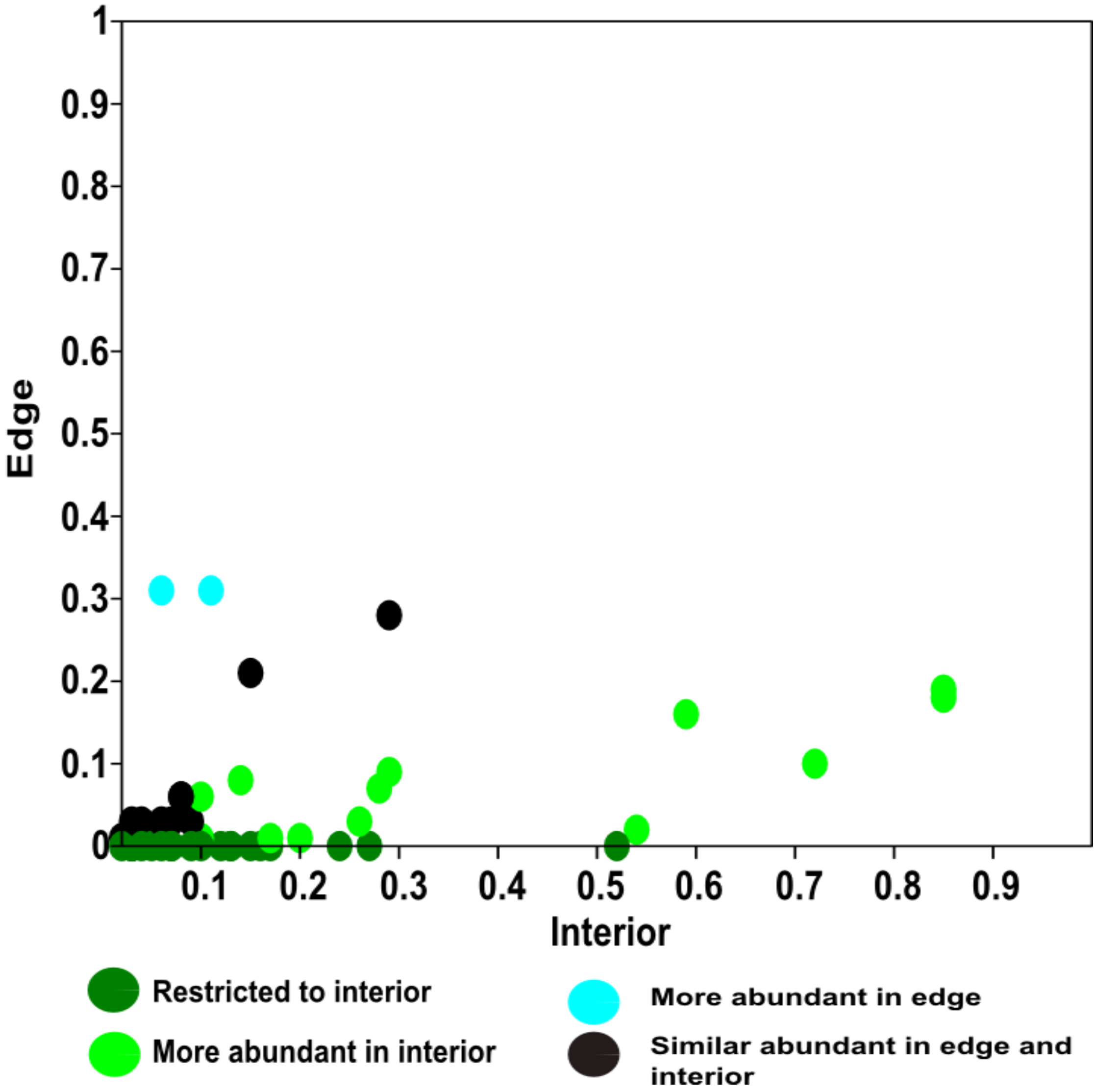


Figure 5.13Frequency of vascular epiphyte species in edge and interior habitats in 18 forest patches in the Illubabor Zone of south-west Ethiopia. Frequency is shown as the number of the 144 sample plots in each habitat in which the species was present. The species are classified into four colour-coded groups based on their relative distribution between the two habitats: (a) restricted to interior (absent from edge); (b) more abundant in interior (> 2.5 times higher frequency in interior); (c) similar in abundance in edge and interior (< 2.5 times different frequency between the two); (d) more abundant in the edge (> 2.5 times higher frequency in edge).

5.4.2 Distribution of vascular epiphyte plants between interior and edge habitats and effects of patch fragmentation and disturbance on density distribution

Combining all vascular epiphyte species together, the density of individual plants (or clumps) was significantly higher in the interior habitat (mean ±SE 114.5±6.3, range 62 - 158/m^2) than the edge habitat (mean±SE 42.7±3.7, range 24 - 85/m^2) of the 18 forest patches ($t = 10.829$, $df = 17$, $p < 0.05$). However, none of the tested indices of patch fragmentation had a significant effect on density distribution: patch size ($r = 0.28$, $p = 0.24$), shape ($r = 0.31$, $p = 0.19$) and edge density ($r = -0.13$, $p = 0.58$). Other environmental factors such as rainfall and altitude did not show a significant effect either ($r = 0.346$, $p = 0.15$ and $r = -0.091$, $p = 0.71$). In contrast, forest disturbance showed different effects on epiphyte density between the two habitats. While the interior habitat cumulative disturbance score showed no effect on epiphyte density ($r= 0.078$, $p = 0.7$) in the edge the cumulative disturbance score had a significant effect on epiphyte density ($r = -0.75$, $p = 0.00032$).

The mean (±SE) density of host trees with vascular epiphytes in the interior habitat (55.3±8.4) was higher than in the edge (16.4±8.5), (paired sample t-test, $t = 13.5$, $df = 17$, $P < 0.05$, Figure 5.14a). Epiphytes were recorded on host trees ranging greatly in size. The size range was 7-520 cm DBH in the interior and 13-350 cm DBH in the edge and the density of epiphytes was significantly positively correlated with host tree DBH in both habitats (interior, $r = 0.72$, $df = 17$, $p = 0.004$; edge, $r = 0.87$, $df = 17$, $p = 0.002$, Figure 5.15a & b respectively). However, the mean DBH of host trees was not significantly different between the interior (86.8±5.9) and edge (88.2±6.2) habitats ($t = -0.17$, $df = 17$, $P > 0.05$, Figure 5.14c) though it was highly variable per patch within both habitats (41.4-125.5 cm DBH in the interior and 52.0-142.5 cm in the edge, Appendix 5.13)

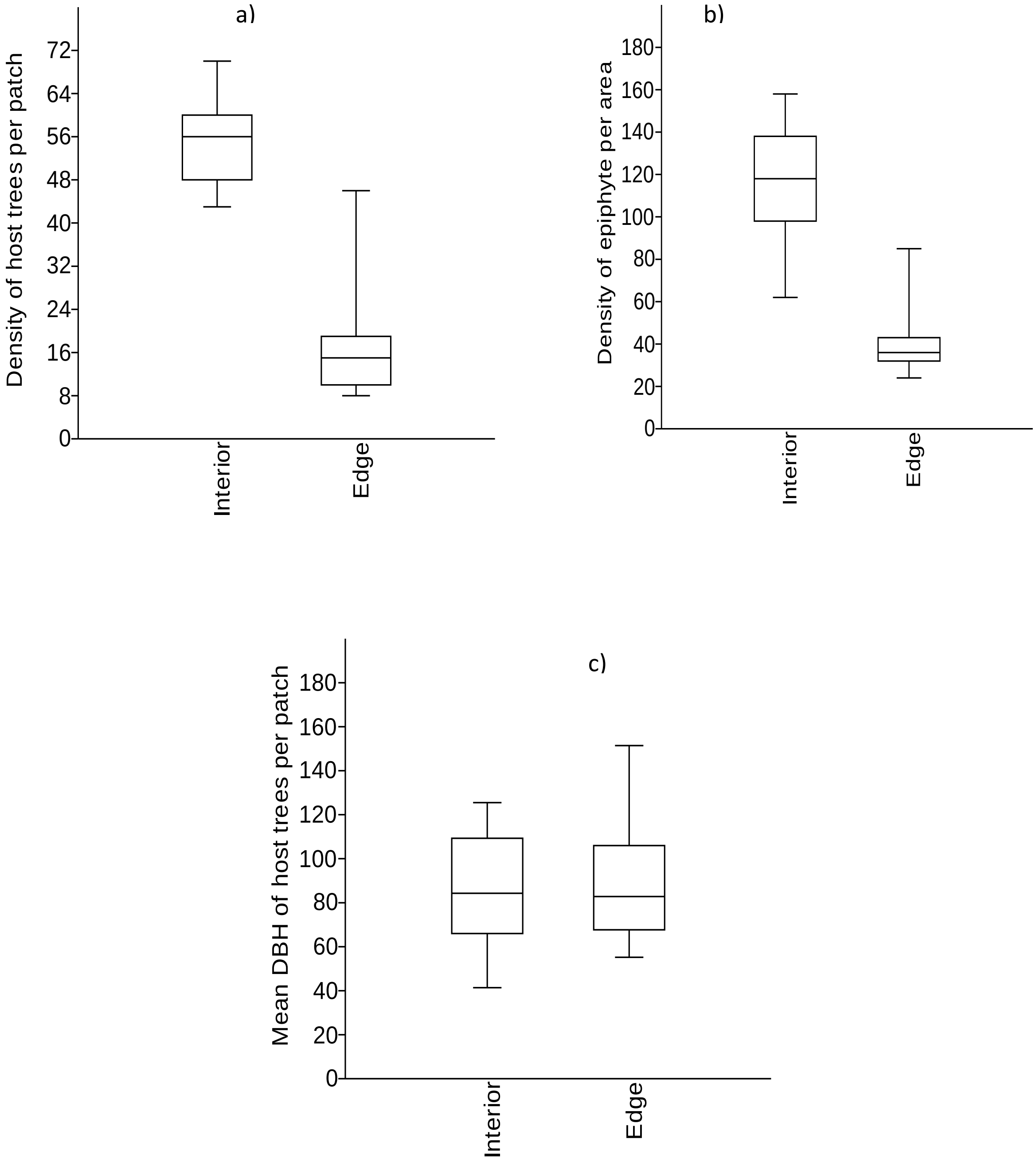


Figure: 5.14 Box plots of (a) individual density of host trees bearing epiphytes, (b) density of epiphytes per patch (plot/m^2), (c) mean host tree DBH (cm) in edge and interior habitats in 18 forest patches in Illubabor Zone south-west Ethiopia. The centre line shows the median, the box depicts the central 50th percentile of data, the whiskers depict the smallest and greatest value in the distribution.

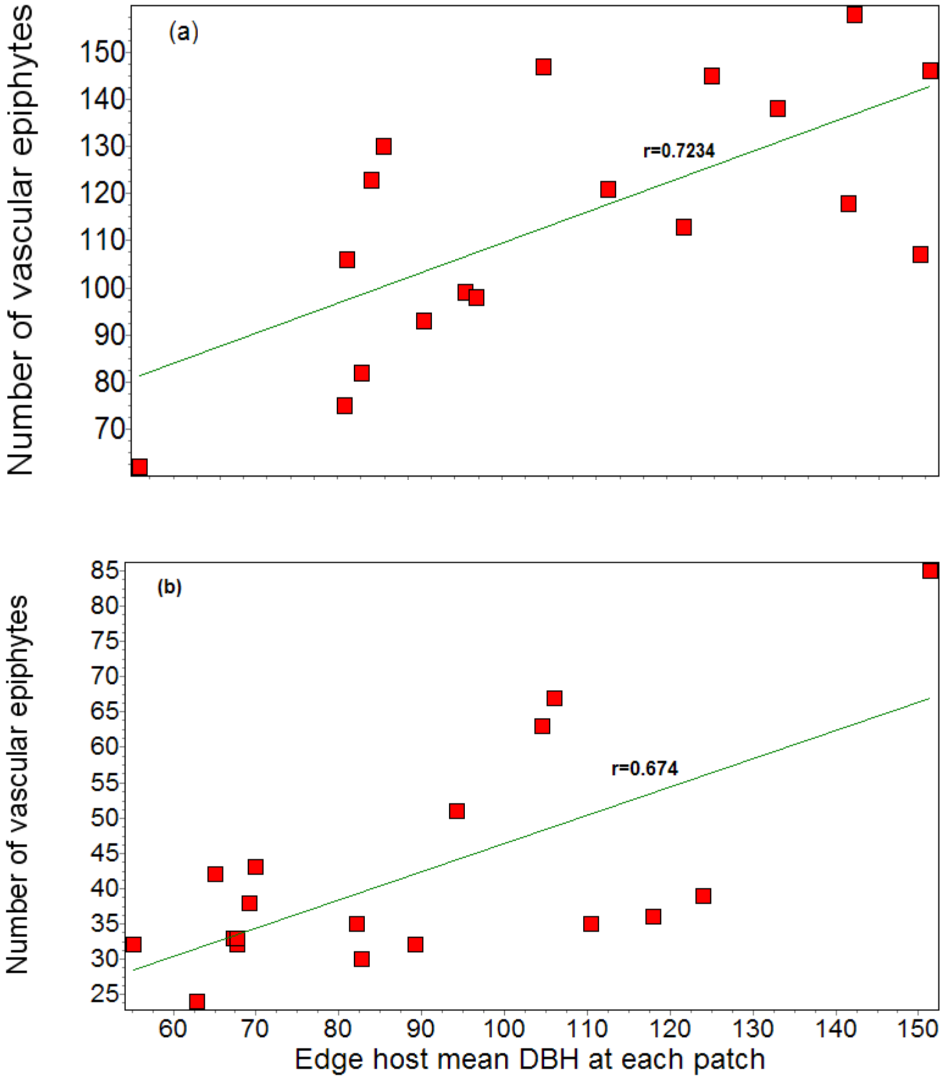


Figure 5.15 Relationship between host tree trunk diameter and number of individuals/clumps of vascular epiphytes in (a) interior and (b) edge habitats of 18 forest patches in Illubabor Zone south-west Ethiopia.

Species rank abundance (dominance-diversity) curves (Magurran (2004) for vascular epiphyte species showed a different shape in edge and interior habitats (Figure 5.16). In both habitats the community was dominated by three epiphyte species (*Peperomia tetraphylla* followed by *Peperomia rotundifolia* and *Asplenium theciferum*). Epiphyte abundance showed a sharp decrease with species rank over the first five species before flattening in the interior habitat while showing a more moderate decline over the first eight species before flattening in the edge (Figure 5.16). In the interior habitat 25 of the

50 species were “rare” (with an abundance < 10 per 400 m^2 of sample plots, whereas for edge habitat only 4 of the 28 species were this rare.

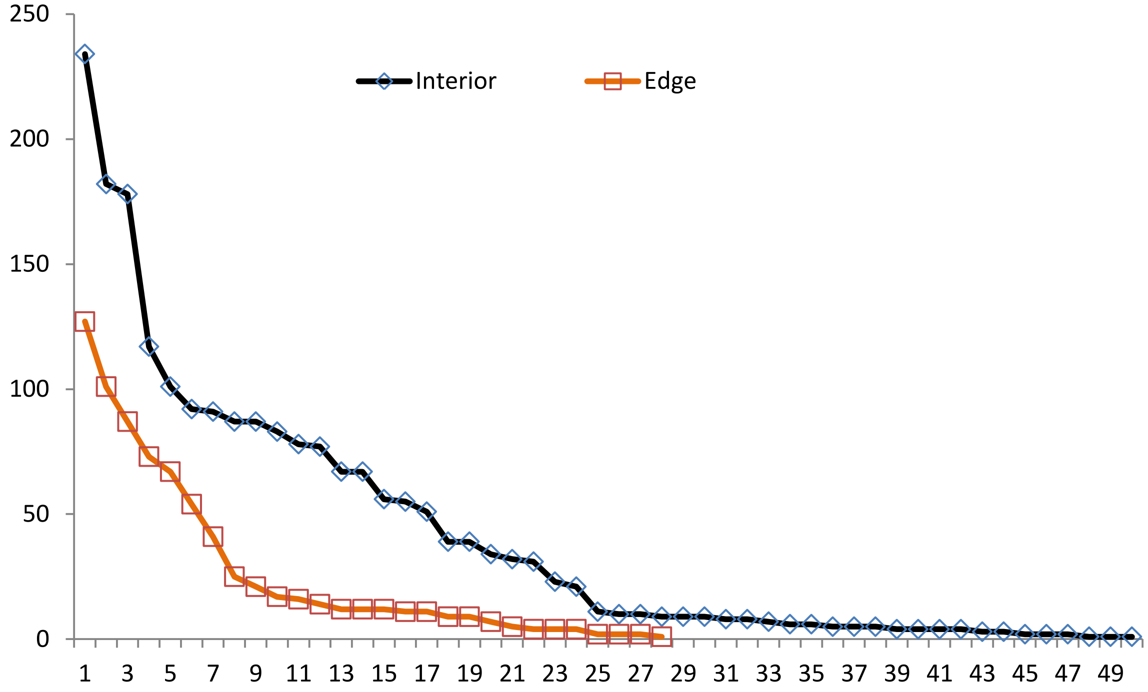


Figure 5.16 Rank abundance curves of vascular epiphyte species in interior and edge habitats in 18 forest patches in Illubabor Zone south-west Ethiopia. The index of abundance was density (number of individuals per 400 m^2 of sample plots).

The distribution of epiphytes amongst host trees of different DBH differed between interior and edge habitats (Figure 5.17a & b). The DBH of tree stems with epiphyte were grouped into 11 classes with 10 cm intervals. The percentage of individual tree stems with epiphytes generally increased with DBH in both habitats (though in the edge habitat there was no increase between 75 cm and 95 cm DBH (Figure 5.17). In the edge habitat, more than 95% of vascular epiphyte individuals were recorded on host trees > 30 cm DBH, compared with 85% in the interior. The percentage of individual trees < 30 cm DBH with epiphytes was more than 3 times greater in the interior habitat (9.9%) than the edge (3.2%). In terms of epiphyte density, individual trees supported from 1 to 54 epiphyte individuals clumps in interior habitat and 1-25 in edge. The mean number of epiphytes per individual host tree stems< 30 cm DBH in interior habitat (3.3%) was twice as greater as in edge habitat (1.3%). However, in both habitats the mean number of epiphytes increased with host stem DBH (Figure 5.17a & b) for the first five classes (10-19.9 cm to 50-59.9 cm) in the interior and for the first four classes for the edge habitat (10-19.9 cm to 40-49.9 cm). The distribution between 70 cm and 109.9 cm DBH showed more or less even number of epiphytes per host stem in the interior habitat while it kept increasing gradually in the edge habitat.

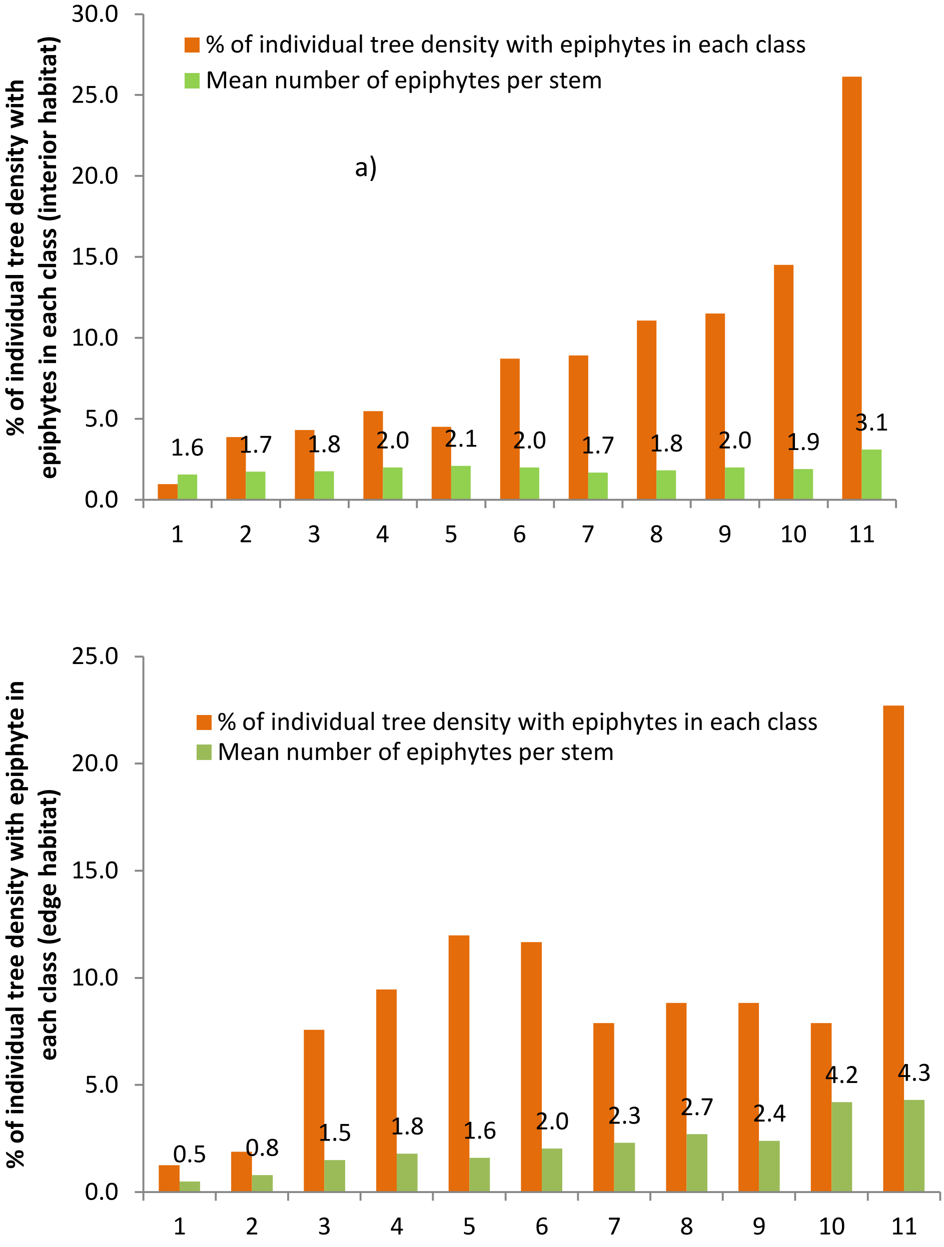


Figure 5.17 The distribution of vascular epiphytes between host tree stems of different DBH classes in a) interior habitat, b) edge habitat in 18 forest patches in Illubabor Zone south-west Ethiopia. The tree stems on which epiphytes were recorded are grouped into eleven classes with 10 cm DBH intervals. The minimum DBH for the classes is 10 cm (1= 10-19.9, 2 = 20-29.9, 3 = 30-39.9, 4 = 40-49.9, 5 = 50-59.9, 6 = 60-69.9, 7 = 70-79.9, 8 = 80-89.9, 9 = 90-99.9, 10 = 100-109.9, 11= > 110 cm).

5.4.3 Abundance of epiphyte functional groups in edge and interior forest habitats

The recorded epiphyte species were classified into two functional groups according to known distribution (forest-specialist species, forest-margin/widely-distributed species) and four functional groups according to morphology (woody and pseudobulbus stemmed species, herbaceous stemmed species, leathery and succulent leafed species, herbaceous leafed species) in order to compare their distribution between interior and edge forest habitats (Table 5.7, Appendix 5.6 and 5.15). Woody and pseudobulbous stems were classified into a single group, and leathery and succulent leaves were also classified into a single group. The forest-specialist species distribution functional group (mean±SE*1.96, interior = 41.4±17.9, edge = 15±10; paired t-test, t = 4.9, $p < 0.05$), and all four of the morphologically defined functional groups, woody and pseudobulbous stemmed species (interior = 17.3±8.1, edge = 5.2±2.2; t = 3.4, $p < 0.05$), herbaceous stemmed species (interior 47.2±20.5, edge = 13.6±11; t = 4.4, $p < 0.05$), leathery and succulent leaved species (interior = 46.2±31.2, edge = 19.3±4.9; t = 3.3, p <0.05, and the herbaceous leaved species (interior = 38.9±15.9, edge = 12.9±8.6; t = 4.4 $p < 0.05$) all had a significantly higher density in the interior than the edge habitat (Figure 5.18a, c, d, e, and f). Therefore, the only functional group with weaker (marginal) statistical evidence for a higher density in the interior habitat was the widely-distributed species group (interior = 34.7±22, edge = 12.8±4.9; t = 2.3, p = 0.05) (Figure 5.18b).

Table 5.7 The mean (± one confidence interval, SE*1.96) density of each functional groups of vascular epiphyte species in the interior and edge habitats in forest patches 18 forest patches in Illubabor Zone south-west Ethiopia.

	Density of each group	
Functional group	Interior habitat	Edge habitat
Forest-habitat/specialist distribution species	41.9±17.9	15.8±10
Widely-distributed species	34.7±22.0	12.8±4.9
Species with woody and pseudobulbous stems	17.3±8.0	5.2±2
Species with herbaceous stems	47.2±20.5	13.6±11
Species with leathery and succulent leaves	46.2±31.2	19.3±16.8
Species with herbaceous leaves	38.9±15.9	12.9±8.6

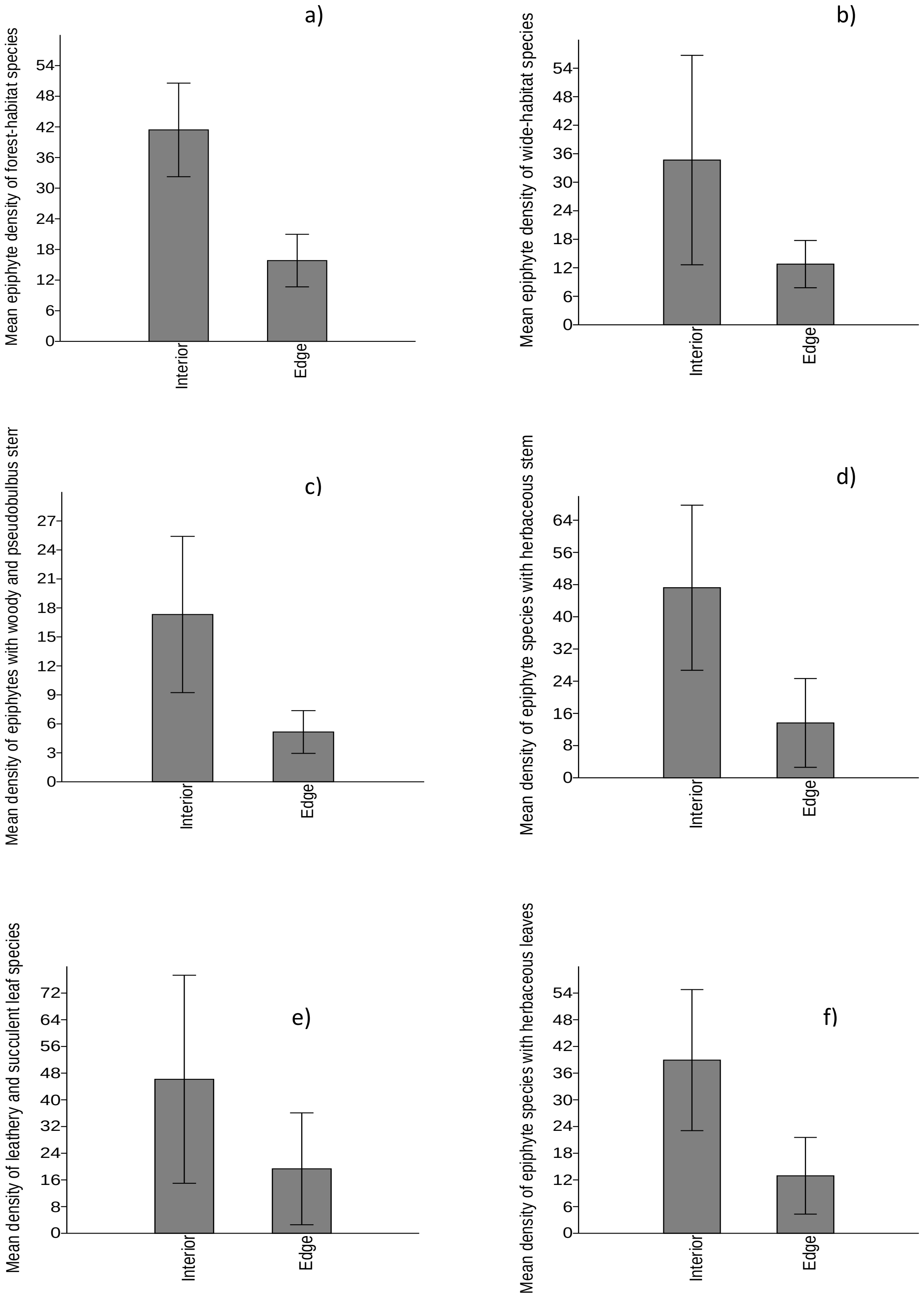


Figure 5.18 The mean (± one confidence interval, SE*1.96) density of six functional groups of vascular epiphyte species in interior and edge habitats of 18 forest patches in Illubabor Zone south-west Ethiopia: a) forest-specialist species, b) widely-distributed species, c) woody and pseudobulbous stemmed species, d) herbaceous stemmed species, e) leathery and succulent leaved species and f) herbaceous leaved species.

5.4.4 Geophytic fern species richness and species frequency between interior and edge habitats

A total of 30 species of geophytic fern were recorded in the interior and 26 in the edge habitats. Mean (±SE*1.96) species richness per forest patch was significantly greater in the interior (29.4±1.8, range 23-37) than the edge (22.1±1.4, range 17-27) habitats (paired t-test, t = 8.592; df = 17, P < 0.05, Figure 5.19).

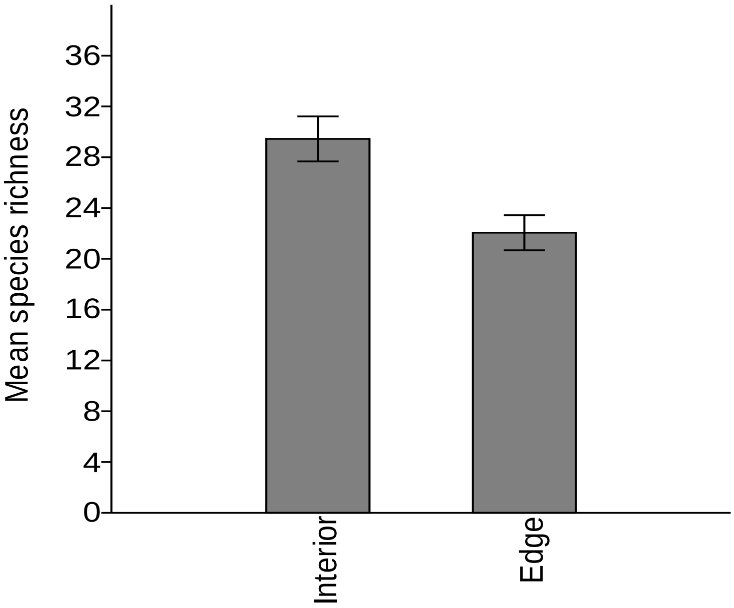


Figure 5.19 The mean (± one confidence interval, 1.96*SE) species richness of geophytic ferns in interior and edge habitats of 18 forest patches in Illubabor Zone south-west Ethiopia.

In terms of frequency amongst plots, across all 18 patches, of the 31 identified fern species, 16 were more than 2.5 times more frequent in the interior habitat (of which four, *Asplenium formosum, Asplenium sandersoni, Thelypteris confluens* and *Conigramme africana* were restricted to interior habitat), five in the edge habitat (of which one, *Pteridium aquilinum*, was restricted to the edge habitat) and 10 more evenly distributed between the two habitats (Figure 5.20, Appendix 5.7). There was a significant association between species allocation to these two groups and their a priori functional (distribution) groups: species more frequently in the interior habitat were more likely to forest-specialist species and those more frequent in the edge habitat were more likely to be forest-margin/wide-habitat species.

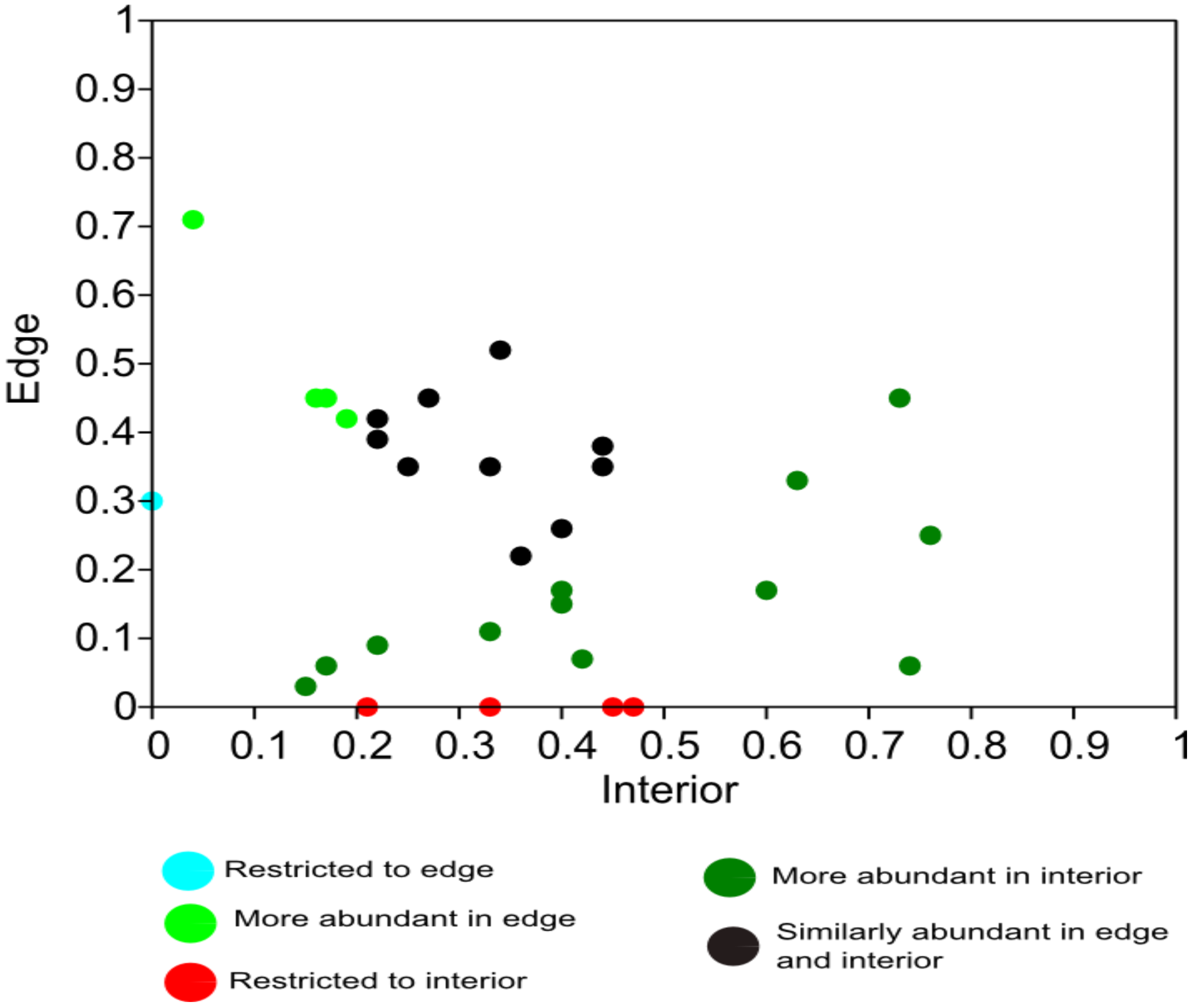


Figure 5.20 Frequency of geophytic fern species in edge and interior habitats in 18 forest patches in the Illubabor Zone of south-west Ethiopia. Frequency is shown as the number of the 144 sample plots in each habitat that the species was present in. The species are classified into five colour-coded groups based on their relative distribution between the two habitats: (a) Restricted to edge, b) more abundant in edge (> 2.5 times frequency in edge); c) restricted to the interior; d) more abundant in interior (> 2.5 times higher frequency in interior); (e) similar abundance in edge and interior (< 2.5 times different frequency between the two)

5.4.5 Geophytic fern species density between edge and interior habitats

The total density of geophytic ferns of all species was significantly higher in interior habitat (104±22.3 per patch (3200 m^2 of plots), range 77-137) than in edge habitat (59.8±13.7, range 34-88) (paired samples $t = 10.37$, DF = 17, $p < 0.05$). However, density across both habitat types was not significantly correlated with any tested indices of patch fragmentation: interior habitat epiphyte density, patch area ($r = 0.16$, $p = 0.53$), edge density ($r = 0.07$, p 0.73). or patch shape ($r = 0.44$, $p = 0.06$) (Appendix 5.14). The correlation of epiphyte density in interior was positive with shape of patch even though statistically not significant. Edge habitat epiphyte density: patch area ($r = 0.03$, $p = 0.90$), patch shape ($r = 0.07$, $p = 0.78$) and edge density ($r = 0.20$, $p = 0.40$). Likewise rainfall had no significant correlation with fern density ($r = 0.23$, $p = 0.35$), however there was a significant positive correlation for altitude ($r = 0.50$, $p = 0.03$). Similarly, summed disturbance score showed a significant negative correlation with fern density in both interior and edge habitats ($r = -0.60$, $p = 0.008$ and $r = -0.897$, $p < 0.001$ respectively, Figure 5.21, Appendix 5.14).

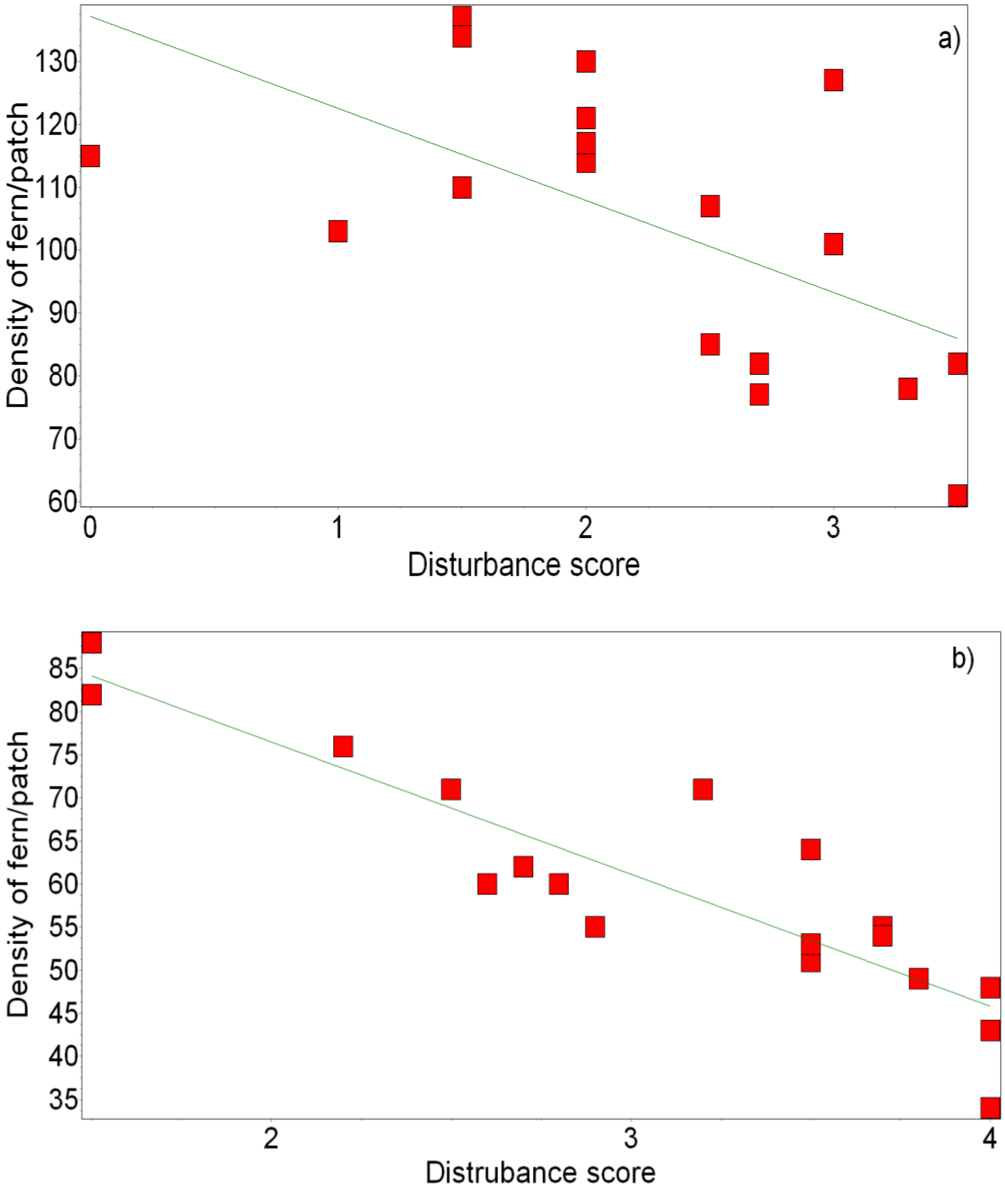


Figure 5.21 Relationship between summed disturbance score and density of geophytic ferns (number of individuals/clumps per 3200 m^2 of sample plot in each patch) in (a) interior and (b) edge habitats of 18 forest patches in Illubabor Zone south-west Ethiopia.

Species rank abundance (dominance-diversity curves) for geophytic fern species in edge and interior habitats showed a similar shape, with equivalent high levels of evenness, and low numbers of "rare" species (relative to the vascular epiphytes) (Figure 5.22). In the interior habitat the community was dominated by three species of Asplenium (*A. theciferum* followed by *A. gemmiferum*, and *A. lunulatum*). In the edge habitat there were also three dominant species in different genera (*Pteridium aquilinus* followed by *Arthropteris monocarpa* and *Drynaria volkensii*).

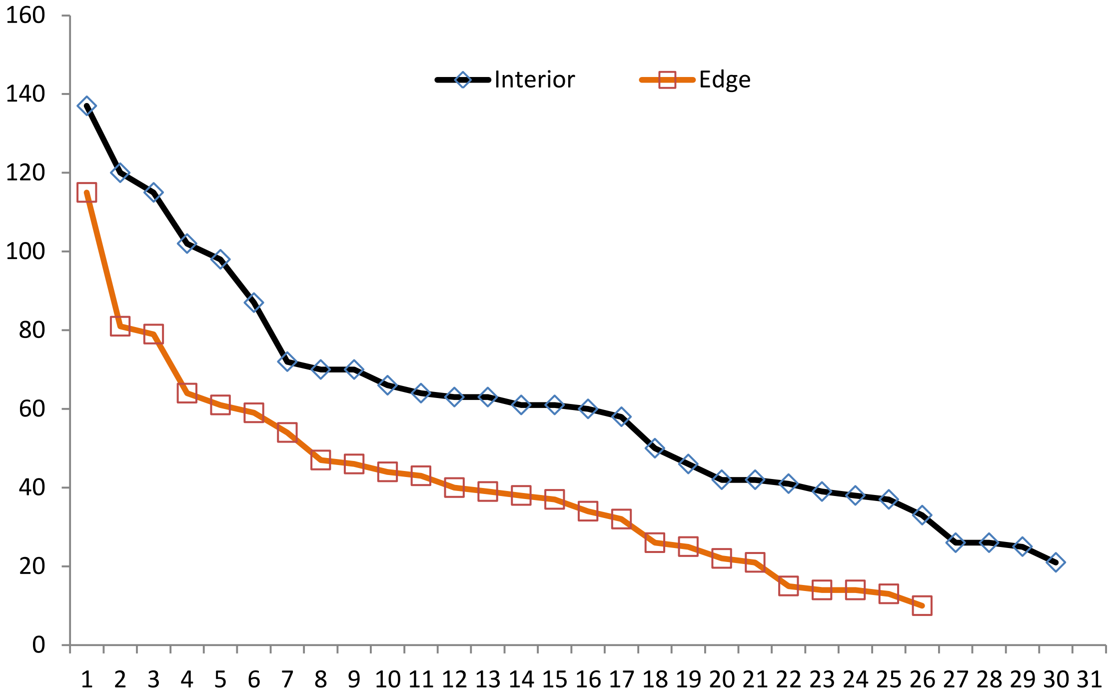
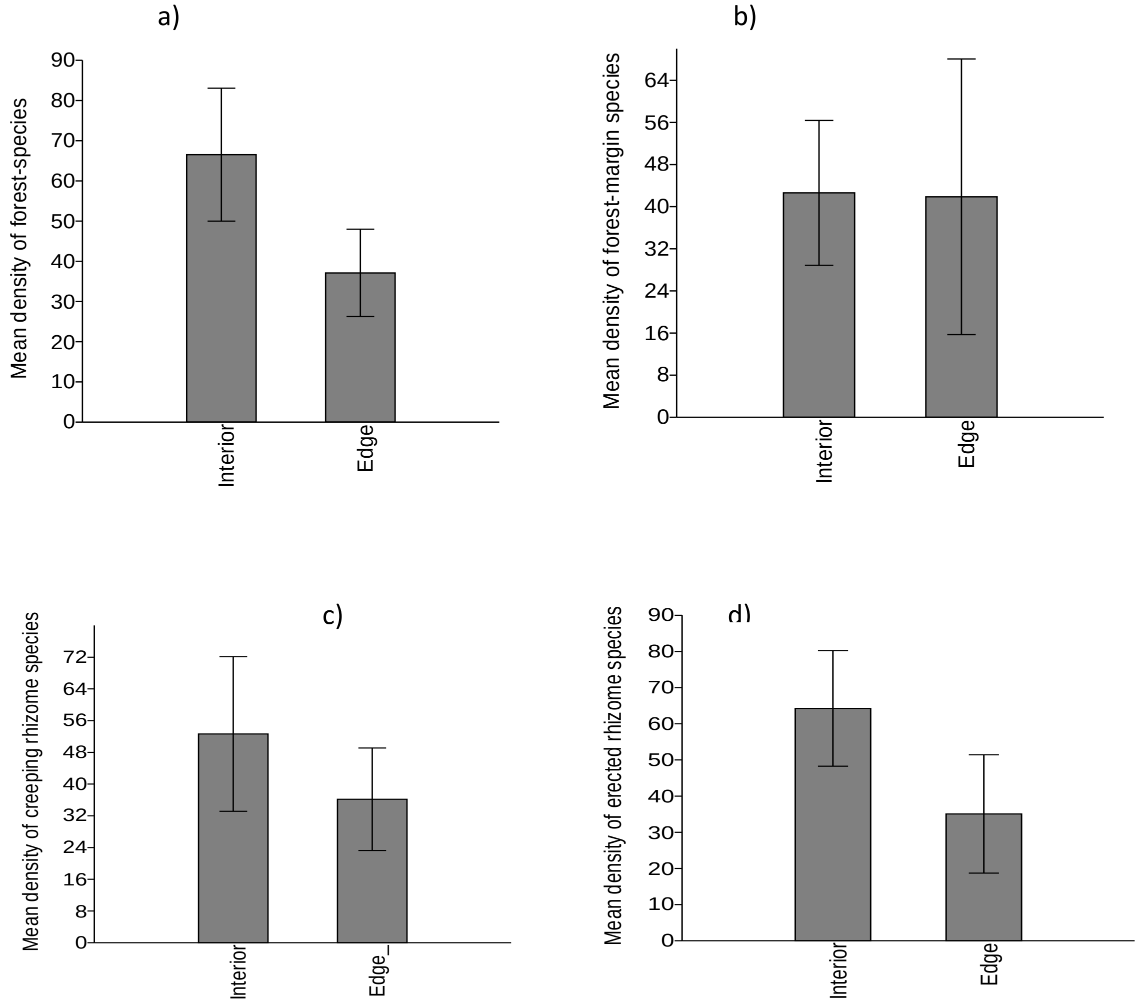


Figure 5.22 Rank abundance curves of geophytic fern species in interior and edge habitats in 18 forest patches in Illubabor Zone south-west Ethiopia. The index of abundance was density (number of individuals per 400 m^2 of sample plots).

The species functional groups differed in the variation of their density between interior and edge habitats (Table 5.8, Figure 5.23). Of the functional groups defined by distribution, forest-specialist species had a significantly higher density in interior (mean±SE*1.96, 66.5±16.5) than in edge (37.1±10.8) habitat ($t = 3.88$, $p < 0.05$, Figure 5.23a), however forest-margin species showed no difference in density between the two habitats (interior 42.6±23.1, edge 41.9±26.2, $t = 0.04$, $p > 0.05$, Figure 5.23b). Of the functional groups based on morphology, species with both creeping (interior 52.6±19.5, edge 36.2±12.9) and erect (interior 64.3±15.9, edge 35.1±16.4) rhizomes had a significantly higher density in the interior habitat ($t = 2.82$, $p < 0.05$ and $t = 2.23$, $p < 0.05$ respectively, Figure 5.23c & d). Similarly species with tufted fronds (interior 75.2±17.6, edge 35.2±14.1) had a significantly higher density in interior habitat ($t = 3.808$, $p <0.05$, Figure 5.23f), however those with spaced fronds did not differ significantly between the two habitat types (interior 47±14.3, edge 35.5±16.9; t = 1.008, p > 0.05, Figure 5.23e, Appendix 5.16).

Table 5.8 Mean (± one confidence interval, SE*1.96) density (number of individuals per 400 m^2 of sample plots) of six functional groups of geophytic fern species in the interior and edge habitats of 18 forest patches in Illubabor Zone south-west Ethiopia.

	Density	
Functional groups	Interior habitat	Edge habitat
Forest-specialist species	66.5±16.5	37.1±10.8
Forest-margin/wide-habitat species	42.6±23.1	41.9±26.2
Creeping rhizome species	52.6±19.5	36.2±12.9
Erect rhizome species	64.3±15.9	35.1±16.4
Spaced frond species	47±14.3	35.5±16.9
Tufted frond species	75.2±17.6	35.2±14.1



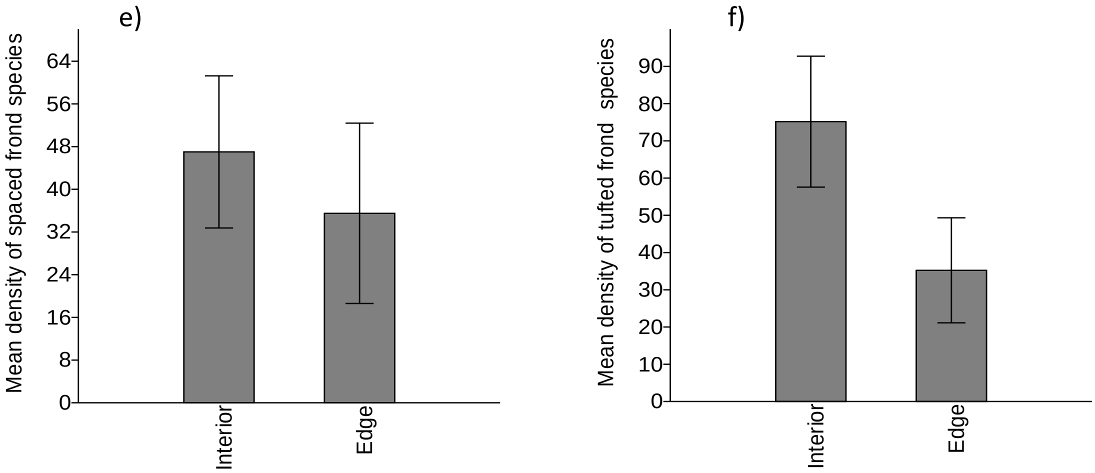


Figure 5.23 Mean (± one confidence interval, SE*1.96) density (number of individuals per 400 m^2 of sample plots) of six functional groups of geophytic fern species in interior and edge habitats of 18 forest patches in Illubabor Zone south-west Ethiopia. a) forest-specialist species, b) forest-margin species, c) creeping rhizome species, d) erect rhizome species, e) spaced frond species and f) tufted frond species.

5.5 Discussion

5.5.1 Variation in woody species composition between edge and interior habitats

There was a significance difference in species composition between edge and interior habitats of the studied fragmented forest patches. The difference in species composition between these two habitats types was due to species of different habitat preference and morphological groups tending to be associated with either the interior or edge habitats. In the edge habitat there was high abundance of either forest-margin or wide-habitat distribution species. In terms of morphology these tended to be species of types well-suited to drier and/or hotter environments. The tree species with a distribution largely restricted to forest habitats were, as expected more abundant in patch interior than edge environments, reflecting the way that they have become “filtered out” of edge habitats and confined to the interior. This process may have been quite rapid in patches that have only recently lost a high proportion of their original area from the margin. The relative density and frequency of forest-specialist woody species were respectively eight and three times higher in the interior habitat compared with edge habitat. These findings are in accord with previous studies carried out in human-modified landscapes that have described the effects of fragmentation on species composition in habitat patches (Honnay et al., 2002; Echeverria et al., 2007; Galanes and Thomlinson, 2009; Gonzalez et al., 2010). Human-induced fragmentation and within-habitat disturbance (forest clearance, extraction of canopy trees for timber, grazing and firewood collection) were found to strongly influence species composition and habitat conditions (Honnay et al. (2002). Through time, these human-induced fragmentation and disturbance effects in Illubabor Zone have been found to lead to simplification of ecosystems through a reduction in species diversity (Vanderwel et al., 2011). In such simplified forest habitats, the species composition had become dominated by a generalist set of widely dispersed or forest-margin species, some of which could be characterised as light-demanding pioneers, supporting the findings of Hill and Curran (2005). This has been shown to reduce the diversity of composition amongst habitats, especially in their understory forest-habitat-dependent/shade-tolerant species (Chapter 4).

The greater abundance of forest-specialist woody species in the interior than the edge habitat could be attributed to: (a) their intolerance of edge environments; (b) selective cutting of these species in the edge habitat; and (c) suppression of their seedlings by grazing and possibly competition from herbaceous plants in the edge habitat. Forest-habitat-associated species like *Vepris dainellii, Chionanthus mildbraedii, Allphylus abyssinicus, Oxyanthus speciosus, Canthium oligocaprp* and *Cassipurea malosana* were reported to be amongst the most valued plants by local communities for farm tools, roofing and fencing material and they were observed to be selectively cut and harvested in the edge habitat. These species usually grow in forest under the shade of canopy trees where there is an organic layer of decomposing leaf litter on the soil surface and where there is a lower cover of herbaceous plants and less vegetation trampling and soil compaction (Pual and Fine, 2002; Flora of Ethiopia and Eritrea, Vol. 3). These factors may all contribute to lower establishment and survival rates in edge habitats. These species may also be suppressed in the edge by browsing, grazing and trampling by cattle from the surrounding matrix. Consequently, at the time of this study, forest-specialist species were often seen in the edge habitat as coppice re-growth from the stumps of recently cut stems. The seedling and sapling of these species were rare relative to coppice re-growth.

The species found in high abundance in the edge habitat, in contrast, belong to the group of forest-margin and wide-habitat distribution tree species, e.g. *Syzygium guineense, Maesa lanceolata, Croton macrostachyus, Macaranga capense* and *Albizia gummifera*. Most of these species have little value to the local community because of their soft wood (pers. comm. from local elders). On the other hand, most of these forest-margin and wide-habitat distribution species have a pioneer ecology and were observed to be regenerating at a high density in the studied edge habitats associated with high rates of fruiting. These species must therefore also be tolerant of other edge effects such as grazing and vegetation trampling because of their close association with such disturbed parts of forests (Tadesse, 2003). Hill and Curran (2003) observed a similar trend that light-demanding/forest-margin species regenerate in exposed gaps in edge habitat and cover the area faster than forest-habitat species. They found that the proportion of regenerating forest-margin/light-demanding trees increased with increasing edge effect as compared with shady interior conditions. As a corollary of this, Echeverria et al. (2007) found that shade-tolerant species gradually increase in abundance towards the patch interior while shade-intolerant species declined. They showed a spectrum of tree species responses to habitat fragmentation, some benefiting and some disadvantaged. The species benefiting from fragmentation had a higher density of seedlings in edge habitat and were characterised as light demanding (Metzger 2000; Honnay et al. 2002; Echeverria et al., 2007) as was the case in the present study. This evidence suggests that further fragmentation and within-patch disturbance of the fragmented habitats in Illubabor might seriously affect forest-habitat/shade-tolerant species (Echeverria et al., 2007).

5.5.2 Forest structural composition

Forest composition or edge and interior habitats was characterised on the basis of stem density, basal area, diameter and height class distributions, and dominance (basal area). Although there was high variation amongst patches in the density of individual trees, there was no significant difference in mean individual tree density between edge and interior habitats. This may be attributed to the replacement of the density of forest-habitat species in the interior habitats by a similar density of forest-margin/light-demanding and wide-habitat/generalist species in the edge habitat. In contrast, the basal area of trees was significantly greater in patch interior than edge habitats which were attribuTable to the existence of a higher density of big trees in the interior, corresponding to the findings of (William-Linera, 1990) in Mexico. In the edge habitats there had been much more intensive selective tree cutting for timber extraction in the past. A reduction of basal area in edge as compared with interior habitat represents a modification of the forest structure in which the forest has returned to an earlier successional stage (Echeverria et al., 2007).

When the structural composition of different functional groups of species is considered, the stem density, basal area and abundance (IVI calculated from the sum of relative basal area, relative stem density and relative frequency) of forest-specialist species was very significantly higher in the interior habitat than in the edge (Table 5.2). The interior: edge ratios of basal area and stem density of forest-specialist trees were 8:1 and 6:1 respectively, indicating a huge impact of the edge effect on forest stand composition.

5.5.3 Vertical structure

The vertical structure of vegetation in a forest is as important to many species as the size of the forest (Zehm et al., 2003). Change of this structure may cause changes in processes and habitat condition such as microclimate, nutrient cycling and plant-plant interaction. The present study found that the vertical structure of interior habitat was different from the edge. Edge disturbance simplifies the complex structural diversity of interior habitats. The four distinguishable vertical components in the interior forest stands were reduced to three in the edge. The top structural component of interior habitat entirely consisted of mature canopy trees from the functional groups of forest-margin and wide-habitat/generalist species: *Syzygium guineense, Pouteria adolfi-friederici, Prunus africana, Albiza gummifera, Trilepisium madagascariense, Olea welwitschii* and *Elaeodendron buchananii*. The tallest species in the interior habitat (reaching 30-45 m in height) was *Pouteria adolfi-friederici* followed by *Prunus africana*. Below this, the second component of interior habitat comprised a mixture of forest-margin/light-demanding, wide-habitat/generalist and forest-habitat species like *Macaranga capense, Croton macrostachyus, Milletia ferruginea, Diospyros abyssinica, Allophylus abyssinicus* and *Cassipourea malosana*. The third component comprised the functional group of forest-specialist species such as *Vepris dainellii, Chionanthus mildbraedii, Allphylus abyssinicus, Oxyanthus speciosus, Canthium oligocaprpa, Cassipurea malosana, Dracaena fragrans, Galinera saxifrage, Lepidotrichlia volkense, Psydrax parviflora, Psychotria orophila, Diospyros abyssinica* and *Ilex mitis*. The fourth (lowest) component comprised forest-habitat shrub species, like *Rytigynia neglecta, Erythrococcaa trichogyne, Maytenus gracilipes* and *Justicia betonica*, and forest-habitat vine species such as *Oncinitis tenuiloba*, *Hippocratea africana*, *Paullinia pinnata* and *Keetia gueinzii* intermingled with the shrubs.

The edge habitat was characterized by a simpler structure. The top component comprised a low density of high-canopy trees from the same species group as in interior habitats, but the tallest species were different and lower: *Syzygium guineense and Macaranga capense* reaching up to 25 m height. The second component differed from the interior and comprised only the forest-margin/light-demanding species *Maesa lanceolata, Croton macrostachyus, Nuxia congesta, Rhus glutinosa* and *Rhus quartiniana*. In contrast, the third component of edge habitat had a similar physiognomic character, but was comprised of different species, than the fourth component of interior habitat. In the edge habitat it consisted of shrub species such as *Vernonia spp.*, *Lippia adoensis*, *Lantana terifolia* and *Phytolocca dodecandera* and vine species such as *Clematis spp.*, *Helinus mystacinus* and *Gouania longispicata*. The mean height of trees was significantly lower in edge than interior habitat. The third structural component of interior habitat was missing in the edge habitat. However, the density of trees in the 7-12 m height range was notably higher in edge than interior habitat and they almost all consisted of forest-margin species. The density of trees was markedly less in edge than interior habitat in all height classes taller than 12 m.

5.5.4 Effect of fragmentation and environmental factors on species composition

The species composition and distribution in forest fragments was found to be mainly influenced by habitat disturbance, altitude, rainfall, and patch shape. The CCA result suggests that habitat disturbance alone could largely explain the species composition and patterns of distribution amongst patches. However, edge disturbance was highly confounded by variables such as altitude, rainfall and patch shape so their effects could not be distinguished. A number of studies have also shown that there is a strong stratification of species along gradients of environmental factors and forest spatial patterns (Arias-Gonzolez et al., 2008). Understanding these influences of environmental and spatial variables in driving variation in species composition is an important and a growing area of community ecology research (Jones et al., 2008).

5.5.5 Epiphyte species composition and distribution in edge and interior habitats

Orchids dominated the epiphytes identified in this study, followed by ferns and then other vascular plants, which is consistent with other findings from Africa (e.g. Biedinger and Fischer, 1996; Zapfack et al., 1996). Epiphytes constitute a specialized plant community because of their extreme habitat. They are good indicators of environmental quality of an area since they are sensitive to changes in humidity caused by opening up of forests by fragmentation (Benzing, 1998; Werner and Gradstein, 2009). In the present study, the species richness of epiphytes was significantly greater in interior than in edge habitats. This difference was strongly linked to disturbance, epiphyte species functional group and presence of large host trees. The lower epiphyte richness in edge habitat coincides with patterns described from other areas (e.g. Kromer and Gradstein, 2003; Wolf, 2005). Those studies found that epiphyte diversity is positively associated with the occurrence of host trees and microclimatic conditions. In the present study, epiphyte community composition was noticeably different between edge and interior habitats due to differences in the relative abundance of epiphyte functional groups differing in distribution/habitat association and morphological characteristics. I found that forest-habitat species largely dominated the composition of the patch interior habitats, but were relatively less common in the edge habitats. On the other hand, the patch edge habitats were strongly dominated by forest-margin and wide-habitat distribution species that were poorly represented in the interior habitats.

Leaf and stem morphological characteristics were also linked to the distribution of epiphyte species between the two habitats. Species with herbaceous leaves and stems, such as *Elaphoglossum acrostidoides*, *Elaphoglossum lastii*, *Asplenium sandersonii* and *Bolusiella iridifolia*, were relatively scarce in edge plots, which were dominated by species with woody, succulent or bulbous stems, e.g. *Cyrtorchis arcuata*, *Calyptrochilum christyanum, Microcoelia globulosa, Diaphananthe fragrantissima* and *Polystachya tessellata*. This result is expected from the evidence of previous studies that edge habitat is exposed to multiple micro-climatic effects including increased light levels, wind velocity and temperature, and reduced air humidity (Laurance, 2004). The resulting desiccating microclimate conditions have been found to be unsuiTable for epiphytes with herbaceous leafs and stems (Hylander 2005; Werner et al, 2005). Desiccation increases with forest structural disturbance that results in increased direct sunlight through canopy opening and wind penetration (Laurance, 2004). Thus, the requirement for high humidity and low energy loads by epiphytes with herbaceous stems and leaves is likely to limit their population persistence in edge habitat. The morphological characters of the epiphyte species found to be more common in the edge habitat in the present study (succulent leaves and succulent, woody or bulbous stems) are associated with a suite of morphological and physiological traits that greatly reduce rates of water loss (Werner and Gradstein, 2009) and store water and nutrients (La Croxi, 2008).

Epiphyte abundance increased with increasing host tree size in both the interior and edge habitats, which is in accord with previous studies showing that host tree size distribution impacts on epiphyte community structure, with tree size distributions closely mirroring gradients of disturbance (Heizer, 1999; Are´valo and Betancur, 2006). In the present study, as recorded by others, edge habitat appeared to have suffered higher rates of tree mortality from natural wind throw (Murcia, 1995) and from selective tree cutting for timber extraction, which had lead to a reduced density of large (height and diameter) tree stems and greater canopy openness (Laurance, 2004). In the present study, the higher density of large trees, and more complex vertical canopy structure, in the interior habitat were observed to be associated with a greater heterogeneity of epiphyte microsites, ranging from fully exposed to very shaded, with larger amounts of trapped organic matter. This is in agreement with the findings of Friedel et al. (2006) who recorded an increasing number of epiphyte species with an increasing host tree size along an edge to interior gradient. However, I observed some situations where some species previous considered to be highly forest habitat-specialist had survived in fragmented and disturbed edge habitats (Appendix 5.6). These species (e.g. *Peperomia tetraphylla* and *Peperomia rotundifolia*) were noTable for possessing some desiccation-resistant traits, e.g. succulent leaves, and succulent or leafless stems. It is not known for how long they can persist in the edge habitat: it is possible that their presence is restricted to more recently exposed edge habitat, or that which is adjacent to a less open surrounding matrix. Nonetheless, this observation indicates the potential value of using data on the functional traits of species, rather than just their known distribution and habitat-associations from past specimen collection records (as reported in floras), for predicting their response to forest disturbance and fragmentation (Appendix 5.6)

5.5.6 Geophytic fern species richness, diversity and composition in edge and interior habitats

Fragmentation and disturbance in forest fragments in Illubabor Administrative Zone did not appear to have the same impact on all fern species. While the majority of fern species (14 out of 31) suffered from the effect of fragmentation and disturbance on edge habitats, a few (perhaps five of the species) seemed to benefit from the altered edge habitat. Overall, however, it was clear that forest fragmentation and disturbance led to a major compositional difference in geophytic ferns between edge and interior habitat. Observations of the ground vegetation in edge habitat showed that it had been densely colonized by grasses, other weedy herbaceous plants and vines, which are likely to have impacted negatively on the abundance of forest-habitat specialist ferns through reducing the availability of suiTable microhabitats for their establishment (Laurance et al., 2007). It was noted that within both the edge habitat and under canopy gaps in the interior habitat, forest-specialist geophytic ferns were completely absent amongst dense ground layer vegetation dominated by weedy grass, herb and vine species, which is in accordance with the findings of Babaasa et al. (2004). As discussed for epiphytes above, Thuiller et al. (2004) found that the distribution of ferns amongst such forest habitats is closely linked to their physiological traits associated with water relation (Silvia et al, 2011). In the present study, the species *Asplenium gemmiferum, Asplenium sandersoni, Asplenium formosum* and *Conigramme africana* were noted as forest-habitat specialists never seen growing away from deep moist shade that, from their morphology, are unlikely to be tolerant of edge effects. One fern species stood out as benefitting most from disturbed forest-edge habitat, the tall and dense-growing rhizomatous, *Pteridium aquilinum* in accordance with its known ecology world-wide (e.g. Marrs & Watt, 2006) as well as in Ethiopia (Flora of Ethiopia, Vol. 1).

The density of ferns was much higher in interior than in edge habitats and in less disturbed than more disturbed areas of edge, and this appeared to be attribuTable to the higher density of forest-habitat specialist species in the interior and less-disturbed habitats. However, the species rank abundance curves in both habitats followed a similar shape (with a lower abundance in the edge habitats across all species rank positions). The loss of forest structural complexity in the edge also causes great differences between the two habitat types in addition to disturbance, forest area reduction and patch shape complexity (Paciencia and Prado. 2005).

5.5.7 Implication for conservation

It has been suggested that habitat fragmentation may be more a problem of edge effects than of spatial habitat structure, e.g. shape and area of patches (Honnay et al., 1998). Forest-habitat specialist species were found to be greatly reduced in abundance, or even absent, from the forest patch edge habitats. The results could be interpreted as showing a process by which the upper story and under story structures of the forest stand become increasingly simplified and dominated by a single group of widespread forest-margin/light-demanding species. Extrapolating from the results of this study to forest patches across south-west Ethiopian landscapes it is likely that these habitats are becoming increasingly structured into two zones with an edge dominated by such forest-margin/light-demanding, or wide-distributed generalist species of little conservation value, whereas the forest-habitat specialist species of greater conservation value (and also of greater value for a range of other locally important ecosystem services) are becoming mostly confined to the interior habitat of the patches. As disturbance and microclimatic change generated from the surrounding matrix penetrate further into patches (effectively expanding their edge zone) the forest interior habitats capable of supporting these vulnerable species will become increasingly diminished (or even eliminated in small patches) which poses a significant threat to biodiversity conservation. Therefore, the importance of the protection of forest interior habitats is very important for future biodiversity conservation in this region.

CHAPTER 6

SACRED GROVES AS SITES FOR BIODIVERSITY CONSERVATION IN SOUTH-WEST ETHIOPIA

Abstract

There have been many studies on the importance of sacred groves in biodiversity conservation but only a few have compared sacred groves with non-sacred forests. Sacred groves in Ethiopia are generally considered to be small and not to support many species of conservation importance. I tested this assumption through comparative study of plant species richness, diversity, composition and regeneration dynamics in sample plots within six sacred groves and four non-sacred forests differing in their level of disturbance in the Gamo highlands of south-west Ethiopia. Open-ended individual and group interviews were conducted with the custodians of each sacred grove to evaluate its importance for biocultural diversity. To provide a wider context of the past reduction in area and fragmentation of sacred groves and non-sacred forests in the whole 66,765 ha Gamo highlands study area, 1995 and 2010 Landsat Thematic Mapper (TM) satellite images were analysed using maximum-likelihood classification to map land-use and land-cover.

Species accumulation curves showed no difference between the two forest types. However, analysis of the sample plot data by Renyi species diversity profiles showed higher species diversity in the sacred groves than in the non-sacred forests. There were no noTable differences in tree stem density or sapling density, between the two forest types, however seedling density and mean basal area of trees > 5 cm dbh was significantly higher in the sacred groves than the non-sacred forests. The species composition of sacred groves differed from non-sacred forest. However, this cannot be attributed to sacred status as the two forest types are different in topographic position (aspect and slope) and geology.

The local custodians reported that the sacred groves had great religious and social significance, and provide local people with benefits such as plants used for medicine and rituals, environmental protection as wind breaks and soil cover that reduces erosion, and cultural services in the form of a symbols of identity and beauty.

The satellite image analyses indicated a high rate of land-cover modification around the sacred groves during the 1995-2010 periods. The farm and settlement land-cover types increased from 28.8% to 60.4% while others reduced: wooded grassland decreased from 35.1% to 17.2%, open pasture land from 25.3% to 15.6% and forest from 10.8% to 6.8%. The total number of forest fragments decreased from 1718 to 1441 due to the complete conversion of some to other land uses. The mean patch size of forests decreased from 4.1 to 3.0 ha. Likewise mean shape index and edge density decreased from 1.5 to 1.3 and 14.8 to 10.5 respectively, indicating human impacts on the margins of the groves. At the patch level, while all non-sacred forests decreased in size, only a few of the sacred groves decreased indicating that the sacred groves have been better protected.

In conclusion, sacred groves suffered a lower rate of deforestation than did the non-sacred forests. Sacred groves contained a higher density of national priority species for conservation, e.g. *Cordia africana* and *Hagenia abyssinica*, and a species recognized as internationally vulnerable, *Prunus africana*, indicating their important role in biodiversity conservation in south-west Ethiopia. Despite the lack of recognition of sacred groves by conservation officials in Ethiopia, the custodians of sacred groves have employed cultural practices that do maintain biodiversity. Giving appropriate legal support to both culture and value systems of the communities, as they are applied to sacred groves, could reduce the current threat that sacred groves are facing and enhance the conservation of biodiversity. The value of the existing network of sacred groves is well suited to this objective as they cover a wide variety of habitat types and are well distributed across the landscape.

6.1 Introduction

Deforestation and degradation of wooded habitats due to anthropogenic activities are among the major contributors to today's global climate change and biodiversity reduction. Land-use change is considered to be the main factor (Lambin et al., 2003). Human population pressures are expanding the area of land-use, such as agriculture and settlement, into natural habitats in all parts of the world to meet the demand for food and housing (Lambin et al., 2003). These land-use changes have led to deforestation further aggravating fragmentation of forest habitats (FAO, 2003). In response to this, large investments have been made in the establishment of nature reserves across the world to preserve large pristine areas. The management of most of these reserves, however, has not been successful as the initiatives are often politically driven and aspire to achieve environmental benefit without the involvement of immediate users or local communities (Khan et al., 1997; Brown, 2003; reviewed in Bhagwat and Rutte, 2006). On the other hand, many local communities conserve tracts of forest habitats in their habitation area in the form of sacred groves for cultural purposes. Despite increased pressures, these forests have existed for centuries where they are under the custody of local communities and religious leaders (Swamy et al., 2003). In this context, successful conservation of biodiversity may require the incorporation of cultural traditions and conservation practices of indigenous communities for future mitigation of deforestation, degradation, habitat loss and climate change.

Sacred groves are patches of forest located in human habitation areas and protected by the local community in accordance with their customary laws (Gerden and Mtallo, 1990), taboos and religious beliefs (Aumeeruddy and Bakels, 1994, Campbell, 2004, Madeweya et al., 2004) for their cultural, social and material values (FAO, 2003). They exist in various forms, including natural relict forests as a site of ancestral or deity worship (Ramakrishana et al., 1998; Khan et al., 2008) and burial grounds (Mgumia and Oba, 2003; Madeweya et al., 2004). The management of these communal sacred places has often been found to be very effective where it is firmly rooted in the local communities' institutions and cultural traditions (Kokou, 2006). However, the major role sacred groves can play in biodiversity conservation has generally been undervalued due to a lack of comparative studies on their importance in conservation. This is despite such areas of forest being known to contain valuable biodiversity that has survived despite pressures for forest conversion (Salick et al., 2007). Conservation of these sacred groves and associated biocultural diversity, in collaboration with local stewards, can therefore make a potentially important contribution to reduction of biodiversity loss and habitat degradation.

The importance of sacred groves in biodiversity conservation has been studied and acknowledged by many authors who have documented the role of traditional conservation systems supported by religious beliefs of the indigenous community (Aumeeruddy and Bakels, 1994; Wadley and Colfer, 2004; Sukumaran and Jeeva, 2008, Khan et al., 2008; Wild, 2008). Examples of such preserved forests are found widely in many countries and are particularly well known in India (Chandrakanth et al., 2004; Bhagwat and Rutte, 2006; Ormsby and Bhagwat, 2010), where their high density of plant species of conservation importance has been documented (Sukumaran and Jeeva, 2008; Page et al., 2009, Rao et al., 2011). The existence of sacred groves has also been reported in, Europe, Asia, America and Australia (Khan et al., 2008), for example sacred mountains surrounded by woodland in Italy (Borys, 2000). In Asia, Sherpa people in Nepal are noted for their management of sacred groves and these places continue to support a rich biodiversity (Stevens, 2008). In America sacred forests are set aside as memorials of dead people (Svendsen and Campbell, 2010). In some African countries like Benin, Togo, Congo, Ghana, Tanzania and Kenya sacred groves have been documented as depositories of plant species due to their careful management by traditional stewards (Chouin, 2002; Mgumi and Oba, 2003; Madeweya et al., 2004; Combell, 2004, 2005; Lebbie and Guries, 2008; Fomin, 2008; Msuya and Kideghesho, 2009; Sheridan, 2009). Of these studies, however, very few have made comparative analyses between sacred and non-sacred forests to adequately substantiate their frequent claims of the high level or even unique importance of sacred groves for biodiversity conservation (Mgumi and Oba, 2003).

In Ethiopia, recent studies of sacred groves have showed that they are important for biodiversity conservation (e.g. Binggeli et al, 2003; Alemayehu and Teketay, 2006; Bongers et al., 2006; Aerts et al., 2006; Aerts, 2007). These studies mainly focused on sacred Ethiopian Orthodox Church forests, surrounding churches and monasteries, particularly in the northern part of the country. However, Ethiopia also has many sacred groves that belong to the indigenous communities, are more associated with non-Christian belief systems and do not have church buildings inside them or in their vicinity. These indigenous sacred groves have been comparatively neglected by researchers and conservationists (Desalegn, 2007). These indigenous sacred groves are particularly commonly among communities in south-west Ethiopia, where they have been protected by strong local belief in their status as sacred sites and consequent strict application of religious practices, taboos, customary laws for their protection by indigenous institutions (Desalegn, 2007). These practices are notably different from those promoted by non-governmental conservation organizations and government agencies, in that they are generally applied to sites of a smaller size, under the traditional tenure of community institutions, are based on indigenous religions and taboos and are closely aligned with local cultural traditions. Because of this these sacred groves may have had a continuity of forest cover for a very long time whilst cycles of anthropogenic deforestation and forest regeneration have occurred over thousands of years across most of the rest of the Ethiopian highland landscape (Tewolde, 1988, 1989). In this case they may act as important refugia for relic flora and fauna. However, no assessments have been made on their biodiversity importance in contrast to the attention paid to large areas of state-managed or non-sacred community-managed forests. Consequently, the possible role of these sacred groves for biodiversity conservation has been comparatively neglected by conservation organizations in Ethiopia (Desalegn, 2007).

The sacred groves of south-west Ethiopia are currently under threat from various anthropogenic pressures due to land-use changes such as farm and settlement expansion, deforestation and forest degradation, e.g. through over-exploitation of resources (Desalegn, 2007). Religious beliefs and community rules that were central to the protection of sacred groves and their biodiversity are also being threatened by changes in value systems, acculturation and integration of traditional communities into dominant religious beliefs, ultimately leading to loss of indigenous knowledge (Desalegn, 2007). Locally adapted indigenous management practices and governance institutions have become weakened in this area due to lack of recognition by institutions external to the communities. The absence of legal backing for the role and authority of the traditional custodians of the sacred groves and lack of recognition of customary laws and use rights by local government has further aggravated the situation leading to deforestation and degradation of sacred groves (Desalegn, 2007). A loss of appreciation for the sacred groves has also occurred due to cultural change and economic pressure within the local communities, reflected in particular by a change in attitudes between the current generation of young people and older generations (Desalegn, 2007). All of these factors have combined to cause a reduction in the area, integrity and the strength of associated culture of the sacred groves of south-west Ethiopia.

The underlying rationale of this study was the conservation of biodiversity in Ethiopia with a particular focus on components of biodiversity with high “biocultural” value. The specific objectives were to establish the conservation value of the biodiversity of the sacred groves in south-west Ethiopia, the level of threat that these habitats are currently under, and to use this information to propose the highest priorities for conservation action. It is intended that the results of this study will provide the evidence to influence the policies of international and national institutions towards the conservation of sacred groves of south-west Ethiopia.

To meet these specific objectives, a set of sacred groves and non-sacred forests were selected and plot-based vegetation survey was used to record their floristic composition and conservation status. Evidence of recent disturbance within the forests was evaluated as an additional indicator of threat. The biocultural value of the groves (current sacred status, people’s perceptions, management and uses) were evaluated qualitatively by interviewing local elders and custodians. The effect of agricultural expansion leading to encroachment reducing the number and area of sacred groves, and increasing their fragmentation, was explored with satellite images by analyzing land-use and land-cover changes.

6.2 Materials and Methods

6.2.1 Site selection and method of vegetation data collection

An exhaustive reconnaissance survey was undertaken all over the Gamo highlands which identified 272 sacred groves. From these six sacred groves were selected and matched to four non-sacred forests based on elevation similarity and physical environment and proximity to each other (Figure 6.17).The site selection was also constrained by local permission to enter sacred groves and availability of an approximately equal number of non-sacred forests. Remaining non-sacred forests are very few in number in the Gamo Highlands, so only a maximum of four could be sampled, which constrained the sampling regime for the comparison.

The size of the six sacred groves ranges between 0.7 and 33.6 ha, whereas the four non-sacred forests range from 16.6 to 400 ha. I made a pre-assessment within each sampled sacred and non-sacred forest to determine the representative vegetation cover. After pre-assessment, a main plot of size 60 m x 100 m was chosen. This size was chosen to have standard plot size for both small and large forests. One main plot was established in five sacred groves and two main plots in one sacred grove (in Tele). Likewise two main plots were established in three non-sacred forests and a single main plot in one non-sacred forest (Oche forest). The position of each main plot was chosen subjectively to best represent the structure and composition in the interior of each forest. The two plots in the same forest were laid at a 20 m distance parallel to each other. Within each of the ten sites (six sacred groves and four non-sacred forests) a systematic plot sampling design was used. Each plot was divided into fifteen 20 m x 20 m sub-plots. Of these, eight sub-plots were sampled (Figure 2.10) which gave a total of 56 sub-plots in the sacred groves and 56 sub-plots in non-sacred forests. Within each sampled sub-plot, the diameter of trees with ≥ 5cm at breast height was measured using diameter tape. The density, i.e. number of individual tree/plot and cover abundance of trees species was recorded. In addition, the diameter and height of each individual tree was recorded. Five sub-sub-plots of 5 x 5 m were established within each sampled 20 x 20 m sub-plot, one in each corner and one in the centre for seedling and sapling data collection. Seedlings and saplings were identified to the species level and counted. In each sub-plot sprouting from cut stumps was recorded for each species as (i) the number and diameter of stumps which had any sprouts and (ii) the number of all young sprouts per stump. Large size tree stem that have had a sprout origin was considered as individual tree. The presence/absence of herbaceous species, including epiphytes and ferns, was recorded in each sub-plot. All vascular plant species per plot were identified and recorded. Herbarium specimens of all species present in a sub-plot that could not be identified in the field were collected and taken to Addis Ababa University and identified using authenticated specimens. The nomenclature of plant names in this study follows the flora of Ethiopia (Hedberg and Edwards, 1989 &1995; Edwards et al., 1995, 1997, 2000; Hedberg et al., 2003, 2006; Mesfin Tadesse, 2004).

The effect of disturbance was evaluated subjectively for each sub-plot. Disturbance from (i) grazing (rating proportion of herbaceous ground cover removed and vegetation trampled), (ii) forest clearance (trees cut for timber or fuelwood collection and associated damage caused during tree fall) were separately given a score between 0 and 4, where 0 = no disturbance, 1 = low disturbance, 2 = intermediate, 3 = moderate, 4 = highly disturbed. The scoring was based on the intensity of the disturbance in terms of proportion of the ground cover vegetation and tree biomass removed from 0 to 100% in approximate equal size categories. The scores for each sub-plot were summed for each main plot separately to generate a cumulative value. For the forests with two main plots, the average value of each of the two disturbance types from each main plot was calculated.

6.2.2 Vegetation analyses

6.2.2.1 Species richness and diversity

While the number of individual sites sampled differed between the two forest types (six for the sacred groves and four for the non-sacred forests) the total area of forest sampled was the same for both (2.24 ha). Therefore, a single estimate of species richness and composition at this fixed area across the whole landscape was calculated for each of the two types to provide an initial gross comparison between them. The mean species richness of the six sacred groves and four non-sacred forests was compared using the Kruskal Wallis test (selected because of the unbalanced data). For each of the two forest types, area-based accumulation curves (the accumulation of species as a function of the sample plot/area) were generated using EstimateS (Colwell, 2010) with 100 iterations to see how they differ in species richness. The number of observed species per whole sample plot (seven in sacred groves and seven in non-sacred forests) were compared with the non-parametric estimators Chao 1 and Jacknife-2. These two estimators were chosen because they calculate species richness and give an estimate of absolute number of species in a community incorporating rare species (Magurran, 2004).

Species diversity in each sacred grove and non-sacred forest was further investigated by rarefaction analysis and Renyi’s diversity ordering to assess variability within sites. Renyi’s diversity ordering (sometimes known as the diversity profile) was calculated using the Species Diversity and Richness Analysis Package (Seaby and Henderson, 2007). Renyi’s diversity ordering values ($H\alpha$) are calculated from the proportional abundance of each species (abundance of each species/total abundances of species) and the scale parameter α ranges from zero to infinity (∞) (Kindt et al., 2004 in Robiglio, 2008). Renyi’s diversity ordering is based on the concept of entropy and is defined as $H\alpha = \frac{\log \sum_{i=1}^{s} pi^{\alpha}}{1-\alpha}$, where α is the ordering ($\alpha \geq 0$, $\alpha \neq 0$), pi is the proportional abundance of the i^{th} species, and log is the logarithm to base of e. Diversity orders that start at a higher level have higher richness. Orders that are higher at α= infinity have a lower proportional abundance of the dominant species (Robiglio, 2008). If two communities cross over, they are non-comparable. Such non-comparable communities occur when relative abundance and species richness are summarized using a single diversity index indicating that a single diversity index will not provide sufficient information about the given community (Kindt et al., 2004 in Robiglio, 2008). Diversity ordering such as Renyi's diversity profile offers a solution by identifying those communities that are consistent in their relative diversity.

6.2.2.2 Species abundance

The abundance of each tree species recorded in each sacred grove and non-sacred forest site was described by their stem density (number of stems/ha), dominance (basal area in m^2/ha) and frequency (the proportion of the sub-plots in each site that they were present in). The mean stem density/plot and basal area/plot were compared between the two forest types using paird T-test. The mean density of tree seedling of six sacred groves and four non-sacred forests, and mean sapling density of six sacred groves and four non-sacred forests also compared using Kruskal Wallis test (Community Analysis Package IV). The importance value of each tree species in each site was calculated from their values relative to the total for all species as the sum of relative dominance (relative basal area), relative density (stem density) and relative frequency.

Tree species were divided into DBH classes to investigate past dynamics and regeneration status. The data were sorted accordingly from the lowest to the highest range DBH. Trees ≥ 5 cm were divided into DBH classes of 5 cm interval. All individuals of tree species ≤ 1 m in height were termed seedlings whereas those with DBH < 5 cm and height > 1 m were termed saplings. For dominant species individual population DBH structure in each forest types was also explored.

All tree species present as individuals ≥ 5 cm DBH were classified into three functional groups: wide-habitat/generalist species, forest-habitat species and forest-margin/light - demanding forest species using a priori information on their distribution from Flora of

Ethiopia (Edward et al., 1995, 1997, 2000; Hedberg et al., 1989, 1995, 2003, 2006; chapter 5). The importance value (IVI) of each species within each functional group was calculated and the mean was used to compare the difference between the sacred and non-sacred forests.

6.2.2.3 Similarity of community composition

Plant community composition was analysed using multivariate techniques. The analysis-of-similarities (ANOSIM) permutation procedure was applied to test for differences in community composition between forest types (Clarke, 1993; Clarke and Warwick, 2001). ANOSIM uses the test statistic (R) based on the differences between the average of all the rank dissimilarities between species and between sites and the average of all the rank dissimilarities between species within the sites. It generates a value of R which is scaled to lie between -1 and +1, a value of zero indicates no difference among a set of samples while R values > 0 indicate that species are more dissimilar between sites than within sites (Clarke and Warwick, 2001). Negative R values indicate that dissimilarities within sites are greater than dissimilarities between the sites (Clarke and Warwick, 1994). In this study, R values < 0.5 indicate little difference and values > 0.5 indicate substantial difference. To test for significance, the ranked similarity between sites was compared with the similarity that would be generated by random chance.

The samples were randomly assigned to sites 1000 times and R was calculated for each permutation. A difference between communities was considered to be significant if the P-value was below 1% (0.01). To examine the species that contributed most to differences between sacred and non-sacred, similarity percentage (SIMPER) was used (Clarke and Warwick, 2001). SIMPER computes the percentage contribution of each species to the dissimilarity between all pairs of sample plots between sites and the percent contribution of each species to the similarities between all pairs of sample plots within the site.

The results of the analyses were visualized using Non-metric Multi-Dimensional Scaling (NMDS) ordination. Being non-parametric, NMDS was more suited than other ordination methods to the complexity of the uneven distribution of sample sub-plots between and within sites in this study. Ordination was performed because it expresses the similarity between sacred groves and non-sacred forests in a reduced number of dimensions by reducing a complex interrelationship to a simple Figure and provides a visual representation of how well a model accounts for variation in species composition between sites (Seaby and Henderson, 2007; Keleher and Rader, 2008). NMDS also calculates the stress (residual modeling error) as a rank function instead of absolute dissimilarity values (Jongman et al., 1995). NMDS was run using the Bray-Curtis (Curtis, 1951) dissimilarity matrix based on species presence/absence. The Bray-Curtis index gives less weight to outliers and is the recommended distance measure for NMDS (Southwood and Henderson, 2000). In NMDS two-dimensional ordination spaces, sample subplots are represented by a point and the distance between points show the degree of dissimilarity or similarity between the samples (Hahs and Mcdonnell, 2007). Canonical Correspondence Analysis (CCA) was used to relate the composition difference between the two forest types to environmental variables (Ter Braak and Smilauer, 2002). Environmental variables include altitude, disturbance and topographic data (Altamirano et al., 2010) such as slope and aspect.

6.2.2.4 Disturbance

The disturbance scores from grazing and for forest clearance for each sampled sub-plot (20 x 20 m) ranged from 0 (no disturbance) to 4 (highly disturbed). The values for grazing as well as forest clearance taken separately in a single sub-plot summed and divided into two to get cummulative score of disturbance for each eight sub-plots. Then, value for each sample sub-plot was summed and divided by eight to get an average score for each single sacred grove as well as each non-sacred forest. Correlation analyses were performed between disturbance and (a) the sum of seedling and sapling density, (b) tree density, (c) alpha diversity (Rao et al., 2011). The disturbance from tree cutting was also quantified separately and rated using the disturbance index

$= \frac{a}{a+b} \times 100$ (Bhuyan et al., 2003), where a is the total number of individual tree stumps ≥ 5 cm, and b is the total number of uncut trees ≥ 5 cm.

6.2.2.5 Biocultural diversity value of the forests

Biocultural diversity is described as the link between culture and biological diversity (Maffi and Woodley, 2010). The biocultural value of the studied sacred groves was qualitatively analyzed from interviews conducted with the custodians of the groves. I interviewed 24 custodians, four at each sacred grove, who are responsible for protecting the groves within their communities. They are all men. According to local traditions women are not allowed to be direct observants of the sacred grove but they participate in festivals, preparation of food and drinks for festival and liquors for sacrifice. I selected the custodians for interview on the assumption that they were the members of the community most knowledgeable about the groves. Group interviews were conducted with the custodians at each of the six sites, supplemented by individual interviews where this was agreed. These interviews took place near the sacred groves. The interviews were semi-structured and based around a pre-determined set of questions that were designed to gather information about the guardians' perception of the groves, their significance in both spiritual and material terms, the impact of the groves on their own lives and those of other community members, and their perception of the current threat to the groves (the questionnaire is in Appendix 6.4).

6.2.3 Land-use and land-cover data and analyses

Landsat Thematic Mapper (TM) images of an area of 183 x 183 km covering the majority of the Gamo highlands were obtained for the years 1995 and 2010. They gave a total sample area in the Gamo highlands of 66,765 ha. A maximum-likelihood classification of the data of each TM image was carried out to map land cover. The classification was aggregated into four cover classes of: cultivated land and settlement, forest, open pasture, and wooded grassland. Classification accuracy was verified to 78%

for 1995 and 85% for 2010 in ERDAS 9.1 (ERDAS, 1997) using ground control points (GCP) obtained during the field survey. The contour map and administrative boundary vector maps of the area produced by Ethiopian Map Agency were also used to clip the area of interest and to determine the position and ensure that the specific area covered by the two images was the same. The extent of land-cover types in each image and their reduction or gain in total area (1995-2010) in the whole 66,765 ha were evaluated by computing the proportion of each land-cover class (CA). Fragmentation was evaluated for each image by quantifying the number of patches (PN), mean patch size (MPS), mean shape index (MSI), edge density (ED), Shannon diversity (SDI) and Shannon evenness (SEI) using Patch Analyst 4.2 (Elkie et al., 1999) at landscape, class and patch levels for each year.

6.3 Results

6.3.1 Floristic composition of sacred groves and non-sacred forests

The land area of each of the six sampled sacred groves ranged from 0.72 to 33.6 ha (mean 8.5 ha, total area 58.8 ha) and for the four sampled non-sacred forests from 16.6 to 400 ha (mean 116.4 ha, total area 556.8 ha). A total of 152 species of plants belonging to 129 genera and 60 families were recorded in the six sacred groves. Recorded richness was marginally lower at the species and genus levels, but higher at the family level, in the non-sacred forests: 142 species belonging to 123 genera and 74 families (see Appendix 6.1 and 6.2 for the total species lists). The dominant families in terms of species number were similar between the two site types, with a greater abundance of Fabaceae species in the sacred groves and of Asteraceae in the non-sacred forests. In the sacred groves and non-sacred forests six and seven families, respectively, contained at least five species (Table 6.1). Fifty families in the sacred groves and 64 families in the non-sacred forest were represented by less than 5 species. Of the 152 plant species recorded in the sacred groves, 30.5% were tree species, 29.8% shrubs, 9.9% climbers, 25.2% terrestrial angiosperm forbs, 2.0% grasses, 2.0% angiosperm epiphytes and 0.7% geophytic ferns; in the non-sacred forests, of the 142 species recorded, 27.5% were trees, 31.7% shrubs, 9.9% climbers, 24.6% terrestrial angiosperm forbs, 1.4% grasses, 1.4% angiosperm epiphytes and 3.5% geophytic ferns. Summing these data, the woody species contributed a very similar 70.2% and 69.1% of the total species richness in the sacred groves and non-sacred forests respectively, though this was slightly more weighted to tree species in the sacred groves. The tree species with the highest frequency in the 56 20x20 m sample plots in both forests was *Syzygium guineense* (85.7% and 60.7% in the sacred groves and non-sacred forests respectively). Only one other species (*Ilex mitis*) occurred with a frequency > 50% in both forest categories.

Table 6.1 The dominant families and their number of species in sacred groves and non-sacred forests in Gamo Highlands, south-west Ethiopia.

Family	Number of species	
	In sacred grove	In non-sacred forest
Fabaceae	11	5
Lamiaceae	11	7
Asteraceae	9	13
Rubiaceae	8	6
Urticaceae	6	5
Acanthaceae	6	5
Euphorbiaceae	6	5
Celasteraceae	5	5

Nineteen and 13 species were found in the sacred groves and non-sacred forests respectively that are endemic to Ethiopia based on the work of Ensermu Kelbessa et al. (1994) and Vivero et al. (2005) (Table 6.2). The endemic species contributed a slightly higher proportion of the recorded flora of the sacred groves (12.5%) than of the non-sacred forests (9.0%). For each of the endemic species, the mean number of sites present in four sacred groves was 2.9 (out of 6) and for non-sacred forest was 1.3 (out of 4) (Table 6.2). All 13 of the endemic species present in the non-sacred forests were also present in the sacred-groves, leaving six endemic species only recorded in the sacred groves (though in the case of *Erythrina brucei* it was also recorded in non-sacred land managed as agroforestry systems). Of the 19 endemic species, the IUCN red list (Vivero et al 2005) places four in the "near threatened" category: *Acanthus sennii, Spermania macrocarpa, Maytenus addat* and *Crotalaria rosenii* (the latter two being recorded only in the sacred groves). Another non-endemic species that was present in both forest types was also recorded in the IUCN red list, *Prunus africana*. In addition there were two species identified as national priorities for conservation (*Cordia africana* and *Hagenia* *abyssinica*) (Bekele and Berhanu, 2001.) both of which were only recorded in the sacred groves.

Table: 6.2 Endemic species recorded in six sacred groves and four non-sacred forests in the Gamo Highlands of Ethiopia and their conservation status.

	Species	Family	Life form[1]	No. of sacred grove sites recorded in	No. of non-sacred sites recorded in	Conservation status[2]
1	*Acanthus sennii*	Acanthaceae	S	3	1	NT
2	*Kniphofia foliosa*	Asphodelaceae	H	1	1	DD
3	*Erythrina brucei*	Fabaceae	T	4	0	LC
4	*Maytenus addat*	Celastraceae	T	1	0	NT
5	*Millettia ferruginea*	Fabaceae	T	3	2	LC
6	*Acacia abyssinica sub sp.abyssinica*	Fabaceae	T	3	0	DD
7	*Crotalaria rosenii*	Fabaceae	S	4	0	NT
8	*Plectocephalus varians*	Lamiaceae	S	6	1	DD
9	*Solanecio gigas*	Asteraceae	S	4	3	LC
10	*Urtica simensis*	Urticaceae	H	6	0	DD
11	*Vepris dainellii*	Rutaceae	T	3	3	LC
12	*Pentas caffensis*	Rubiaceae	H	1	2	DD
13	*Spermania macrocarpa*	Malvaceae	S	4	2	NT
14	*Vernonia leopoldi*	Asteraceae	S	2	1	LC
15	*Lippia adoensis*	Lamiaceae	S	3	0	LC
16	*Bothriocline schimperi*	Asyeraceae	S	4	3	LC
17	*Vernonia rueppellii*	Asteraceae	S	1	1	LC
18	*Justicea diclipteroides*	Asteraceae	H	1	3	DD
19	*Phylanthus mooney*	Euphorbiaceae	H	1	2	DD
Mean number of sites present in				2.9	1.3	

[1] T, tree; S, shrub; H, herb.[2]NT, near threatened; LC, least concern; DD, data deficient (Source IUCN, 2006).

6.3.2 Species composition similarity

The ANOSIM test based on Bray-Curtis similarity using 1000 permutations at the 0.01% significance level showed that there was a significant difference in total species composition between the sacred groves and non-sacred forests (R = 0.61, p = 0.001). This was due to differences in the abundance of a range of species (29 species were responsible for 69.6% of variation between sites, Table 6.3). The presence of species in large abundance in sacred groves, such as *Syzygium guineense, Bersama abyssinica, Maytenus undata, Prunus africana, Hagenia abyssinica* and *Cordia africana,* was the major contributor to the difference. Species that were more abundant in the sacred groves were predominatly light-demanding and wide-habitat in distribution whereas those more abundant in the non-sacred forests were predominatly forest-habitat and wide-habitat in distribution (Table 6.3, 6.4). However, the mean IVI of the wide-habitat and forest-habitat distribution species functional groups did not differ significantly between the two habitat types (t = -0.68, p > 0.05; t = -0.69, p > 0.05 respectively). The forest-margin species had higher abundance in sacred groves than non-sacred forests but this difference was just not significant (t = 2.069, p = 0.06).

Differences between the two categories (sacred groves and non-sacred forests) accounted for a much greater variation in species composition amongst the 10 studied sites than did differences within each category (within sacred groves as well as within non-sacred forests). The average dissimilarity between sacred groves and non-sacred forests was 69.6% whereas the average dissimilarity amongst the six sacred groves was 42.3%, and amongst the four non-sacred forests was 46.3% (from SIMPER analysis).

Table: 6.3 Species more abundant in sacred grove and non-sacred forest, in the Gamo Highlands of Ethiopia, assessed by a similarity percentage (SIMPER). For species with higher abundance in one site type, this Site Is indicated by highlighting. For each species the average dissimilarity between the forest categories, and its percentage contribution to the overall variation in species composition between them is also shown.

	Sacred groves	Non-sacred forests		
Name	Average abundance	Average abundance	Average Dissimilarity	% Contribution
Albizia gummifera	0.1	0.0	0.2	0.3
Allophylus abyssinicus	0.4	0.6	0.8	1.2
Apodytes dimidiata	0.7	0.6	0.8	1.1
Brucea antidysenteria	0.3	0.0	0.6	0.8
Canthium oligocarpum	0.4	0.5	0.8	1.2
Celtis africana	0.4	0.2	0.7	1.0
Dracaena fragrans	0.2	0.7	1.0	1.4
Ekebergia capensis	0.6	0.4	0.8	1.2
Embelia schimperi	0.5	0.8	0.8	1.2
Euphorbia candelabrum	0.4	0.0	0.6	0.8
Hippocratea goetzei	0.4	0.1	0.6	0.9
Jasminum abyssinicum	0.4	0.7	0.9	1.2
Lepidotrichilia volkensii	0.3	0.6	0.9	1.2
Maytenus addat	0.6	0.3	0.9	1.2
Myrsin africana	0.0	0.7	1.1	1.6
Ocimum lamifolium	0.4	0.0	0.7	1.1
Olea capensis subsp macrocarpa	0.0	0.6	1.0	1.4
Olea welwitschii	0.2	0.0	0.3	0.5
Oxyanthus lepidus	0.8	0.4	0.9	1.4
Psychotria orophilia	0.3	0.8	1.1	1.5
Pouteria adolfi-friedericii	0.0	0.2	0.3	0.4
Prunus africana	0.3	0.2	0.6	0.9
Rhamnus prinoides	0.2	0.1	0.4	0.6
Schefflera abyssinica	0.2	0.2	0.5	0.7
Solanecio gigas	0.0	0.2	0.3	0.4
Syzygium guineense	0.6	0.9	0.7	1.0
Teclea nobilis	0.0	0.4	0.7	1.0
Vepris dainellii	0.3	0.6	0.8	1.2

Table 6.4 Abundance (importance value index, IVI) of trees ≥ 5 cm DBH of each species classified into three different function groups in sacred groves and non-sacred forests in the Gamo Highlands of Ethiopia.

Functional group	IVI		Functional group	IVI	
Wide-habitat/ generalist species	Sacred grove	Non-sacred forest	Forest-habitat species	Sacred grove	Non-sacred forest
Syzygium guineense	20.86	27.08	*Ilex mitis*	12.36	9.76
Schefflera abyssinica	4.28	4.33	*Macaranga capensis*	9.07	3.25
Dracaena steudneri	2.27	0	*Olea capensis*	3.25	8.49
Allophylus abyssinicus	2.17	1.21	*Galiniera saxifraga*	4.63	3.41
Prunus africana	1.81	1.40	*Apodytes dimidata*	9.34	1.49
Ficus sur	1.72	3.07	*Aguaria saxifolia*	0	6.86
Ficus vasta	0.48	0	*Lepidotrichilia volkensii*	0.39	6.17
Croton macrostachyus	1.66	0.64	*Vepris dainellii*	1.09	1.32
Bersama abyssinica	0.92	1.37	*Schefflera volkensii*	0	0.64
Flacourtia indica	0	0.46	*Canthium oligocarpum*	0.8?0	1.4?0
Pouteria adolfi-frederici	0	0.30	*Chionanthus mildbraedii*	0	1.58
Polyscias fulva	0	0.35	*Dracaena fragrans*	0	4.71
Albizia gummifera	0.18	0	*Olinia rochetiana*	0	3.33
Celtis africana	1.38	0	*Ficus palmata*	0.28	0
Dombeya torrida	0.37	0.15	*Pittosporum viridiflorum*	0.35	0.15
Scherebra alata	0.36	0	Mean (±SE)	2.8±2.5	3.5±1.6
Myrica salicifolia	0	0.34			
Millettia ferruginea	0.11	0			
Mean (±SE)	2.1±3.3	2.3±0.9			

Forest-margin/Light demanding	Sacred grove	Non-sacred forest		Sacred grove	Non-sacred forest
Maytenus undata	6.24	1.07	*Nuxia congesta*	0.4?0	1.18
Euphorbia candelabrum	5.67	0	*Dovyalis abyssinica*	0.3?0	0
Maytenus arbutifolia	2.31	0.17	*Dracaena steudneri*	2.27	0
Ekbergia capensis	1.59	1.82	*Maytenus addat*	0.58	2.08
Maesa lanceolata	0.88	0.15	*Erytrina brucei*	1.18	0
Cordia africana	0.64	0	*Albiziz schimperiana*	0.18	0
			Mean (±SE)	1.9±1.5	0.5±1.1

6.3.3 Ordination

The NMDS ordination also showed a very clear separation of species composition between the sacred groves and non-sacred forests with no overlap of their clusters of individual sample plots (Figure 6.1).

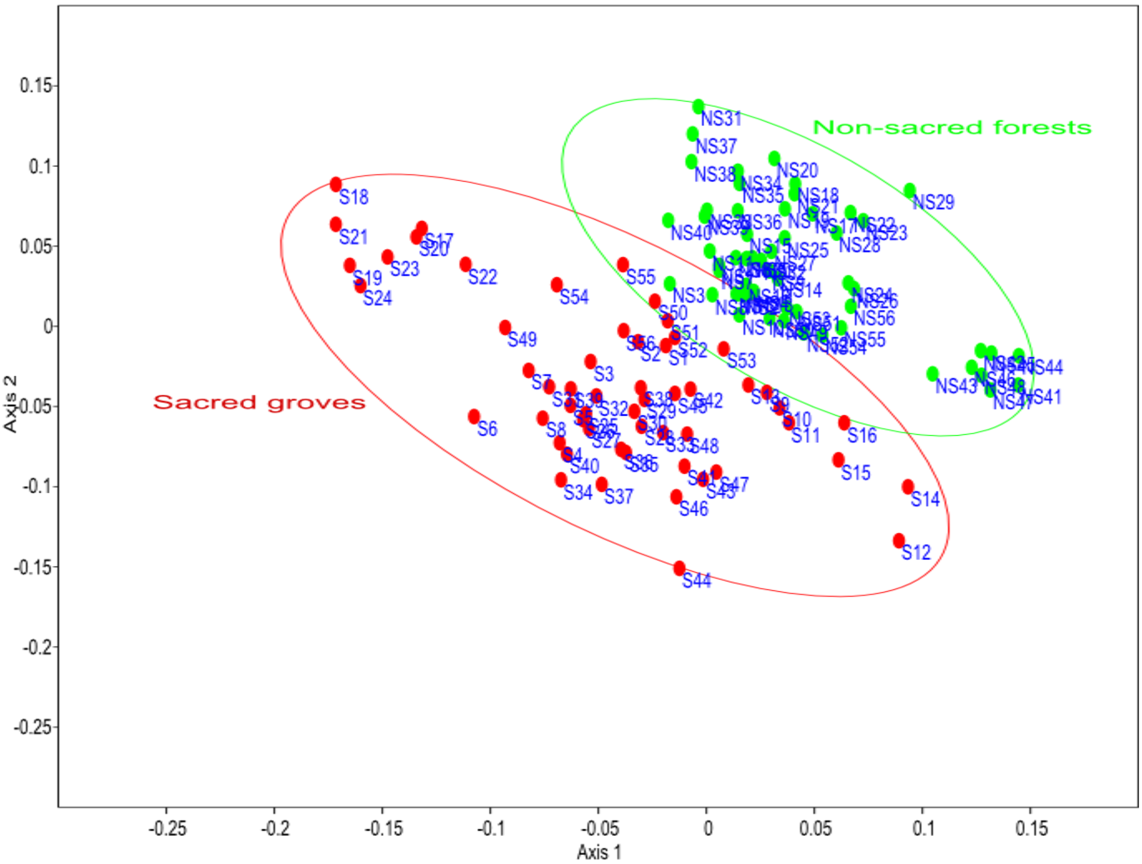


Figure 6.1: Non-metric multidimensional scaling (MDS) ordination of species composition, based on two dimensions with Bray-Curtis dissimilarities using species presence/absence data recorded in each of 56 20x20 m sample plots in sacred groves and non-sacred forests respectively in the Gamo Highlands of Ethiopia. The two-dimensional stress was 0.226. The plots in the sacred groves are coloured red and in the non-sacred forests in green. The sample plots were arranged in sets of eight within each sample unit, thus plots with numbers 1-8 were in the first sample unit, 9-16 in the second etc. The plots in each of the two site types are clustered separately.

Ordination with environmental variables performed using canonical correspondence analysis (CCA) (TerBraak and Smilauer, 2002) produced a first axis which explains 35.59% of the variation and a second axis which explains 31.38% (Figure 6.2,

Appendix 6.3). The third and fourth axes did not show significant in variation (P > 0.05).

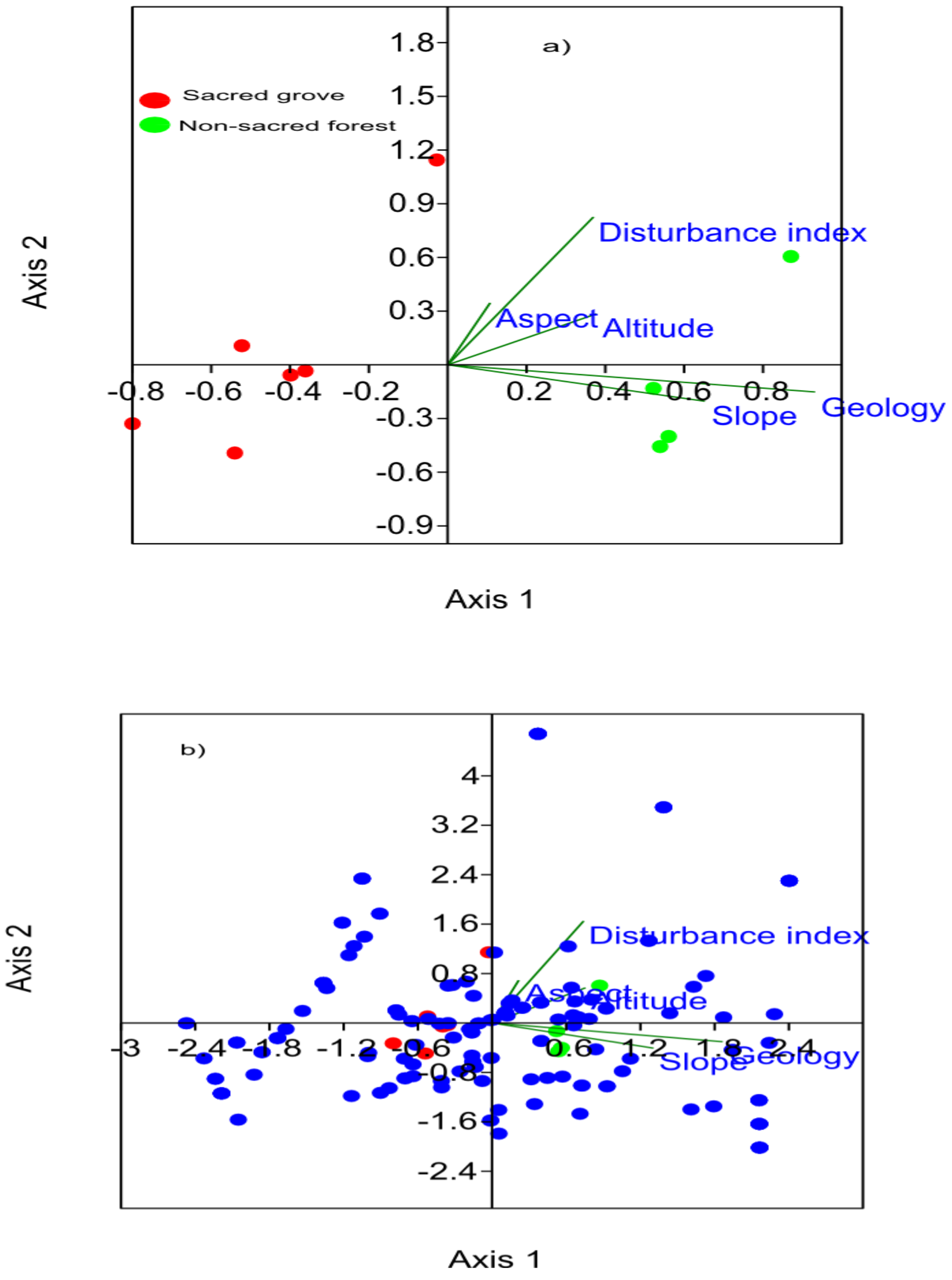


Figure 6.2 CCA ordination plots of (a) 10 sites, and (b) plant species. Environmental variables shown as vectors are geology, disturbance, slope, altitude and geographical aspect. Red circles show the six sacred groves, green circles show the four non-sacred forests and blue circles in (b) show plant species.

The first axis (Eigenvalue 0.28) mainly represented geology, slope and altitude. The second axis (Eigenvalue 0.25) was most strongly linked with disturbance and aspect. The non-sacred forests were primarily split from the sacred groves by axis 1, though one sacred grove (Gufae) and one non-sacred forest (Dhule) were separated from the other eight sites by axis 2. Non-sacred forests were associated with higher values of slope and altitude, and a different geology, than the sacred groves. There was a weak association between the single sacred grove and non-sacred forest sites with high axis 2 scores and high disturbance index. The species most strongly associated with the non-sacred forests and their high values of slope and aspect (and different) geology were *Schfflera volkensii*, *Aguaria saxifolia*, *Lepidotrichilia volkensii*, *Dracaena fragrans, Olea capensis subsp. macrocarpa*, *Olinia rochetiana* and *Polyscias fulva*.

6.3.4 Species diversity and richness

The Renyi diversity profile showed considerable variation amongst the individual sample sites within as well as between the two forest types (Figure 6.3). The shapes of the curves for all ten sites were similar indicating a comparable relationship between richness and evenness, though the shallower curve for the Oche non-sacred forest indicates that it has the highest evenness. For approximately half of the individual sites within each category the diversity curves for $\alpha > 0$ did not overlap, indicating a clear hierarchy in their relative diversity (in terms of both richness and evenness). However, within the sacred groves the curves of three of the six sites overlapped with each other showing that there is no overall consistent difference in their diversities: for Osha-Ocha and Gufae at α=0.5, Osha-Ocha and Tele at α=0.8, and Tele and Gufae at α=1 (Figure 6.3a). Richness (the y-intercept) was very similar for Osha-Ocha and Gufae. For the non-sacred forests none of the four curves intersected, however for Dhule and Oche they were very close over the range of α between 0 and 0.5 indicating a similar richness and limited confidence in there being an overall difference in their diversity (Figure 6.3b). The overall comparison between the two categories of site shows a small but consistently higher diversity of the sacred groves than the non-sacred forests, which is largely based on a difference in richness rather than evenness (Figure 6.3c).

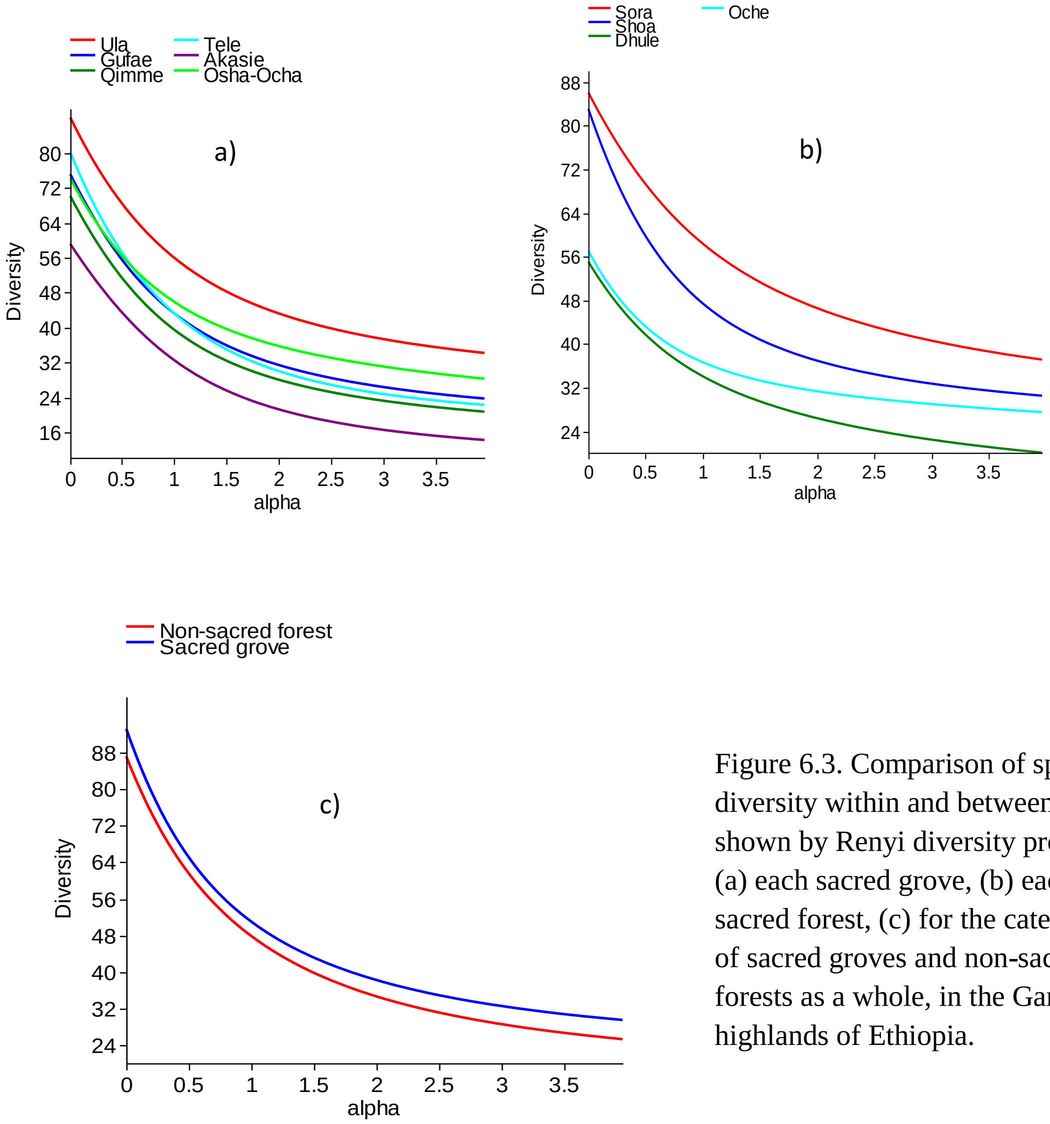


Figure 6.3. Comparison of species diversity within and between sites shown by Renyi diversity profiles in (a) each sacred grove, (b) each non-sacred forest, (c) for the categories of sacred groves and non-sacred forests as a whole, in the Gamo highlands of Ethiopia.

The total number of species present in the sites of each category was estimated from accumulation curves: observed number of species (using Mao-Tau), Chao 1 and jackknife 2 (Colwell, 2010). Lumping together the data from the sample plots in the six sacred groves (Figure 6.4a) as well as four non-sacred forest (Figure 6.4b) sites produced landscape-scale estimates for each forest type. Species richness was higher in the sacred groves than in the non-sacred forests using all three indices: with Mao-Tau it was observed as 152.0±4.7 for the sacred groves and 142.0±4.5 for non-sacred forests, with Chao 1 it was 154.6±3.2 and 145.8±2.6 respectively, and with jackknife 2 it was 176.5±6.1 and 165.6±5.2 respectively. The ratio of observed (Mao-Tau) to estimated

(Chao 1) species richness was 97.6% for the sacred groves and 98% for the non-sacred forests suggesting that at least 2.4% more species in sacred groves and 2% more species in the non-sacred forest are expected to be present than were observed.

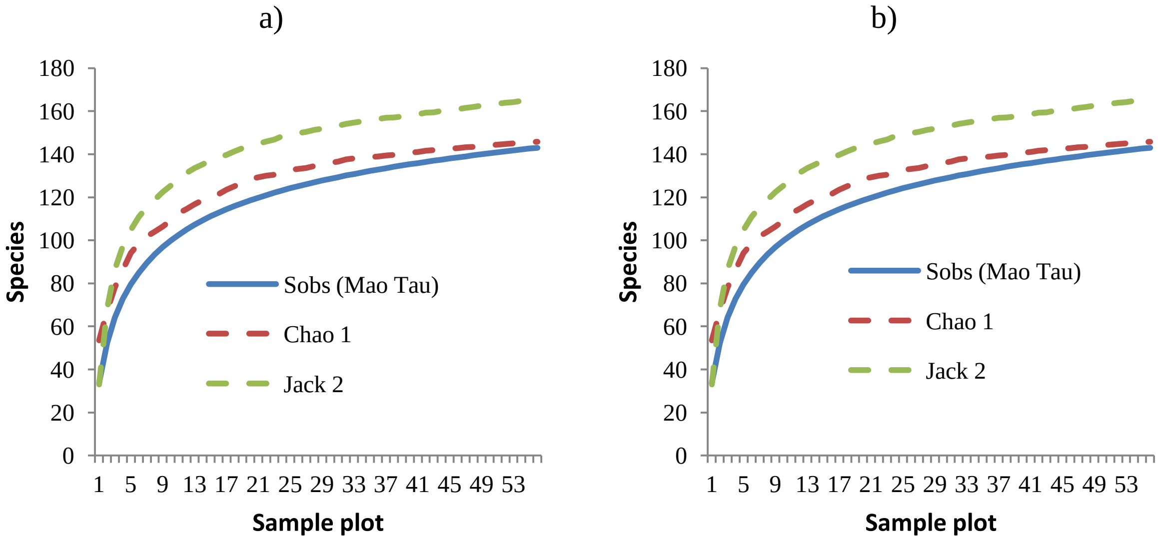


Figure 6.4 Species accumulation by sample plot, based on observations (Mao Tau method), and Chao 1 and Jacknife 2 estimators in (a) sacred groves and (b) non-sacred forests in the Gamo Highlands of Ethiopia. The x-axis labels show the number of plots and the y axis the number of species.

Species accumulation by all individual sampled plants within each of the ten sites (relating to alpha diversity) assessed using the EstimateS rarefaction technique (Colwell, 2010) showed considerable variation amongst the sites (Figure 6.4). This variation matched well with the richness values revealed by the Renyi curves (Figure 6.3): four of the sacred groves had a very similar rate of accumulation whereas it was greater for Ula, and less for Akasi (Figure 6.5a); the non-sacred forests divided into two pairs with greater accumulation in Shoa and Sora and less in Dhule and Oche (Figure 6.5b). Of the ten sites, the curves of only Shoa and Sora approached an asymptote, for the others a noTable proportion of the species likely to be present had not been recorded even after the sampling of up to 500-1300 individual plants per site, supporting the result from Chao 1 and Jackknife 2 estimates for the two categories of forest as a whole (Figure 6.4).

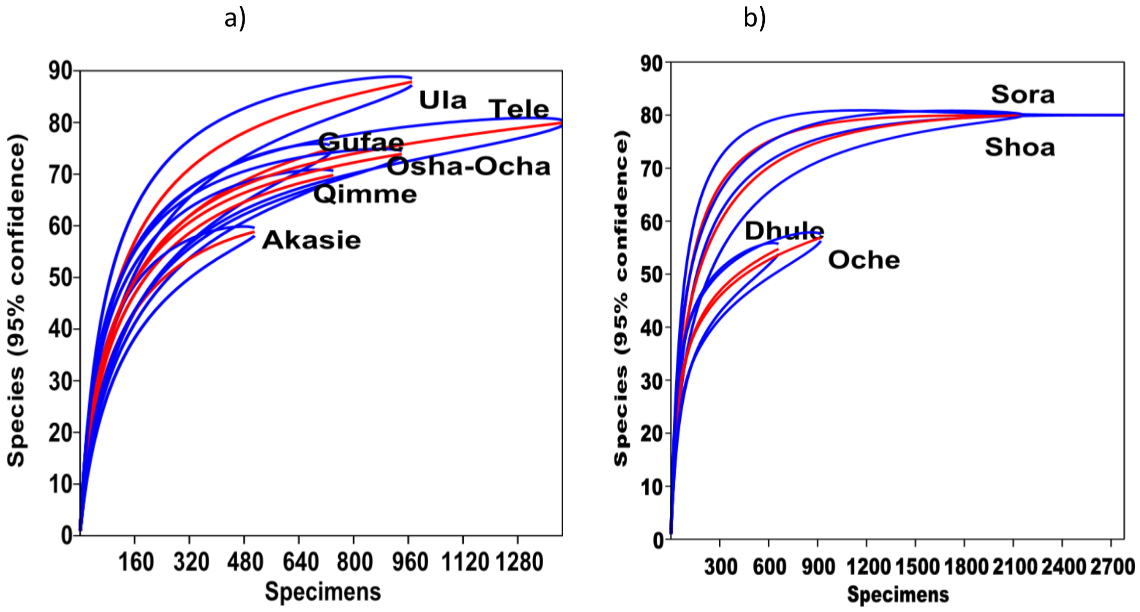


Figure 6.5 Individual plant-based rarefaction curves for ten sampled forests in the Gamo highlands of Ethiopia: (a) six sacred groves, (b) four non-sacred forests. The red line is the mean curve and the upper and lower lines (blue) for each accumulation curve indicate confidence limits at 95%.

6.3.5 Structural composition

A total of 876 and 944 individual trees $\geq$ 5cm DBH were recorded in the sacred groves and non-sacred forests in 56 plots respectively. The absolute stem density calculated for the plots in each forest type ($n = 56$) was higher in the non-sacred forests (421.4 ha^{-1}) than in the sacred groves (350.9 ha^{-1}) however the mean (±SE) of individual trees/plot in non-sacred forest (12.04±2.98) was not significantly different than sacred groves (10.03±2.09) ($t = 0.42$, $p > 0.05$). The absolute basal area of trees $\geq$ 5 cm DBH in each forest type was also calculated: sum of basal area for sacred grove was 54.4 m^{2} ha^{-1} and 44.6 m^{2} ha^{-1} for non-sacred forest respectively. The mean basal area of all trees $\geq$ 5 cm DBH/plot in each forest types ($n=56$) was greater in the sacred groves (0.97±0.37) than in the non-sacred forests (0.79±0.34) ($t = 2.98$, $p < 0.05$).

The basal area in both forest categories was dominated by a few tree species, with *Syzygium guineense* being overwhelmingly dominant in both forest types (Table 6.5).

There was considerable similarity in the composition of dominant species between the forest categories though *Apodytes dimidata, Macaranga capensis, Euphorbia candelabrum, Prunus africana Maytenus arbutifolia* and *Erytrina brucei* were noTable for having > 2.5 times greater basal area in the sacred groves than in the non-sacred forests, while comprising > 1.5% of the total basal area in the sacred groves. On the other hand, *Olea capensis, Ekbergia capense, Aguaria saxifolia, Olinia rochetiana* and *Maytenus addat* had > 2 times greater basal area in the non-sacred forests than the sacred groves, while comprising > 1.5% of the total basal area in the non-sacred forests.

Large trees are a good indicator of the structural condition and disturbance history of forests (Spies, 1998). In the 2.24 ha of sample plot area in each forest category there were 53 trees with a basal area ≥ 1 m^2 in the sacred groves and 32 in the non-sacred forests (a high density of 24 and 14 ha^{-1} , accounting for 31.4% and 27.3% of all trees, and comprising 10 and 9 species respectively). Large trees were found in all sites. This indicates that a high percentage of total basal area in the sacred groves and in non-sacred forests was contributed by a few dominant species.

Table 6.5: Basal area (m^2 ha^{-1}) by species of all trees with DBH ≥ 5 cm in six sacred groves and four non-sacred forests in the Gamo highlands of Ethiopia.

Sacred groves		Non-sacred forests	
Species	Basal area (m^2 ha^{-1})	Species	Basal area (m^2 ha^{-1})
Syzygium guineense	18.18	*Syzygium guineense*	17.62
Apodytes dimidiata	6.65	*Schefflera abyssinica*	5.39
Ilex mitis	6.49	*Ilex mitis*	3.85
Schefflera abyssinica	4.93	*Olea capensis*	3.54
Macaranga capensis	3.62	*Ficus sur*	3.02
Euphorbia candelabrum	2.53	*Ekbergia capensis*	1.65
Ficus sur	1.95	*Aguaria saxifolia*	1.50
Olea capensis	1.73	*Olinia rochetiana*	1.46
Prunus africana	1.48	*Maytenus addat*	1.06
Maytenus arbutifolia	1.24	*Macaranga capensis*	0.88
Erytrina brucei	0.93	*Lepidotrichilia volkensii*	0.59
Galiniera saxifraga	0.69	*Dracaena fragrans*	0.57
Allophylus abyssinicus	0.68	*Apodytes dimidiata*	0.56
Ekbergia capensis	0.56	*Prunus africana*	0.55
Croton macrostachyus	0.53	*Allophylus abyssinicus*	0.45
Maytenus addat	0.39	*Vepris dainellii*	0.39
Maytenus undata	0.38	*Galiniera saxifraga*	0.28
Cordia africana	0.27	*Polyscias fulva*	0.27
Dracaena steudneri	0.22	*Maytenus undata*	0.19
Ficus vasta	0.22	*Bersama abyssinica*	0.18
Hagenia abyssinica	0.22	*Schefflera volkensii*	0.17
Celtis africana	0.15	*Nuxia congesta*	0.15
Nuxia congesta	0.09	*Croton macrostachyus*	0.07
Canthium oligocarpum	0.05	*Chionanthus mildbraedii*	0.06
Dombeya torrida	0.05	*Myrica saxifolia*	0.05
Vepris dainellii	0.03	*Canthium oligocarpa*	0.05
Maesa lanceolata	0.03	*Flacourtia indica*	0.03
Ficus palmata	0.03	*Maytenus arbutifolia*	0.02
Schrebera alata	0.03	*Pouteria adolfi-frederici*	0.01
Bersama abyssinica	0.02	*Dombeya torrida*	0.01
Millettia ferruginea	0.02	*Dovyalis abyssinica*	0.01
Lepidotrichilia volkensii	0.01	*Maesa lanceolata*	0.001
Pittosporum viridiflorum	0.01	*Pittosporum viridiflorum*	0.001
Albizia gummifera	0.01		
Albiziz schimperiana	0.01		
Cupressus lucitanica	0.01		
Total	54.44	Total	44.63

Combining relative dominance, relative frequency and relative density into an importance value index indicated that *Suzygium guineenses, Ilex mitis, Apodytes dimidata, Macaranga capensis, Maytenus undata* and *Euphorbia candelabrum* had > 1.5 times greater abundance in the sacred groves than the non-sacred forests, while comprising > 1.5% of the total importance value in the sacred groves (Table 6.6). On the other hand, *Olea capensis, Aguaria saxifolia, Lepidotrichilia volkensii, Dracaena* *fragrans, Scheflera abyssinica* and *Galineria saxfiraga* had > 2 times greater abundance in the non-sacred forest than the sacred groves (Table 6.7).

Table 6.6 Importance value index (IVI) of tree species in six sacred groves in the Gamo highlands of Ethiopia, calculated as (relative dominance + relative frequency + relative density)/3.

Species	Relative dominance	Relative frequency	Relative density	IVI
Syzygium guineense	33.53	13.28	15.78	20.86
Ilex mitis	11.97	12.5	12.6	12.36
Apodytes dimidiata	12.26	10.55	5.22	9.34
Macaranga capensis	6.68	7.42	13.11	9.07
Maytenus undata	0.71	3.13	14.89	6.24
Euphorbia candelabrum	4.67	5.47	6.87	5.67
Galiniera saxifraga	1.27	6.64	5.98	4.63
Schefflera abyssinica	9.09	2.73	1.02	4.28
Olea capensis	3.19	2.74	3.82	3.25
Maytenus arbutifolia	2.29	3.12	1.53	2.31
Dracaena steudneri	0.41	1.95	4.45	2.27
Allophylus abyssinicus	1.25	3.48	1.78	2.17
Prunus africana	2.73	1.95	0.76	1.81
Ficus sur	3.6	1.17	0.38	1.72
Croton macrostachyus	0.98	2.73	1.27	1.66
Ekbergia capensis	1.03	2.73	1.02	1.59
Celtis africana	0.28	2.34	1.53	1.38
Erytrina brucei	1.72	1.17	0.64	1.18
Vepris dainellii	0.05	1.95	1.27	1.09
Bersama abyssinica	0.04	1.95	0.76	0.92
Maesa lanceolata	0.05	1.56	1.02	0.88
Canthium oligocarpum	0.09	1.56	0.76	0.80
Cordia africana	0.5	0.78	0.64	0.64
Maytenus addat	0.71	0.78	0.25	0.58
Ficus vasta	0.41	0.78	0.25	0.48
Nuxia congesta	0.16	0.78	0.25	0.40
Lepidotrichilia volkensii	0.02	0.78	0.38	0.39
Dombeya torrida	0.09	0.78	0.25	0.37
Schrebera alata	0.05	0.78	0.25	0.36
Pittosporum viridiflorum	0.02	0.78	0.25	0.35
Ficus palmata	0.05	0.4	0.38	0.28
Albizia gummifera	0.02	0.4	0.13	0.18
Albiziz schimperiana	0.02	0.4	0.13	0.18
Cupressus lucitanica	0.02	0.4	0.13	0.18
Millettia ferruginea	0.04	0.04	0.25	0.11
Total	100	100	100	100

Table: 6.7 Importance value index (IVI) of tree species in four non-sacred forests in the Gamo highlands of Ethiopia, calculated as (relative dominance + relative frequency + relative density)/3.

Species	Relative dominance	Relative frequency	Relative density	IVI
Syzygium guineense	39.49	16.22	25.53	27.08
Ilex mitis	8.63	10.47	10.17	9.76
Olea capensis	7.95	6.08	11.44	8.49
Aguaria saxifolia	3.36	3.04	14.19	6.86
Lepidotrichilia volkensii	1.31	9.46	7.73	6.17
Dracaena fragrans	1.28	5.74	7.1	4.71
Schefflera abyssinica	12.09	0.68	0.21	4.33
Galiniera saxifraga	0.62	6.42	3.18	3.41
Olinia rochetiana	3.27	4.06	2.65	3.33
Macaranga capensis	1.96	4.39	3.39	3.25
Ficus sur	6.77	1.69	0.74	3.07
Maytenus addat	2.37	2.7	1.06	2.04
Ekbergia capensis	3.70	1.35	0.42	1.82
Chionanthus mildbraedii	0.14	2.7	1.91	1.58
Apodytes dimidiata	1.25	2.36	0.85	1.49
Canthium oligocarpa	0.11	3.04	1.06	1.40
Prunus africanus	1.23	1.69	1.27	1.40
Bersama abyssinica	0.39	2.36	1.37	1.37
Vepris dainellii	0.87	2.36	0.74	1.32
Allophylus abyssinicus	1.00	1.69	0.95	1.21
Nuxia congesta	0.34	2.36	0.85	1.18
Maytenus undata	0.44	2.03	0.74	1.07
Croton macrostachyus	0.16	1.35	0.42	0.64
Schefflera volkensii	0.38	1.01	0.53	0.64
Flacourtia indica	0.06	1.01	0.32	0.46
Polyscias fulva	0.61	0.34	0.11	0.35
Myrica saxifolia	0.12	0.68	0.21	0.34
Pouteria adolfi-frederici	0.02	0.68	0.21	0.30
Dovyalis abyssinica	0.01	0.68	0.21	0.30
Maytenus arbutifolia	0.05	0.34	0.11	0.17
Dombeya torrida	0.01	0.34	0.11	0.15
Maesa lanceolata	0.00	0.34	0.11	0.15
Pittosporum viridiflorum	0.00	0.34	0.11	0.15
Total	100	100	100	100

6.3.6 Tree size-class structure

A single diameter size-class distribution for all individual trees ≥ 5 cm DBH was drawn for the sacred groves and for the non-sacred forests (Figure 6.6a). An indication of the density of smaller individuals of trees was added as a 0-5 cm DBH class comprising all measured stems > 0 cm tall and < 5 cm DBH. The seedlings and saplings were recorded in 5 x 5 m sub-subplots and the values were multiplied by 16 to get them on the same area basis as the ≥ 5 cm DBH stems measured in the 20 x 20 m plots. The overall size-class structure of all tree species was similar in both forest categories, a successive decrease in density from 0 to 15 cm DBH and very low density in each class above that (Figure 6.6a). However, while the density of stems < 5 cm DBH in the sacred groves was 2.75 times than in the non-sacred forests, the density of stems in the 5-10 and 10-15 cm DBH classes was higher in the non-sacred forests than the sacred groves. If this reflects an equilibrium situation it implies much greater mortality rates over this size range in the sacred-groves. However, an alternative non-equilibrium explanation is that there has been a recent increase in seedling/sapling recruitment in the sacred groves that has not occurred in the non-sacred forests. The density of trees ≥ 50 cm DBH was twice as great in the sacred groves (76 ha^{-1}) as in the non-sacred forests (35 ha^{-1}).

Size class distributions were assessed for tree height (Figure 6.6b). Comparison with the diameter class distributions is complicated by the effect of using different size class intervals. Nonetheless, it is clear that both forest categories also show a successive negative decline in tree density with height but the rate of decline up to 10 m height is less for the non-sacred forests than the sacred groves, though for both the rate of decline appears less steep for height than for diameter.

For the four most abundant tree species in each forest category the population DBH size-class distributions show a wide range of shapes (Figure 6.7 a-d). For the two species in common between the forests categories both *Ilex mitis* and *Syzygium guineense* show a higher density in the 0-4.9 cm DBH class followed by a steeper decline in numbers with size in the sacred groves than the non-sacred forests. For *I. mitis* density over 5-24.9 cm DBH is greater in the non-sacred forests, but then density > 25 cm DBH is greater in the sacred groves. The distributions for the other two abundant species in the sacred-groves is very similar to *S. guineense*, whereas in the non-sacred forests, the other two species show a similar distribution to *I. mitis* (though for *Aguaria saxifolia* the density of 0-4.9 cm DBH is less than that in the next two classes indicating some restriction in its seedling/sapling recruitment or survival rate.

In the sacred groves, for *Macaranga capensis*, the density increased from 5-9.9 cm DBH to 20-24.9 cm. The density then decreased between 25 and 49.9 cm DBH while the population of *Apodytes dimidata* was dominated by seedlings/saplings and by trees ≥ 50 cm DBH. In the non-sacred forests, the density of *Aguaria saxifolia* increased from seedlings and saplings to the 5-9.9 cm DBH classes and then sharply decreased after that while *Olea capensis subsp. macrocarpa* declined in density from seedlings and saplings over the whole size range.

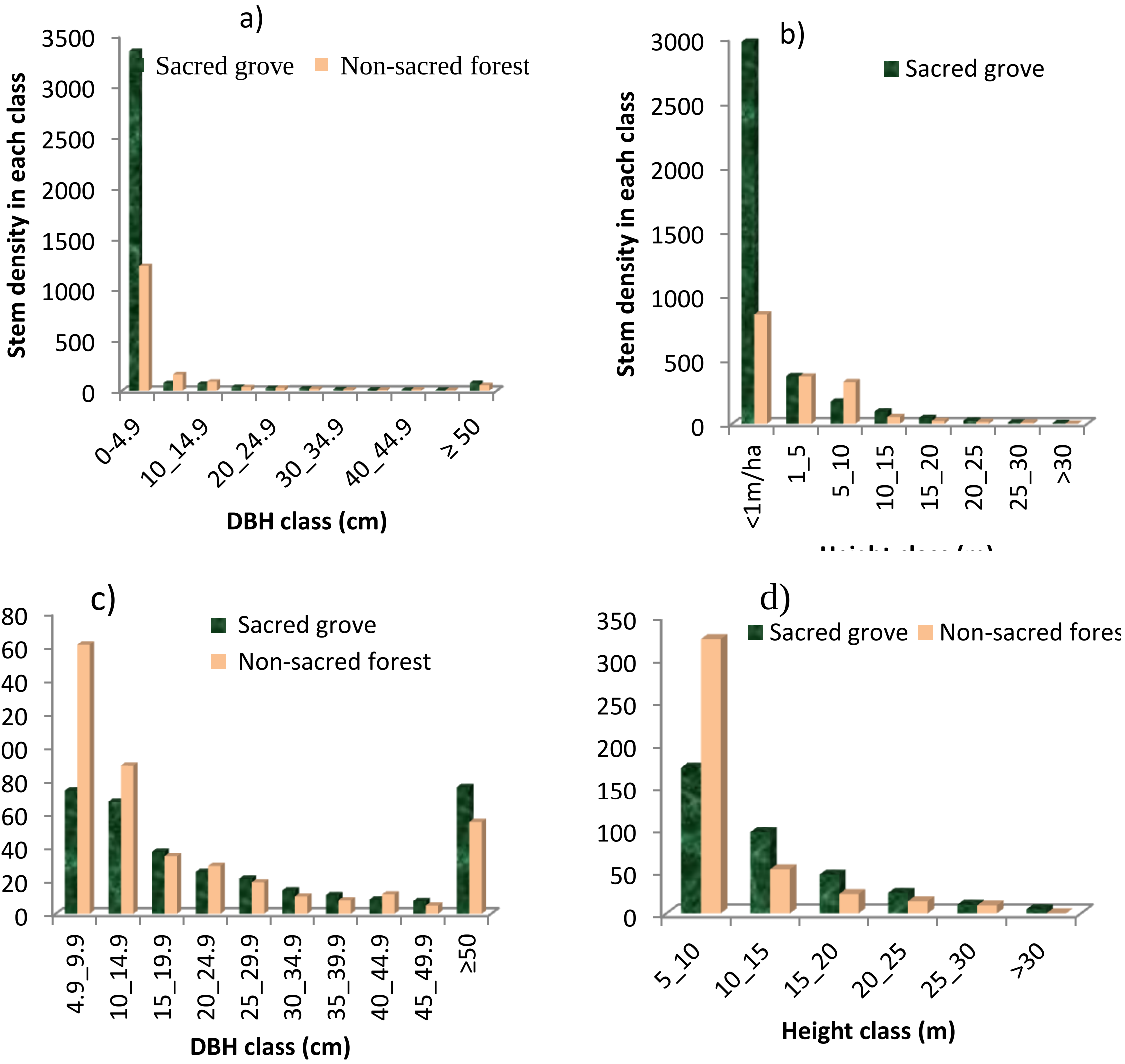
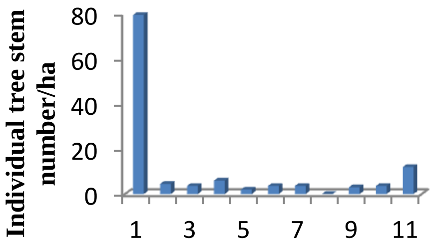


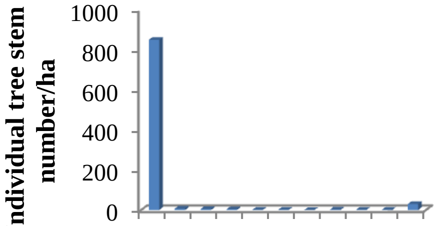
Figure 6.6. Individual tree size class distribuitions for all species in sacred groves and non-sacred forests in the Gamo Highlands of Ethiopia: (a) by DBH class, (b) by height class. For DBH the 0-5 cm class includes all individuals of tree species > 0 cm tall and < 5 cm DBH. For height the smallest class (0-5 m) has been divided into two sub-classes 0-1 m (which includes all individuals of seedlings of tree species > 0 cm and < 100 cm tall) and saplings of height 1-5m. (c) and (d) show the distribution of only the individuals in DBH class ≥ 5 cm and height class ≥5 m respectively.

(a) Sacred groves

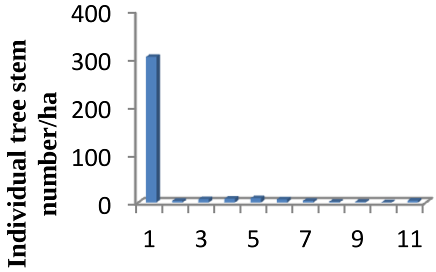
Ilex mitis



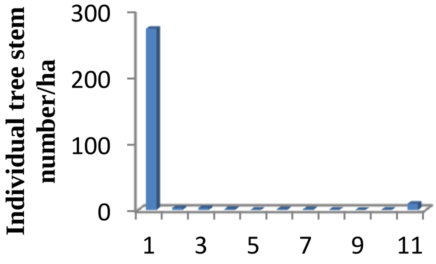
Syzygium guineense



Macaranga capensis

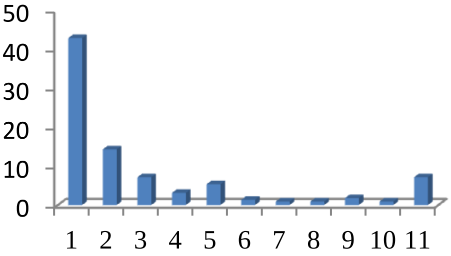


Apodytes dimidiata

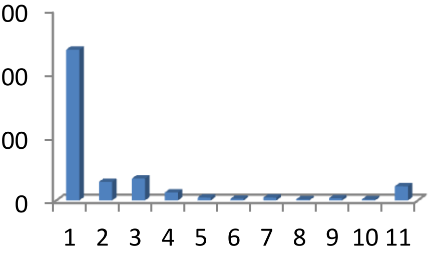


(b) Non-sacred

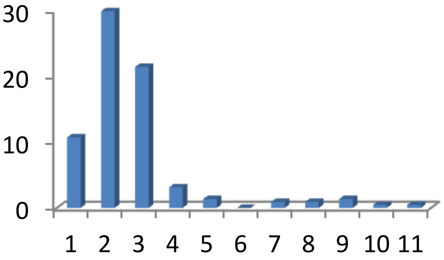
Ilex mitis



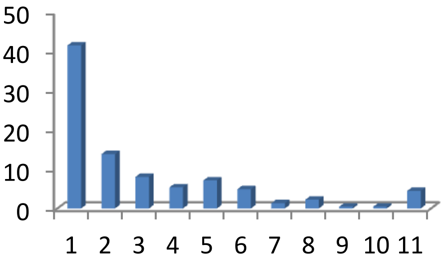
Syzygium guineense



Aguaria saxifolia

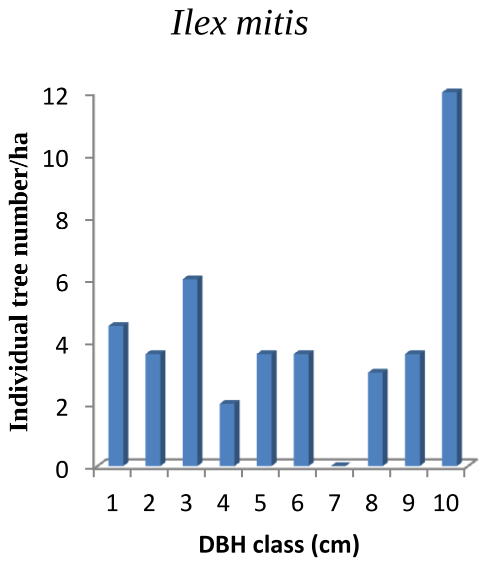


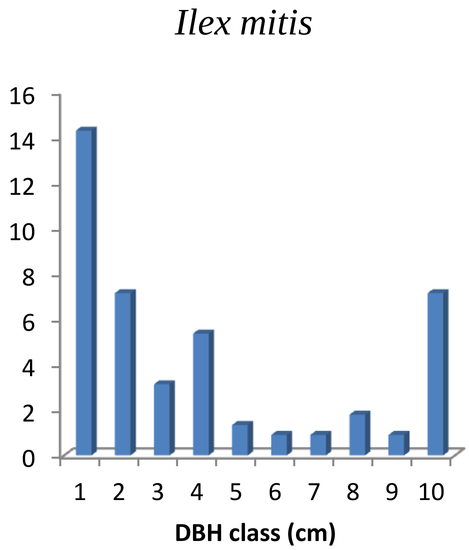
Olea capensis

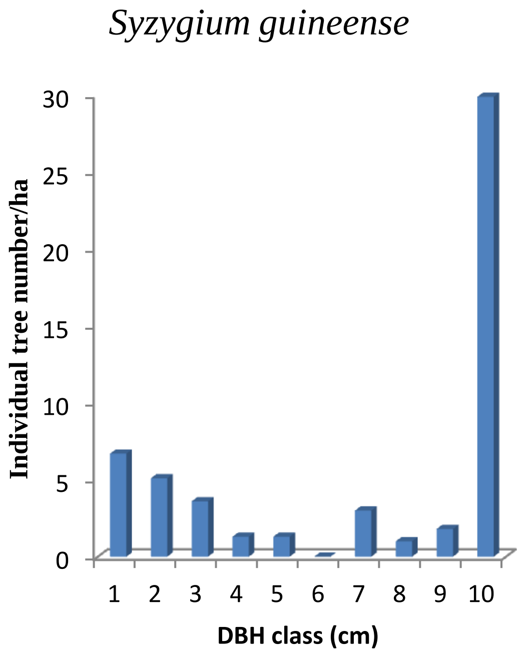


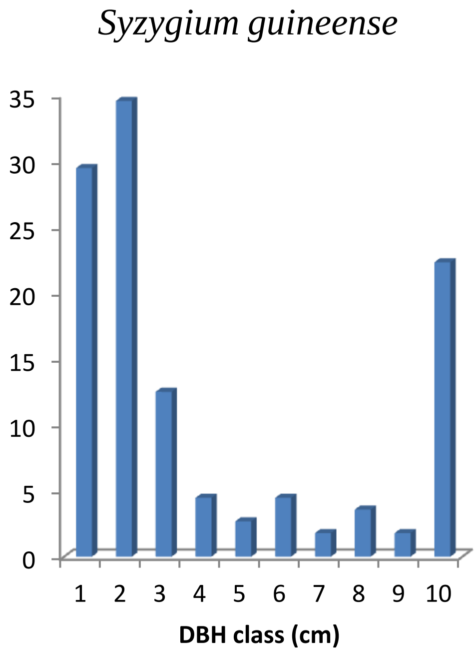
c) Sacred groves

d) Non-sacred groves









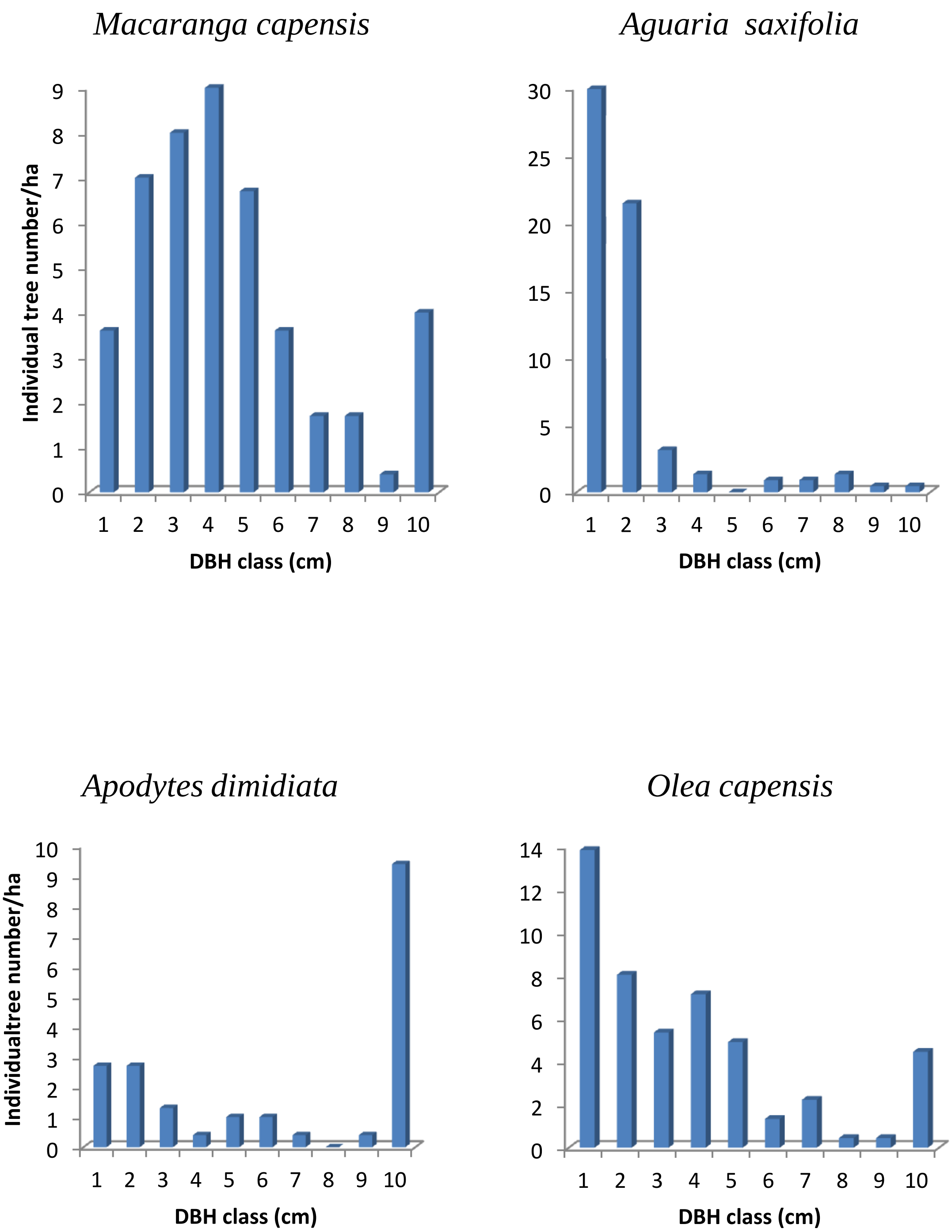


Figure 6.7 Population DBH size-class distributions of the four most abundant tree species in sacred groves and non-sacred forests in the Gamo Highlands of Ethiopia shown including and excluding the 0-4.9 cm DBH (the latter in order to better see the shape of distribution of individuals ≥ 5 cm DBH). DBH size classes are: (a, b) 1, 0-4.9 cm; 2, 5-9.9 cm; 3, 10-14.4 cm; 4, 15-19.9 cm; 5, 20-24.9cm; 6, 25-29.9 cm; 7, 30.-34.9cm; 8, 35-39.9 cm; 9, 40-44.9cm; 10, 45-49.9 cm; 11, ≥ 50 cm, and (c, d) 1, 5-9.9 cm; 2, 10.14.4 cm; 3, 15-19.9 cm; 4, 20-24.9 cm; 5, 25-29.9 cm; 6, 30.-34.9cm; 7, 35-39.9 cm; 8, 40-44.9cm; 9, 45-49.9 cm; 10, ≥ 50 cm.

6.3.7 Current level of disturbance

Levels of disturbance from grazing was scored on a scale from 0 (no disturbance) to 4 (highly disturbed). The value obtained between 0 and 4 for each 8 20 x 20 m plots in a single site were summed and divided into eight to get average disturbance for each site. For the eight 20 x 20 m plots in each sacred grove the summed scores ranged from 0.4 to 3.6, whereas in the non-sacred forests they ranged from 0.3 to 3.3 (Table 6.8). The disturbance did not show any significant correlations with density of trees and alpha diversity (Table 6.8). In contrast, a significant negative correlation between seedling and sapling density (dependent variable) and disturbance score (independent variable) was found: increase in disturbance significantly reduced seedling and sapling density in both forest categories (Table 6.8).

Table 6.8 Spearman rank correlation tests of correlation between scores of the intensity of disturbance (“D score”) and density of tree seedlings and saplings, density of trees and alpha diversity of trees in 56 20x20 m sample plots in six sacred groves and four non-sacred forests respectively in the Gamo Highlands of Ethiopia.

Sites	Density of tree seedlings and sapling (< 5 cm DBH)	Density of trees (> 5 cm DBH)/ha	Alpha diversity in 56 20 x 20 m plots	D score
Sacred groves				
Ula	3131.3	246.9	23.4	2.6
Gufae	8290.6	187.5	20.8	0.4
Qimme	1875.0	709.4	18.8	3.5
Tele	7475.0	309.4	18.3	1.1
Osha-Ocha	1665.6	418.8	17.1	3.6
Akasie	990.6	275.?0	18.8	3.6
r (correlation)	-0.985	0.5507	-0.456	
P (Significance)	< 0.05*	> 0.05	> 0.05	
Non-sacred forests				
Sora	2868.8	456.4	24.5	0.3
Shoa	2590.6	345.6	17.1	2.8
Dhule	1950.0	256.7	14.1	3.3
Oche	1121.9	198.5	13.4	3.5
r (correlation)	-0.948	0.313	0.326	
P (significance)	< 0.05*	> 0.05	> 0.05	

* Significant, $P < 0.05$

Disturbance from tree cutting was observed in both forest categories but its rate was slightly higher in the non-sacred forests (total stump density 198.7±46.4 ha^{-1}) than in the sacred groves (181.5±64.4 ha^{-1}) (Table 6.9). *Syzygium guineenses, Macaranga capensis* and *Croton macrostachyus* had the highest number of stumps in both forest categories. The majority of stumps of all sizes of all three species resprouted and had a high number of sprouts per stump.

Table 6.9 Indicators of tree cutting disturbance for six sacred groves and four non-sacred forests in the Gamo Highlands of Ethiopia.

			Sacred groves				
Sites	Ula	Gufae	Qimme	Tele	Akasie	Osha-Ocha	Mean(+/-SE)
Mean number of sprouts/stump	2.9	9.9	4.9	8.1	3.1	1.5	5.0±0.7
Density of living cut stumps/ha	78.1	293.8	118.8	46.9	175.0	168.8	146.9±35.7
Density of dead stumps/ha	0.0	112.5	53.1	73.4	34.4	37.5	51.8±15.6
Total density of stumps/ha	78.1	406.3	171.9	120.3	209.4	206.3	198.7±46.4
Density of uncut trees/ha	246.9	187.5	709.4	309.4	275.0	418.8	357.8±76.9
Total-stump/ha + Tree density/ha	325.0	593.8	881.3	429.7	484.4	625.1	556.5±78.8
Disturbance index* %	24.03	68.42	19.51	28.00	43.23	33.00	33.0±8.9
			Non-sacred forests				
	Sora	Shoa	Oche	Dhule			Mean (+/-SE)
Mean number of sprouts/stump	3.5	8.9	17.3	7.4			9.3±2.9
Density of living cut stumps/ha	53.0	91.7	28.1	206.3			94.8±39.4
Density of dead stumps/ha	96.9	81.3	18.8	150.0			86.8±27.0
Total density of stumps/ha	149.9	173.0	46.9	356.3			181.5±64.4
Density of uncut trees/ha	456.4	345.6	198.5	256.7			314.3±56.2
Total-stump/ha + Tree density/ha	606.3	518.6	245.4	613.0			495.8±86.2
Disturbance index* %	24.72	33.36	19.11	58.12			33.8±8.6

$$* = \frac{\text{Number of tree stumps}}{\text{Total number of uncut tree} + \text{number of stumps}}$$

The disturbance index of cut tree stumps as a proportion of the total of cut and uncut trees shows a remarkably similar level of disturbance for the two forest categories (33.0% of trees being cut in sacred groves and 33.8% in the non-sacred forests, Table 6.9). Gufae is the most disturbed of all the sacred groves with disturbance index of 68.42%, and has the highest count of living as well as dead stumps/ha. It also has the highest number of sprouts per stump, however most of them were not vigorous which is attributed to the high grazing intensity (Table 6.8). People used this sacred grove as a grazing ground for their cattle and the cattle fed on most of the sprouts. The highest disturbance from tree cutting was also recorded in Gufae sacred grove where people extract biomass for construction. The tradition of respecting the sacred groves was becoming less in this area. It was also observed that people had chopped parts of living trees for charcoal and firewood at this site. According to the elders, most of the dead stumps in this forest were due to cutting for house construction and firewood collection. Akasie was the second most highly tree-cutting disturbed sacred grove (disturbance index 43.23%). The ground under the canopy of this forest was used as a playground by the children from the surrounding village. The anthropogenic disturbance of Osha-Ocha sacred grove was mainly at the edge of the patch, and there was no observed human disturbance at the core of this site. There were a high number of tree falls in the middle of this grove due to wind-throw. No sign of grazing was observed in this forest. Tele had the greatest canopy cover of the sacred groves, however it was under pressure from cattle grazing and there were also a number of cut stumps observed in this site. Less tree cutting disturbance was observed in Ula and Qimme burial forests having 24.03% and 19.51% disturbance rates respectively.

With respect to the non-sacred forests, Dhule was the most tree-cutting disturbed forest with a density of 150 ha^{-1} of dead stumps and 206.3 ha^{-1} of living stumps (disturbance index = 58%). Shoa and Sora were second and third most disturbed with disturbance indices of 33% and 25% respectively. Oche may have suffered the highest rate of disturbance in the past as judged by its high density of trees with multiple stems (data not shown).

6.3.8 Regeneration status of sacred groves and non-sacred forests

6.3.8.1 Tree regeneration

Immature stems of tree species were observed in both the sacred groves and non-sacred forests. In some cases these clearly originated as sprouts from cut stumps. Others emerged from the ground but it was not possible to determine if they had established directly from germinated seeds or as sprouts from below ground organs. In this section the term seedling is used to describe any individual of a tree species with a minimum height of 0 cm and a maximum of 100 cm, and a sapling as any individual of a tree species with a minimum height of > 100 cm and a maximum DBH of < 5 cm. Plants in either size category with multiple stems (e.g. a branched stem or multiple sprouts from the same trunk) were counted only as a single individual. The density of seedlings and saplings together recorded in the sacred groves ranged widely from 317 m^{-2} at Akasie to 2653 m^{-2} at Gufae, and for non-sacred forests from 359 m^{-2} at Oche to 918 m^{-2} at Sora (Table 6.10). The mean (±SE) seedling density m^{-2} of six sacred groves (1111.7±393.2) and four non-sacred forests (476.8±87.3) and sapling density m^{-2} of six sacred groves (137.8±29.4) and four non-sacred forest (205.8±58.9) respectively were not significantly different (Kruskal Wallis test, H= 0.72, $p > 0.05$ and H= 1.64, $p > 0.05$ respectively). However, mean seedling density/plot (n=56) of each species of sacred groves (53.17±17.14) and non-sacred forest (15.18±4.92) was significantly different (t = 3.06, $p < 0.05$) while sapling density/plot of each species of sacred groves (6.60±1.82) and mean density/plot of each saplings of non-sacred forest (6.56±1.56) was not significant (t = 0.019, $p > 0.98$). The ratio of seedlings to saplings was notably higher for the sacred groves (8:1) than the non-sacred forests (1.5:1).

Table 6.10 The total density of seedlings and of saplings of tree species (individuals m^{-2}) for each site of sacred and non-sacred forests.

Sacred grove	Seedling	Sapling	Non-sacred forest	Seedling	Sapling
Ula	831	171	Sora	709	209
Gufae	2468	185	Shoa	470	359
Qimme	426	174	Dhule	442	182
Tele	2187	205	Oche	286	73
Osha-Ocha	481	52			
Akasie	277	40			
	1111.7±393.2	137.8±29.4		476.8±87.3	205.8±58.9

There was a noTable difference in seedling/sapling species composition and community structure between the two forest types. Thirty two species were recorded in the sample plots in the sacred groves and 28 in the non-sacred forests (Table 6.11). It is noTable, however, that *Syzygium guineense* dominated the seedling size class, but was lower than fifth in rank order in the sapling size class, in both forest types. Its absolute density of seedlings was ca. four times greater in the sacred groves, but of saplings was ca. two times greater in the non-sacred forests. Eight of the next nine ranked seedling species in the non-sacred forests, *Macaranga capensis, Ekbergia capensis, Apodytes dimidata, Chionanthus mildbraedii, Canthium oligocarpum, Flacourtia indica, Prunus africana* and *Celtis africana,* all had notably higher relative and absolute seedling densities in the sacred-groves than the non-sacred forests. While *Bersama abyssinica* had a slightly higher absolute seedling density in the sacred groves, it was second in rank (with a higher relative density) in the non-sacred forests. The third ranked species of seedling in the non-sacred forests, *Lepidotrichilia volkensii*, had a higher absolute density than in the sacred groves and the fourth ranked species, *Olinia rochetifoliana,* was (notably) completely absent from the sacred grove plots. Of the top ten ranked seedling species in the sacred-groves only the tenth, *C. africana*, was absent from the plots in the non-sacred forests.

It was noTable that for all of the most abundant seedling species, their ratio of saplings to seedlings was higher in the non-sacred forests than the sacred groves, e.g. *Syzygium guineense* had 0.012 saplings per seedling in the sacred groves but 0.090 in the non-sacred forests, *Macaranga capensis* had 0.021 and 0.652, *Ekbergia capensis* had 0.030 and 0.231, *Apodytes dimidata* had 0.217 and 0.349, *Chionanthus mildbraedii* had 0.133 and 1.222, *Bersama abyssinica* had 0.175 and 0.251, *Canthium oligocarpum* had 0.077 and 0.594, *Flacourtia indica* had 0 and 1.978 and *Prunus africana* had 0.010 and 0.621 respectively. This could indicate higher seedling survival rates in the sacred groves or faster seedling (or slower sapling) growth rates in the non-sacred forests.

In both forest categories, some species were recorded as present as trees but not as seedlings or saplings. In the sacred groves these were *Schefflera abyssinica* (which was eighth in rank in tree abundance with an importance value of 4.3), *Cordia africana* (a nationally conservation priority species), *Maytenus addat* (an endemic species), *Ficus vasta* and *Albizia schimperiana*. Of these five, only one (*S. abyssinica*) was recorded as a seedling in the non-sacred forests. For the non-sacred forests, the following species were recorded as trees but were absent as seedlings or saplings: *Dracaena fragrans* (which was sixth in rank in tree abundance with an importance value of 6.7), *Myrica saxifolia*, *Dombeya torrida*, *Maytenus arbutifolia* and *Polyscias fulva*. Of these five species, two (*D. torrida* and *M. arbutifolia*) were recorded as seedlings and saplings in the sacred groves.

A number of species were recorded as present as seedlings in the plots, but not as trees: *Flacourtia indica* (high seedling density but absent as saplings in the sacred groves, low seedling and sapling density in the non-sacred forests), *Phoenix reclinata* (moderately low seedling density but absent as saplings in the sacred groves, absent from the non-sacred forests), *Erytherinea brucei* (an endemic species, low seedling density but absent as saplings in the sacred groves, absent from the non-sacred forests), *Ficus sur* (low seedling density but absent as saplings in the sacred groves, low seedling and sapling density in the non-sacred forests) and *Ficus palmata* (very low seedling density but absent as saplings in the sacred groves, absent from the non-sacred forests).

Table 6.11 Density of seedlings (se) and saplings (sa) per hectare of (a) individual tree species and (b) three functional groups of species (wide-habitat/generalist, WH; forest-habitat, FH; and forest-margin/light demanding, FM) in six sacred groves and four non-sacred forests in the Gamo Highlands of Ethiopia. (F.group = functional groups). Se = Seedling, Sa = Sapling

(a)

Sacred groves				Non-sacred forest			
Species	Se ha^{-1}	Sa ha^{-1}	F. group	Species	Se ha^{-1}	Sa ha^{-1}	F.group
Syzygium guineense	839.3	10.3	WH	*Syzygium guineense*	218.3	19.6	WH
Macaranga capensis	298.2	6.3	FH	*Bersama abyssinica*	138.8	34.8	WH
Ekbergia capensis	296.9	8.9	FM	*Lepidotrichilia volkensii*	87.5	54.0	FH
Apodytes dimidiata	224.1	48.7	FH	*Olinia rochetifoliana*	53.6	13.4	FM
Chionanthus mildbraedii	167.0	22.3	FH	*Allophylus abyssinicus*	50.0	10.3	WH
Bersama abyssinica	163.4	28.6	WH	*Maesa lanceolata*	42.4	19.2	FM
Canthium ologocarpum	133.5	10.3	FH	*Vepris dainellii*	40.6	46.9	FH
Flacourtia indica	113.0	0.0	FM	*Apodytes dimidiata*	38.4	13.4	FH
Prunus africana	92.9	0.9	WH	*Macaranga capensis*	30.8	20.1	FH
Celtis africana	80.4	71.9	WH	*Ilex mitis*	24.1	18.8	FH
Euphorbia candelabrum	71.9	4.9	FM	*Canthium oligocarpa*	16.5	9.8	FH
Ilex mitis	71.0	8.5	FH	*Prunus africana*	12.9	8.0	WH
Maesa lanceolata	68.8	25.0	FM	*Olea capensis*	12.9	28.6	FH
Maytenus undata	65.2	36.2	FM	*Galiniera saxifraga*	12.5	2.2	FH
Allophylus abyssinicus	61.2	4.9	WH	*Croton macrostachyus*	12.1	6.3	WH
Pittosporum viridiflorum	60.3	3.6	FH	*Ekbergia capensis*	11.6	2.7	FM
Galiniera saxifrage	46.9	8.9	FH	*Schefflera volkensi*	9.8	9.8	FH
Lepidotrichilia volkensii	30.4	26.8	FH	*Aningeria adolfi-friederici*	7.6	15.6	WH
Vepris dainellii	30.4	20.1	FH	*Dovyalis abyssinica*	7.6	2.23	FM
Albizia gummifera	13.8	0.9	WH	*Nuxia congesta*	6.7	8.84	FM
Croton macrostachyus	10.3	0.5	WH	*Chionanthus mildbraedii*	4.9	6	FH
Dracaena afromontane	8.9	11.2	FM	*Aguaria saxifolia*	4.9	5.4	FH
Maytenus arbutifolia	8.0	5.8	FM	*Maytenus addat*	2.2	5.8	FM
Phoenix reclinata	7.6	0	FM	*Ficus sur*	0.9	0.5	WH
Nuxia congesta	5.8	0.9	FM	*Maytenus undata*	0.9	4.0	FM
Erythrina brucei	2.2	0	FM	*Flacourtia indica*	0.5	0.9	FM
Ficus sur	1.8	0	WH	*Pittosporum viridiflorum*	0.5	0.5	FH
Olea capensis	1.3	0.5	FH	*Schefflera abyssinica*	0.5	0	WH
Dovyalis abyssinica	1.3	0.5	FM				
Dombeya torrida	0.5	1.8	WH				
Millettia feruginea	0.5	0.5	WH				
Ficus palmata	0.5	0	FH				
Total	2977.2	369.4			850.0		367.5

(b)

Functional groups	Sacred groves (mean ±SE)		Non-sacred forest (mean ±SE)	
	Seedling	Sapling	Seedling	Sapling
Wide-habitat/generalist (WH)	98.4±22.2	8.6±6.8	35.4±10.9	7.8±3.7
Forest-habitat (FH)	76.0±10.5	11.1±4.1	24.1±4.8	16.4±3.9
Forest-margin/light demanding (FM)	57.8±12.5	8.8±4.5	7.9±4.8	4.7±2.9

6.3.8.2 Sprouting from stumps

Several species were found to sprout from cut stumps. The multiple sprouts on single stumps were recorded separately and summed to quantify total density of individual sprouts per hectare. The total density of sprouts was similar between the two forest types: 831.7 ha^{-1} in the sacred groves and 718.4 ha^{-1} in the non-sacred forests. For the sacred groves the mean of their mean number of sprouts per stump was 5.0±0.7 (with a range between sites from 1.5 to 9.9) (Table 6.9). The number of sprouts per stump was greater for the non-sacred forests (mean of site means 9.3±2.9), ranging between sites from 3.5 to 17.3. The species showing the largest number of sprouts per stump, or else the greatest observed "vigour" of sprouts, in the sacred groves were *Syzygium guineense, Macaranga capensis, Apodytes dimidata, Maytenus undata, Canthium oligocarpum, Croton macrostachyus* and *Nuxia congesta*. All of these species were also recorded as seedlings and saplings, however for the following three their sapling density was < 1 ha^{-1} and so vegetative sprouting might be a particularly important regeneration mechanism: *C. macrostachyus, N. congesta* and *A. gummifera*.

6.4 Biocultural diversity value of sacred groves

Twenty four people were interviewed from the age group 45-70 years. They consisted of religious leaders, clan elders, lineage heads and community chiefs. All of them were men and had lived in the village where they were interviewed since childhood and had acquired local beliefs and values during their upbringing. They fall into the broad group of "elders" who are very conversant with traditional rituals, norms and taboos. They are responsible for organizing rituals on sacred sites and punishing those community members who act against traditional norms. The opening discussions with them were focused specifically on the origin, uses and current threats of the sacred groves, and how the respondents had gained the knowledge and skills necessary to manage the groves.

The interviewees consistently reported that their knowledge has generally been passed down from their ancestors but they all identified their fathers as the direct source of their knowledge. They also stated that they consult other knowledgeable elders within their own clan or sometimes from different clans (social units in communities). The process of gaining this knowledge was largely through participation and observation, which had occurred since their childhood. According to the respondents, the eldest son amongst the family members usually takes the responsibility to perform the ceremonies and other roles connected with the sacred groves that their father did in the past.

The respondents also reported that the origin of sacred groves was back in the time of their ancestors. They also noted that the sacred groves are not the only sacred places in the Gamo Highlands. They stated that the Gamo people believe that every natural object, such as mountains, pasture lands, rocks, rivers, streams, trees, animals, footpaths, the sun and the moon, are potentially sacred and that numerous spirits, both good and evil, inhabit them in local tradition. For example, the sun has its own spirit and according to local traditions this explains why the dry season is sometimes unusually long and hot. It is also believed that the sun has divine power that lights the whole earth. It needs prayer and offerings as a mark of respect. The elder locally known as arshe eqaa, (*arshe* = sun, *eqaa* = name traditionally given to a religious leader) makes offerings and prayers to the sun either together with other elders or alone.

Similarly, the rain has its own spirit, explaining why the onset of rain at the start of the wet season is sometimes delayed or takes longer to stop at the end of the wet season and into the harvesting season. On these occasions *Eraa kawo* (the king of rain) makes a prayer on behalf of the community to bring the rain or sometimes to stop too much rain that people think will destroy the crops. Sacred assembly places known as *dubusha* (a small place surrounded by a stone wall or wooden fences) are common in the Gamo Highlands. They are used for community affairs, such as settling disputes between two villages or individuals by local elders and religious leaders. Sacred pasture lands known as *kalo* are also mentioned as sacred sites. They are used for community gatherings for thanksgiving ceremonies. People get together and celebrate *Meskel* festivals (celebration of New Year) on sacred pasture lands. Collective decision-making related to communal lands also takes place on sacred pasture lands.

Similar beliefs and practices apply to the sacred groves. Even though the whole village is responsible for the groves, each grove has its own eqaa as well. *Eqaa* perform offerings and make prayers each year to upgrade the sanctity of the groves and the wellbeing of the whole community. The respondents indicated that the sacred groves are the most threatened of the sacred sites in the Gamo Highlands and they were increasingly concerned about their deforestation and degradation. According to the interviewees, sacred groves had enormous significance for the people as evidenced by the following: prayers are offered to the deity by the ritual leaders and community elders. Specific beliefs about the sacred groves are that they harbor spirits that help families and protect them from any misfortune. The custodians believed in the sanctity of sacred groves and the spirits inhabiting them. The sacred groves on which this study focused were burial grounds (locally known as *bossa* or *dufo*) and relict natural forests (known as *kasha*).

6.4.1 Management of sacred groves

Because of the beliefs about the spiritual significance and sanctity of the sacred groves, it is not permitted for anyone to enter the sacred groves except ritual leaders when they make thanksgiving sacrifice and children under the age of puberty. Cutting trees within the sacred groves is not permitted by anyone under any circumstances. Hunting in sacred groves is also forbidden because eating meat of animals from sacred groves is considered taboo. Grazing is not allowed. The general belief of the custodians was that if anyone breaks these rules and enters the sacred groves or even cuts trees or kills animals, the spirit would bring wrath to the village in the form of epidemic disease, or death of children and milking cows as a punishment.

The respondents stated that the Gamo people had the custom of confession whenever they break the rules, followed by performing a sacrament to receive a pardon for their wrong action in the presence of ritual leaders. All the respondents agreed that anyone who was not willing to perform the sacrament would encounter misfortune including death to one of his family members. For the wellbeing of his family he must perform the custom. The sacrament is usually the sacrifice of a domestic animal, either a goat, sheep or cock based on what the ritual leaders recommended, which is slaughtered to pacify the wrath of the spirit of the sacred groves.

In addition to this general belief system custodians also recently started protecting the sacred groves from domestic animals. If anyone breaks the rule, the community council that consists of elders, the chief and religious leaders punish the law breakers by demanding a payment in kind or cash based on their preference. Other potential sources of damage to sacred groves, such as forest fire and fire from land clearing for farming, were not considered to be a problem in the area. Even if they manage sacred groves in different ways, according to their interview responses, the custodians do not carry out active management, e.g. restoring degraded sacred groves by planting trees and promoting tree regeneration. Digging sacred ground with a sharp metal implement such as a hoe is considered taboo. This may be an influence against planting trees inside sacred groves. However, non-timber forest products such as wild fruit and medicinal plants are allowed to be collected by children from the sacred groves.

6.4.2 Value of sacred groves

All plant species present in the sacred groves are protected through belief systems (see appendix 6.1 for lists or plant species present in the surveyed sacred groves). Use of these plant species is carefully managed in sacred groves. While cutting trees is not permitted, limited gathering of non-timber forest products such as fruits by children, medicinal plants and plants used for ritual initiations by ritual leader is allowed since they are more common in sacred groves. For example, flowers of *Hagenia abyssinica* are collected from sacred groves for medical purposes. The flower of *H. abyssinica* is widely used in Ethiopia to expel tapeworms from the intestine (Negash, 1995). Leaves of *Arundinaria alpina* are collected from sacred groves for ritual initiations during male circumcision, while the stem is used as a symbol of fortune and holiness. The whole plant part of *Ranunculus multifidus* is used to ritually cleanse grave-digging hoes before they are taken home. *Discopodium penninervium* is used to ritually purify two individuals who are in dispute so that their differences are resolved. Besides these uses, the respondents stated that sacred groves are also valued as a wind break, to minimize soil erosion, and as a symbol of identity and beauty by their presence.

6.4.3 Threats

The respondents stated that sacred groves in the Gamo highlands currently face different threats. They classified the threats into two: physical threats and spiritual threats. Twenty of the 24 respondents said that the two main physical threats to the sacred groves were grazing and tree cutting. The remaining four placed more emphasis on grazing than tree cutting. All 24 of the respondents said that the spiritual value of sacred groves was threatened by being undermined by changes in belief systems between generations and that this, in turn, caused the above physical threats. The following responses are representative of the views of all the respondents:

> *“Today sacred groves and other sacred places are disregarded by some people especially by the young generation. As a result wrath came to our village as well as to the region as a whole. Consequently, there was no harvest, the harvest had no value. Milking cows have died and the living are not giving milk as much as before when the sacred groves were respected by the whole village all together.”*

> “*Today we are spiritually impure because some [village members] are going their own way. They have chosen another religion. On top of that they keep preaching to our family members so that they abandon their customs and beliefs and follow them, which is equivalent to preaching to lose identity”.*

> *“Our children are out of control, we can’t teach them our customs as we used to since they are not willing to participate and learn. We are all spiritually degraded because of them”.*

> *“Because of all these, we are under the wrath of the spirit. You can see many people dying each year due to the wrath. Marriage is not fruitful, rain is unpredicTable, and untreaTable disease [HIV] is killing our people. Before [when sacred groves were respected], if something went wrong we went and prayed to the sacred grove and got an answer immediately. Today, when we pray to the sacred grove, it takes longer to get a response or we get nothing at all.”*

The appropriation of sacred groves by other strong religions was identified as a spiritual threat by the respondents. In corroboration of this, I observed that some sacred groves had been changed into places of modern religion during a broad survey of sacred sites in the Gamo highlands. In discussing this, some of the respondents noted that, in some places, the custodians of the sacred groves were preached at by those promoting new religious beliefs and told that they are backwards and believe in superstition. This type of preaching is occurring widely in the Gamo highlands with the intention of taking over the sacred groves to build churches inside them. Most interviewed custodians also mentioned that they were under pressure from their spiritually converted relatives and forced to believe in both the indigenous and modern religion. From these and other examples highlighted above, it appears that sacred groves and their associated culture are under pressure, which in turn imposes threats to the biocultural diversity of sacred groves.

6.5 Land-use and land-cover change impact on sacred groves

6.5.1 Land-use and land-cover composition at the landscape level

There was a large change in land cover in the study area due to human land-use in the 15-year period, 1995-2010 (Figure 6.17, Table 6.12). Wooded grassland was the dominant and most connected land-cover class in 1995, occupying 35% of the study area, however almost half of the area of this class had been converted to farm and settlement land cover by 2010 (Figure 6.8a). Farm and settlement was the second largest in area (28.8%) in 1995, then it more than doubled in area to become the dominant land cover by 2010. This is evidence that the area of human land-use in the Gamo highlands increased rapidly during this period at the expense of more natural habitats. Open pasture land was 25.3% of the area sampled in 1995 and largely restricted to high elevation areas which include the alpine grassland habitats. This land-cover type reduced to 15.6% of the total area by 2010, a loss of 6488 ha. Forest covered just 7216 ha (10.8%) of the study area in 1995 and was reduced to 4573 ha (6.8%) in 2010 (Figure 6.8a). In summary, between 1995 and 2010 wooded grassland lost 51.0% of its area, open pasture 38.4% and forest 36.6% all predominantly to farm and settlement land (Figure 6.8b).

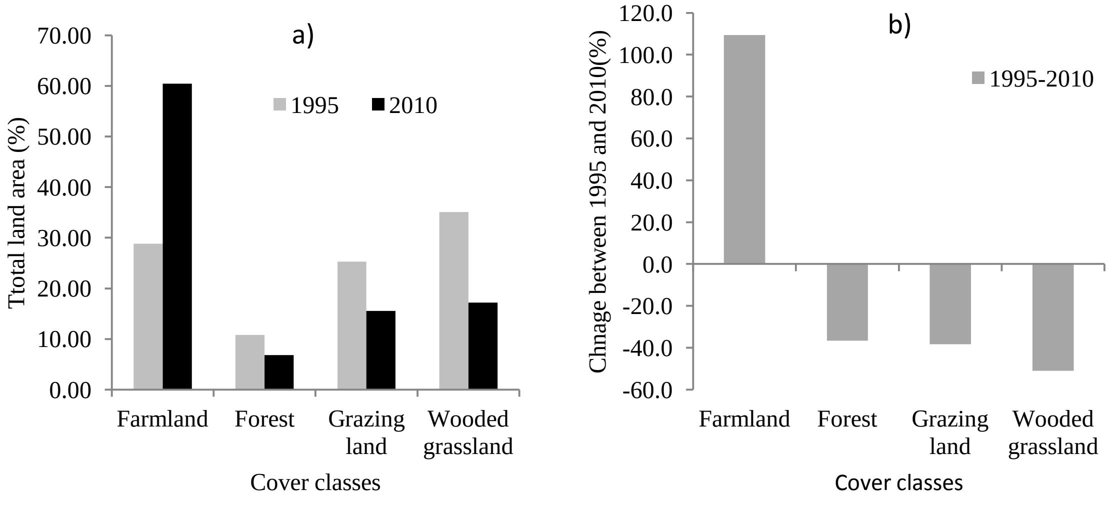


Figure 6.8 Land-cover of the Gamo Highlands study area (66,764 ha): a) area of each land-cover class in 1995 and 2010 and b) percentage change in area of each land-cover class between 1995 and 2010. Farmland comprises cultivated and settled land.

Table 6.12 Land-cover and land-cover change of the Gamo Highlands study area (1995-2010).

Year/period	1995		2010		1995-2010	change
Cover classes	Area (ha)	%	Area (ha)	%	Area (ha)	%
Farm and settlement	19,255	28.84	40,329	60.4	21,074.0	+109.4
Forest	7216	10.81	4573	6.8	-2643.0	-36.6
Grazing land	16,878	25.28	10,390	15.6	-6488.0	-38.4
Wooded grassland	23,415	35.07	11,472	17.2	-11,943.0	-51.0
Total	66,764	100.00	66,764	100.0	0	

There was a decrease in the number of patches of all four land-cover classes between 1995 and 2010 indicating a simplification of the pattern of land cover (Figure 6.9a, Table 6.13). Forests had the smallest reduction in patch number (from 1718 to 1441 (19%, or 18.5 patches per year).

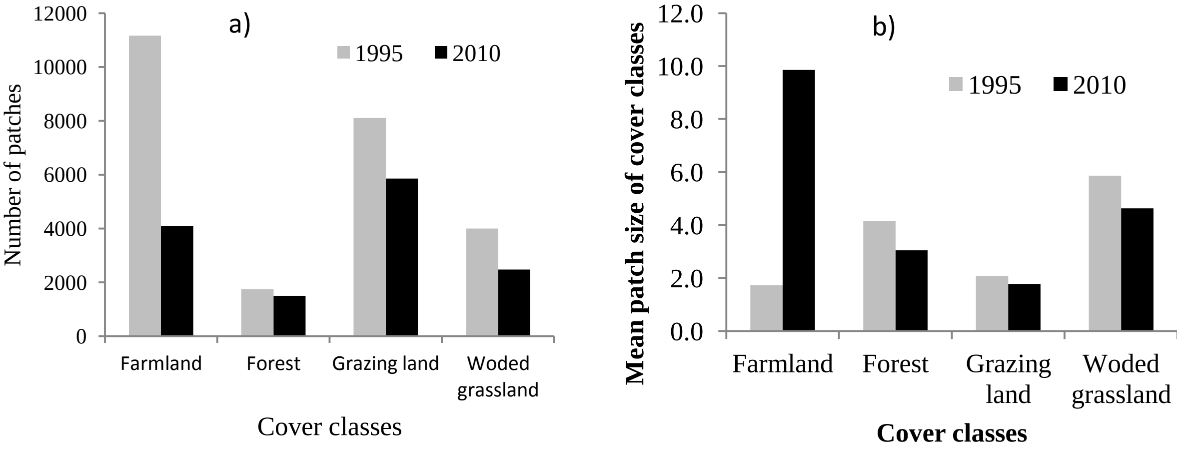


Figure 6.9: Fragmentation of land-cover classes in the Gamo Highlands study area in 1995 and 2010: a) number of patches, b) mean patch size (ha).

In 1995 individual patches of farm and settlement land were often still interspersed by land of the three other land-cover classes having a mean patch size of only 1.7 ha (Table 6.11). By 2010 the patches of farm and settlement land had often coalesced through conversion of the intervening land cover increasing their mean patch size more than five-fold to 9.9 ha (Table 46, Figure 6.9b). During this process there was both a decrease in patch number and in mean patch area of all three of the other land-cover classes. This simplification of the landscape structure was also reflected in the reduction in values of patch Mean Shape Index and Edge Density metrics for all four land-cover classes between 1995 and 2010 (Table 6.13). The percentage reduction in edge density was greatest for wooded grassland and least for farm and settlement, whereas the reduction in mean shape index was greatest for forest. The Patch Size Coefficient of Variation (PSCOV) value, indicating variation of patch size in the landscape, increased for farm and settlement land, while decreasing for the other three classes, especially open pasture land and wooded grassland.

Table 6.13 Fragmentation metrics for four land-cover classes in the Gamo Highlands study area in 1995 and 2010.

	Number of patches		Mean patch size (ha)		Edge density (m ha^{-1})		Patch size coefficient of variation		Mean shape index	
Year/land-cover class	1995	2010	1995	2010	1995	2010	1995	2010	1995	2010
Farm and settlement	11159	4093	1.7	9.9	67.7	62.4	837.0	5683.0	1.5	1.4
Forest	1718	1441	4.1	3.0	14.8	10.5	949.5	816.3	1.5	1.3
Open pasture land	8106	5853	2.1	1.8	47.2	34.4	3712.0	1239.8	1.4	1.3
Wooded grassland	3996	2475	5.9	4.6	34.5	18.9	4420.9	2431.7	1.5	1.4

When fragmentation is assessed at the landscape level (which sums across all four land-cover classes) the same trends of simplification and reduction in heterogeneity between 1995 and 2010 are apparent, with many small patches being converted to the land-cover of their surrounding matrix. There was a reduction in both the Shannon diversity and evenness indices for the landscape of 2010 compared with the landscape of 1995 (Figure 6.10, Table 6.14). Mean patch size greatly increased while mean shape index was slightly reduced (Figure 6.11), and there was a large reduction in the total number of patches (Figure 6.12) and edge density (Figure 6.13). The mean shape index (MSI) decreased from 1.5 in 1995 to 1.4 in 2010 (Figure 6.11) which indicate that the patches had a more irregular shape in 1995. The more regular shape in 2010 indicates an increase in pressure on patches from land-use in the surrounding matrix.

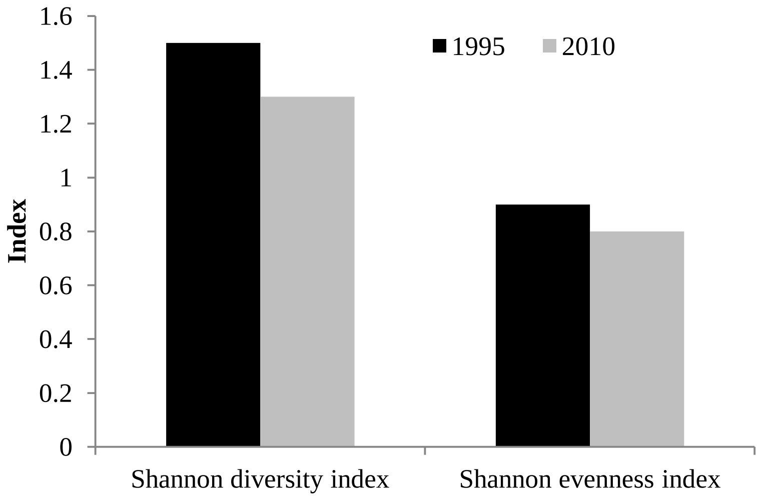


Figure 6.10 Shannon diversity and evenness indices for patches of all four defined land-cover classes in the Gamo highlands study area in 1995 and 2010.

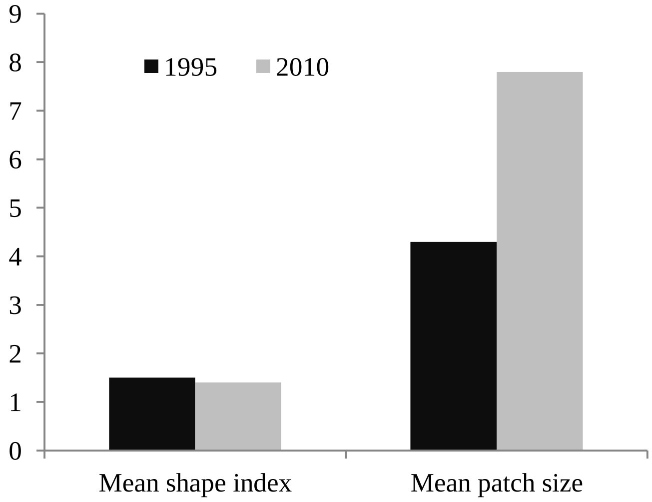


Figure 6.11 Mean shape index and mean patch size (ha) for patches of all four defined land-cover classes in the Gamo Highlands study area in 1995 and 2010.

The reduction of mean patch size (MPS) overtime indicates the fragmentation of patch. In contrast, the mean patch size in Gamo highland increased from 4.3 ha in 1995 to 7.8 ha in 2010 which means that most of the small patches lost to other land-uses. This further confirmed by decrease of patch number in the landscape of the study area. The number of patches (NP) has decreased greatly from 25,004 in 1995 to 13,926 in 2010 due to changes in land use (Table 6.13, Figure 6.12). Taking the evidence of these metrics together, the pattern of landscape change in the Gamo highland is that of attrition of land-cover types rather than fragmentation (Forman, 1995). Many patches have been diminished to destruction and replaced with other land-uses rather than remaining as fragments. The edge density decreased from 166.6 m ha^{-1} in 1995 to 128.6 m ha^{-1} in 2010 (Figure 6.13). Edge density at the landscape level is directly related to the degree of spatial heterogeneity (McGarigal et al., 2002), hence the study area became less heterogeneous in 2010.

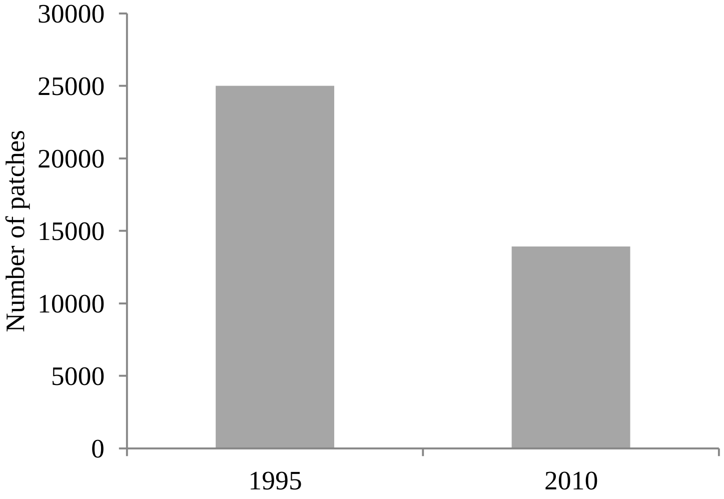


Figure 6.12 Number of patches of all four defined land-cover classes combined in the Gamo Highlands study area in 1995 and 2010.

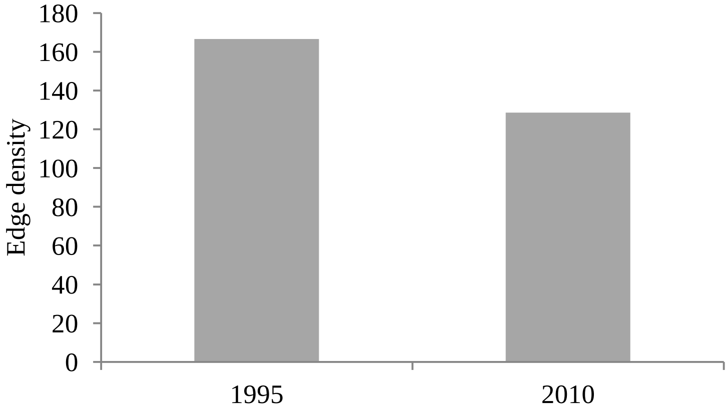


Figure 6.13 Edge density (m ha^{-1}) of all four defined land-cover classes combined in the Gamo Highlands study area in 1995 and 2010.

Table 6.14 Fragmentation metrics of all four defined land-cover classes combined in the Gamo Highlands study area in 1995 and 2010. SDI, Shannon diversity index; SEI, Shannon evenness index; MSI, mean shape index; ED, edge density; MPS,mean patch size; NumP, number of patches; PSCOV, patch size coefficient of variation; TA, total area (ha).

Year	SDI	SEI	MSI	ED	MPS	NumP	PSCOV	TA
1995	1.5	0.9	1.5	166.6	4.3	25,003	6620.3	66,765
2010	1.3	0.8	1.4	128.6	7.8	13,926	6016.8	66,765

6.5.2 Fragmentation of individual sacred groves and non-sacred forests

Fragmentation was also assessed at the scale of the individual patches of the six sampled sacred groves and four non-sacred forests. Change in shape index of the six sacred groves was variable between 1995 and 2010 with little overall change in the mean (Table 6.15): for Ula, Qimme and Tele there was no noTable change, for Osha-Ocha and Akasai a decrease and for Gufae a large increase (Figure 6.14a, Table 6.15). In contrast, shape index decreased notably for three of the four non-sacred forests patches (indicating greater regularity of shape) (Figure 6.14b, Table 6.15).

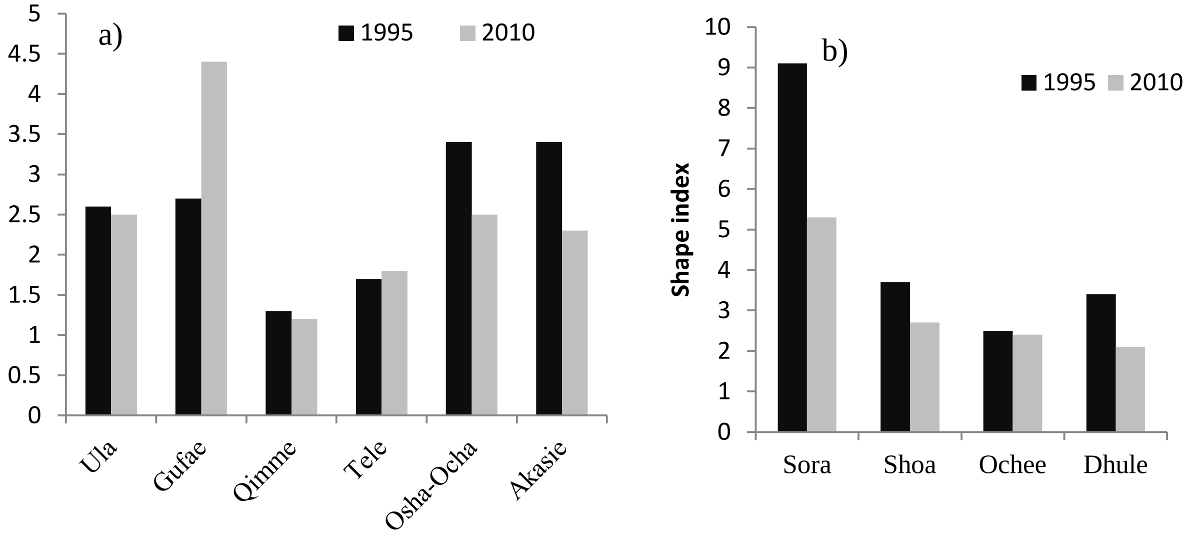


Figure 6.14 Shape index of (a) six sacred grove and (b) four non-sacred forest patches in the Gamo Highlands of Ethiopia in 1995 and 2010.

On average patch size decreased by 6% for the sacred groves, however the most noTable feature was its high variability with a noTable increase in Gufae and a small increase in Tele (Figure 6.15a Table 6.15). In contrast, all non-sacred forests showed a large decrease in size (on average by 43%) (Figure 6.15b).

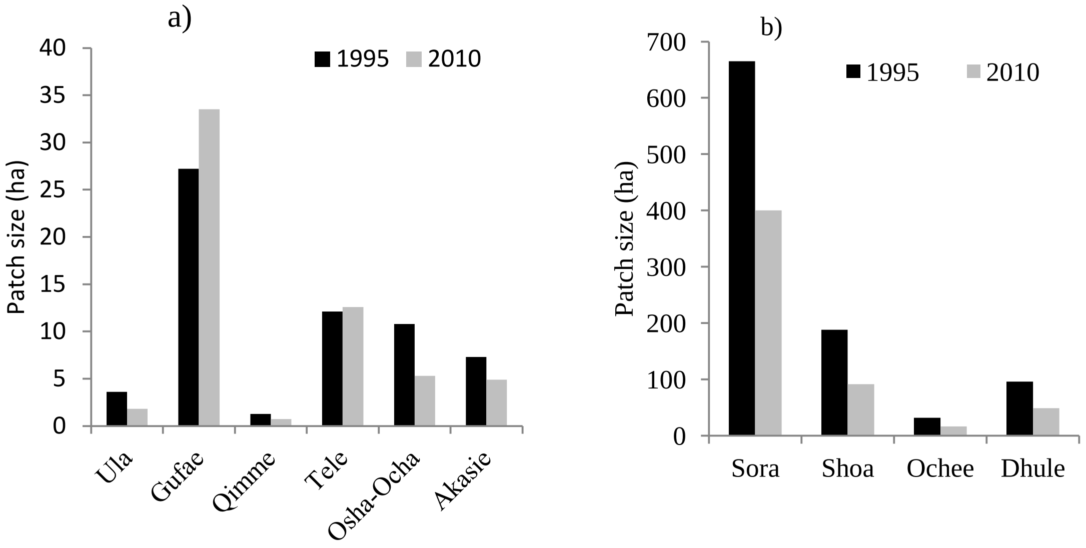


Figure 6.15 Patch size (ha) of (a) six sacred groves and (b) four non-sacred forest patches in the Gamo Highlands of Ethiopia in 1995 and 2010.

The mean edge density for sacred groves increased significantly from 308.2±42.9 m ha^{-1} in 1995 to 392.3±60 m ha^{-1} in 2010 (paired t-test, $t = -3.451$, $p < 0.05$), and in each of the six individual sites (Figure 6.16a, Table 6.15). The change in edge density was much more variable amongst the four non-sacred forests, resulting in a very similar mean for 1995 and 2010 (Table 6.15). The edge density reduced in Sora and Dhule and increased for Shoa and especially Ochee (Figure 6.16b, Table 6.15). In 1995 edge density was higher for all six sacred groves than for all four non-sacred forests; by 2010 this was still the case except for Tele and Ochee.

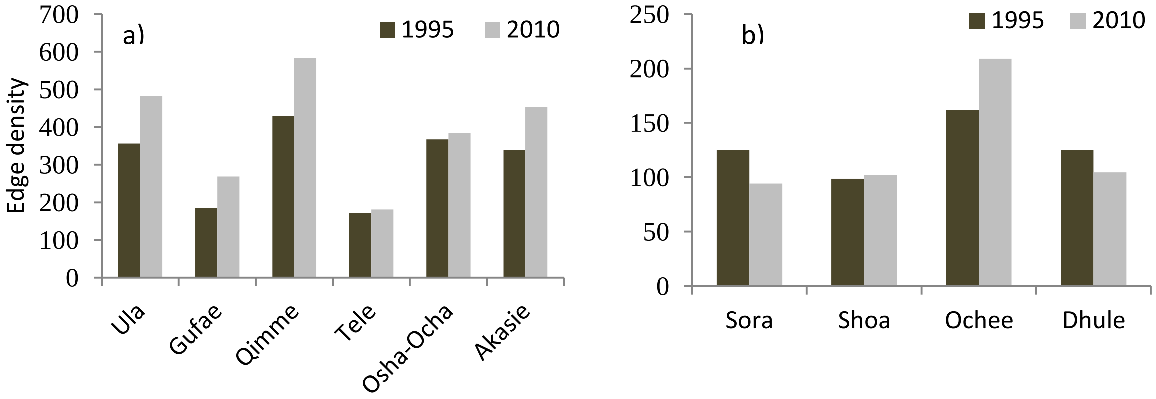


Figure 6.16 Edge densities (m ha^{-1}) of (a) six sacred grove and (b) four non-sacred forest patches in the Gamo Highlands of Ethiopia in 1995 and 2010.

Table 6.15 Shape index, edge density and patch size of six sacred grove and four non-sacred forest patches in the Gamo Highlands in 1995 and 2010. Dch = direction of change.

	Shape Index			Edge Density (m ha^{-1})			Patch Size (ha)		
	1995	2010	Dch	1995	2010	Dch	1995	2010	Dch
Sacred groves									
Ula	2.6	2.5	=	355.7	483.2	+	3.6	1.8	-
Gufae	2.7	4.4	+	184.3	269.6	+	27.2	33.5	+
Qimme	1.3	1.2	=	429.6	583.3	+	1.26	0.72	-
Tele	1.7	1.8	+	172.5	180.9	+	12.1	12.6	+
Osha-Ocha	3.4	2.5	-	367.4	384.2	+	10.8	5.3	-
Akasie	3.4	2.3	-	339.4	452.7	+	7.3	4.9	-
Mean (+/-SE)	2.52±0.3	2.45±0.4	-	308.2±42.9	392.3±60	+	10.4±3.8	9.8±5	-
Non-sacred forests									
Sora	9.1	5.3	-	125.3	94.2	-	665.4	400.2	-
Shoa	3.7	2.7	-	98.7	102.3	+	188.3	91.4	-
Ochee	2.5	2.4	=	161.6	209.4	+	31.7	16.6	-
Dhule	3.4	2.1	-	125.2	104.5	-	96.3	48.8	-
Mean (+/-SE)	4.7±1.5	3.1±0.7	-	127.7±13	127.6±27.2	=	245.4±143.5	139.3±88.2	-

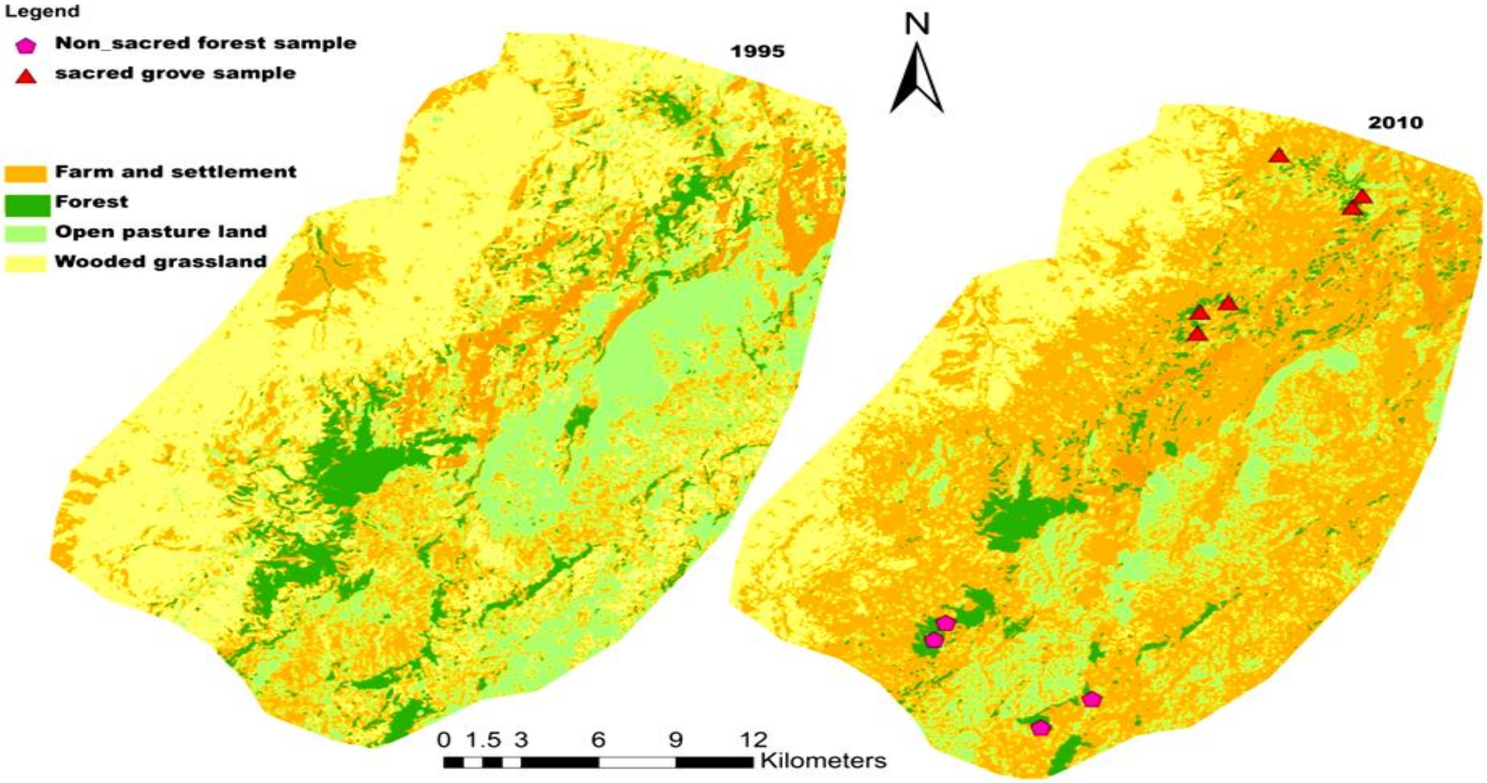


Figure 6.17 Land-cover classification of the Gamo Highlands study area in (a) 1995 and (b) 2010 showing the location of the six sacred groves and four non-sacred forests.

6.6 Discussion

6.6.1 Species richness, diversity, composition, structure and density of sacred groves versus non-sacred forests

Despite the smaller size of sacred groves there were no significant differences in tree species richness (per fixed area of sample plot) compared with larger non-sacred forests in the Gamo highlands of south-west Ethiopia. In addition, no significant differences were found between the two forest types in individual-based species accumulation curves by sample plot. Individual-based rarefaction curves did show differences amongst the sites: however, while two of the non-sacred forests approached a plateau at a much lower species richness than the other two, overall there was no evidence of a consistent difference between the six sacred groves and four non-sacred forests. While a large sampling effort was undertaken in this study, for all six sacred groves and two of the non-sacred forests the rarefaction curves are still far from the plateau when all sampled individuals were included, indicating that more species would be found in the area if more plots had been sampled. Therefore, the data do not allow a firm conclusion to be drawn about difference in species richness between the two forest types. However, tree species diversity (judged by inspection of Renyi profiles) was higher in the sacred groves than the non-sacred forests.

The analysis of similarity (ANOSIM) (Clarke and Warwick, 2001) of species composition between sacred groves and non-sacred forests showed significant differences. These differences in species composition were shown by percentage similarity analysis (PERSIM) (Clarke and Warwick, 2001) to be due to the presence of species that discriminated between the two forest types. For example, *Prunus africana, Celtis africana, Maytenus undata, Euphorbia candelabrum, Apodytes dimidata* and *Matenus arbutifolia* were more commonly found in the sampled sacred groves and *Hagenia abyssinica* was totally restricted to the sacred groves. On the other hand species such as *Lepidotrichilia volkensii, Chionanthus mildbraedii* and *Maytenus addat* were more commonly found in non-sacred forest and *Aguaria saxifolia* and *Olinia rochetiana* were restricted to non-sacred forest.

Ordination analysis provided evidence of major differences between the two forest types in species composition, which are likely to be due to (a) the higher rate of disturbance of the non-sacred forests, e.g. due to tree cutting (discussed below), and (b) topographic factors (Jin et al., 2008). Elevation, aspect and slope are the main factors that control the distribution and pattern of vegetation in mountain areas (Titshall et al., 2000). They determine the microclimate and thus large-scale spatial distribution and pattern of vegetation (Busing et al., 1992). The two forest types are different in slope as well as geology, from which their soil is derived (Chapter 2, Table 2.2) while they are similar in rainfall distribution (Chapter 2, Figure 2.6). The aspects and altitudes of the forests are also fairly different (Table 2.2). These differences in slope, altitude, geology and aspect may have all contributed to species composition differences of the sacred groves and non-sacred forests.

For all species combined, there were no significant differences between the sacred groves and non-sacred forests in tree basal area or in individual tree density. Inspection of the stand size-class distributions also showed that over most of the diameter size range (15-50 cm DBH) the absolute densities and distribution-shape (a gradual decline in numbers with size) was very similar between the two forest types. In addition, the densities of trees larger and smaller than that range did indicate major effects of human disturbance. The density of trees > 50 cm DBH (and > 30 m height) was notably higher in the sacred groves, however it was very clear that the density of trees 5-15 cm DBH (and 5-10 m height) was greater in the non-sacred forests. This difference was not reflected in the density of saplings which was also non-significantly higher in the sacred groves than in the non-sacred forests. However, the trend was reversed for seedling density which was significantly higher in the sacred groves.

6.6.2 Disturbance level

The average level of disturbance quantified as a combination of the impacts of tree cutting and grazing (Mehta et al., 2008) was the same in the sacred groves and non-sacred forests being moderately high in both. The quantification of the density of cut stumps as a proportion of all cut and uncut trees showed no overall difference between the two forest categories. This finding is incompatible with the interview response of the sacred grove custodians that people extract woody biomass in non-sacred forests without any restriction while they are culturally prohibited from entering the sacred groves to cut trees. These responses may reflect more the custodians' views on how things used to be rather than current reality. However, I observed a higher density of recently cut stumps of large trees in non-sacred forests than in sacred groves during the field work. As shown by the results (above) of a much higher density of live trees > 50 cm DBH in the sacred groves, while most of the large trees in non-sacred forests had probably already been removed for timber and other utilities (and their stumps no longer visible), many had been retained in the sacred groves, which is more compatible with the custodians stated perceptions. In the non-sacred forests, the past removal of large trees is likely to have caused the current lower basal area and more open canopy leading to the higher density of small trees. In the sacred groves, the high rate of selective cutting of small trees that had occurred (despite the custodians' statement) may have permitted the higher density of seedlings than the non-sacred forests. In addition, however, the seedling population in the sacred-groves was more dominated by species that could tolerate the higher level of canopy shade (and in some cases might even benefit from it in this seasonally dry climate (McLaren and McDonald, 2003)) than was the case in the non-sacred forests. Cattle grazing disturbance was common in all ten forests and is also likely to have had a major impact on tree species regeneration (through vegetation trampling and soil compaction as well as through browsing).

6.6.3 Land-use and land-cover change impacts on sacred and non-sacred forests in the Gamo highlands

Expansion of agricultural and settlement land-use has greatly reduced the land-cover of natural habitats in the Gamo highlands landscape over the past 15 years; reflecting the trends reported across the tropics by Lambin et al. (2003). This has led to fragmentation (a reduction in the spatial relationship between natural habitats) as the landscape spatial pattern has changed (Turner et al., 2001). Up to 1995 the landscape was dominated by natural/semi-natural habitats of woodland, wooded grassland and open-pastureland, forming the matrix for the comparatively small patches of household farms and settlements. Farms were usually located on gentle slopes and valley bottoms, while the steep slopes and flat plateaus were used as open communal pasture land. In contrast, by 2010 the area of farmland and settlement had expanded massively into land previously occupied by all three natural/semi-natural habitats and now forms the matrix of most of the landscape except for one strip dominated by wooded grassland in the west and one dominated by open pasture to the east of the centre (Figure 6.17). Virtually all the remaining forest patches are now within a matrix of farm and settlement land. This combination of a great reduction in the area of forest patches (through encroachment for agriculture and settlement) and the severe fragmentation of the patches due to being surrounded by this matrix of intensively used land-cover, which is likely to be impermeable to many forest species, is likely to pose a severe threat to the conservation of forest biodiversity in this landscape. This has led to an increase in the relative importance of the sacred groves for conservation as they have suffered a lower rate of area loss than the non-sacred forests. However, their capacity to provide this benefit is also highly threatened by fragmentation and the effect of the surrounding farmed and settled matrix.

The reduction in patch number within this landscape between 1995 and 2010 affected wooded grassland and open pasture land more severely than forest, however the overall area reduction across these habitat types has reduced the landscape’s habitat richness (McGarigal, 1995). Together with the reduction in the size and shape complexity of the patches of these natural/semi-natural habitats this is very likely to have reduced landscape-level biodiversity, indicating the urgent need for assessment of conservation priorities.

6.6.4 Importance of sacred groves for biodiversity conservation

The evidence from this study indicates the important role of sacred groves in biodiversity conservation in southern Ethiopia, supporting the many existing accounts from other countries (e.g. Mgumia and Oba, 2003; Bishra et al., 2005; Bhagwat et al., 2005; Rao et al., 2011). This quantitative ecological evidence was well-supported by the qualitative information obtained from interviews with local custodians about the protection afforded to the sacred groves by traditional practices, and their high cultural value. This high cultural value and protection is associated with the abundance in the sacred groves of a number of species that have become rare (through over exploitation) in other forest habitats, as has been found in other parts of Africa (Mgumia and Oba, 2003). These included tree species of national conservation priority, *Cordia africana* and *Hagenia abyssinica*, the globally threatened species *Prunus africana*, and a number of endemic tree species, e.g. *Vepris daniellii* and *Maytenus addat*.

Despite this national and international conservation value of the sacred groves, in interview very few of the custodians showed awareness of their wider conservation importance. Their protection practices are motivated by the indigenous cultural and spiritual value of these groves. While this represents a complex combination of values it does show the potential for a "win-win" outcome in which enhanced protection of the sacred groves both conserves local traditional culture (manifest in practices, taboos and value systems developed and enriched by the community over centuries) and wider biodiversity values (Claudia, 2008). The sacred groves are still considered to be a central part of life by many community members. The strength of these traditional values is evidenced by the fact that, even during the period of high political instability in the country over recent decades, when there were huge population, economic and social changes in the Gamo highlands, a high proportion of the sacred groves have been protected.

This apparent synergy between actions that would conserve both traditional culture and internationally important biodiversity, through protection of sacred groves, seems to present an obvious way forwards. However, it also presents a potential dilemma in terms of the appropriate role of different institutions. Up to the present, protection of the sacred groves of the Gamo highlands has entirely been the result of the actions of indigenous traditional institutions (predominantly the custodians). There has been no recognition of sacred groves by the official conservation institutions in Ethiopia. Clearly these cultural practices have been sustained as new generations have gained this knowledge through participation, observation and by oral traditions and eventually eldest sons have taken over this responsibility (as reported by the guardians), in common with many other traditional cultures (Berkes et al., 2000). However, the sustainability of this indigenous system appears to be under threat. The custodians themselves stated that this culture is being undermined by the growing strength of non-indigenous (Christian) religious beliefs in the area, and that an increasing number of young people are engaging in cattle grazing and tree cutting which is threatening the sacred groves. The forest inventory results of the present study provided empirical evidence for the high threat that such grazing and tree cutting is now posing to the six studied sacred groves. External national (government and NGO) conservation agencies could respond to this evidence by using existing instruments such as legal registration of remaining sacred groves as cultural heritage sites and attempting to enforce forest protection laws. However, these very externally-imposed actions might well serve to further undermine the status and authority within the community of indigenous traditional institutions, such as the custodians. Therefore, instead, I advocate carefully considered actions by external agencies, preceded by careful negotiation with community institutions, to determine if a truly balanced partnership can be achieved that will support the sacred-grove protection practices of the custodians (if appropriate through legal measures) and not risk undermining their status within their rapidly changing communities. The patience and care with which such a process must be conducted is essential despite the apparent urgency of the biodiversity conservation need indicated by the evidence presented in this study of the severe current threat posed to the sacred-groves through both deforestation and degradation.

CHAPTER 7

GENERAL DISCUSSION, CONCLUSIONS AND RECOMMENDATIONS

7.1 Introduction

The present research was designed to further our understanding of land-use change impacts in southwestern Ethiopia. In this thesis, the research was structured into four components that together were designed to address major gaps in knowledge of land-use change impacts on landscape pattern, forest patch spatial structure and consequent effects on species richness, composition and conservation value within the patches, as well as associated cultural values, in order that their conservation and management can be improved.

The first research component involved analyzing changes in landscape spatial pattern resulting from land use change (Chapter 3). In many countries landscape spatial pattern analysis has been found to be an important means of quantifying landscape change and thus used for forest spatial change modeling (e.g. Echeverria et al., 2007). However, this approach to landscape ecology research has had limited use in Ethiopia. It was applied in the present study in Ethiopia with the objective of developing forest restoration strategies jointly with communities. It assessed how the landscape pattern of forest patches changed over two decades, including the amount of core forest habitat area and its spatial distribution that could be used for identification of potential sites for restoration (Figure 3.28).

The second research component investigated the species composition of forest patches embedded within the increasingly agricultural-dominated landscape and its relationship with their fragmentation (Chapter 4). Particular attention was paid to the presence, abundance and distribution amongst patches of species of conservation value in order to assess the importance of small forest fragments in regional biodiversity conservation.

The impact of patch fragmentation and disturbance on forest structure, species composition and regeneration dynamics was assessed. This information is important to inform regional and national conservation strategy, in particular to draw the attention of conservation authorities in Ethiopia to the value of small forest patches distributed in agricultural landscapes.

The third research component explored whether and how plant species of different functional groups responded to forest patch fragmentation (Chapter 5). The functional groups' approach to analyse plant community response to fragmentation has been found in previous studies to provide an ecological understanding of the consequences of fragmentation (MacIntyre and Lavorel, 1994). This approach was adopted to inform conservation planning for small forest fragments in Ethiopia.

The fourth research component was focused on the value of sacred groves for biodiversity conservation, the cultural values that have led to their protection and the threats that they are currently under (Chapter 6). While the existence of sacred groves and their potential importance for biodiversity conservation in Ethiopia have been recognized (Desalegn, 2007) there is a serious lack of detailed information about their biodiversity and cultural importance. This need was met in the present study through comparison of the species composition and diversity of sacred with non-sacred forests. Linking to the previous two study components, the threat posed to sacred groves by land-use change and other pressures, linked to the status of community institutions, was also assessed. The findings of this research are important for conservation planning targeted at sacred groves, as well as assessing their role in wider regional conservation planning.

7.2 Discussion

7.2.1 Land-use and land-cover change

Consistent with studies in other parts of Ethiopia, which indicated a recent decrease in natural habitat resulting from expansion of land area under human use (Feoli et al., 2002), this study has revealed a very high rate of loss of forest cover in three different study areas in Illubabor Zone SW Ethiopia. In the lower altitude Site I, which started with much higher forest cover, it decreased 6.30% between 1986 and 2007 at an annual rate of 0.30%. There was a big forest loss in Site II which decreased 50.19% between 1986 and 2007 at a rate of 2.39% annually. Whereas there was a 9.45% decrease between 1986 and 2007 at an annual rate of 0.45% in Site III. Thus the two higher altitude areas which had previously lost a higher proportion of their forest cover were suffering the highest rates of deforestation, in one area so great that there is a serious risk that functional forest habitat will soon cease to be a component of its landscape. These forest area losses were associated with an expansion of land area under farming, grazing and settlement (Chapter 3). While crop land was the dominant land-cover class in high altitude areas its change over time showed inconsistent trends between periods as well as between sites. Grazing land showed a clear trend of increased area over the twenty-one years in all three sites, however its greatest expansion occurred at higher altitudes (reaching 42.7% of total land area in Site II and 32.4% in Site III by 2007). The expansion of grazing land at the expense of forest has also been found by previous studies in other continents (e.g. Echeverria et al., 2012). Amongst the three study sites, Site II experienced the highest rate of deforestation and had the highest percentage of farm and grazing land by 2007. In contrast the lowest altitude site (I) retained the highest percentage of forest cover (four times that of Site II and three times that of Site III). There are two possible explanations for this. Firstly, traditional coffee management at such mid-altitudes does not require complete clearance of existing forest vegetation (Aerts et al., 2011), in contrast to the dominant crop agriculture at higher altitudes. However, coffee management does lead to serious forest disturbance, with greatly reduced regeneration of tree species because of frequent slashing by farmers as they make the understorey dominated by coffee. However, some tree biodiversity is retained as coffee shade (Gove et al., 2008; Aerts et al., 2011). Secondly, the human population density is higher at higher altitudes in the Zone (CSA, 2007) leading to increasing pressure on land for food production.

The area of wooded grassland substantially increased in the lowest altitude site (I) while it greatly reduced in area in the two higher altitude sites. In Site I forest land is converted to wooded grassland due to the practice of fallowing, the creation of short grass fallow under remnant trees. In the higher altitude sites, the dominant process affecting wooded grassland is its conversion to crop agriculture. In general, total change of natural landscape to human land-use through deforestation, settlement expansion, agriculture and other human influences substantially altered the landscape pattern of the study area.

7.2.2 Landscape spatial patterns and forest fragmentation

Landscape metrics provided a useful tool to explore differences between the sites and change over time of landscape spatial pattern. The use of landscape and class level pattern metrics enabled assessment of the spatial structure of forest cover and its relation to other major land-cover types, i.e. farm and settlement, and grazing land (as had also been successfully carried out by Echeverria et al. (2007) in southern Chile). Forest fragmentation has three components at the landscape level: a) habitat loss; b) reduction of patch size; and c) increased isolation of patches (Forman, 1995; Bennett, 2003; Farina, 2007; Echeverria et al., 2007). All three of these components were shown to have occurred over the last two decades in all three study sites. Forests have undergone high rates of fragmentation, forest patches have been greatly reduced in area due to encroachment at their margin and in some places there has been a complete loss of patches as they are converted to other land-uses (Figure 3.28). However, the pattern of fragmentation varied between the three sites. In the two highland sites (especially Site II) the dominant trend was reduction in size of the (already small) remnant forest patches. In all three sites the landscape pattern became simplified over time: for all land-cover classes combined the mean patch size increased and patch number decreased. By the end of the study period the landscape cover of Site I remained dominated by forest, in Site III by farm and settlement, though in sites II a huge expansion in the area of grazing land brought it to co-dominance with farm and settlement land by the end of the study period.

The net shape complexity of the forests was more irregular in Site I, as indicated by its relatively high area weighted mean shape index (AWMSI) whereas the complexity was more regular in sites II and III indicating the greater human influence on landscape pattern at higher altitudes. Similarly, the edge density of the forests decreased in Site I while it increased in sites II and III indicating frequent disturbance at forest edges.

The mean patch size of core area for the two higher altitude sites reduced greatly over time. Core area is the patch area remaining after removing an edge buffer to represent the area not influenced by the edge effect (Bridge et al., 2000). A 50 m wide edge buffer was used in this study to delineate core area. The total core area of the forest habitat in all three study sites decreased over time indicating the decline in the total amount of core forest habitat at the landscape level. Reduction of core area reduces the viability of forests for providing adequate protection of species dependent on within-forest habitat conditions (Bridge et al., 2000), therefore this result gives serious concern about the level of threat to forest species in the Illubabor Zone.

7.2.3 Importance of small forest patches in biodiversity conservation in Illubabor Zone, Ethiopia

There were large differences in woody species composition amongst the three sites, especially between Site I at mid-altitude and sites II and III at higher altitude (Chapter 4). Different patterns of forest fragmentation and disturbance were associated with this altitudinal split into two groups of sites, so the study design did not allow clear separation between these potential causes. Canonical correspondence analysis showed that both altitude and disturbance (from grazing and tree cutting) were the strongest factors, while slope angle, patch shape, rainfall and patch edge density were less important in determining composition. Apart from its influence on rainfall, the basic impact of altitude on temperature is likely to have influenced patterns of vegetation distribution (Jin et al, 2008, Kreyling et al., 2010). In this study there are some species which were only recorded in study plot of lower altitude in Site I (*Anthocleista scheinfurthii*, *Pouteria altissima*, *Pappea capensis*, *Blighia unijugata*, *Vanguria* *apiculata, Trichilia dregeana, Morus mesozygia* and *Albizia grandibracteata*). This is in agreement with the findings of Friis (1992) that there are a set of species whose range is restricted to the transitional forest types situated at mid-altitudes in south-west Ethiopia.

While the relative contribution of different factors to the floristic differences between the three sites cannot be determined with certainty, the distinction between Site I and sites II and III was clearly strongly linked to forest patch size. The theory that "a large habitat area supports more species than small ones" diverts attention from the need for simultaneous conservation strategies that include the retention of small patches (Lindenmayer and Fischer, 2006). Protection of large forest reserves in modified landscapes is important for biodiversity conservation, but should not be the only strategy (Lindenmayer and Franklin, 2002). The present study established that small forest patches in Ethiopia contained large numbers of high conservation value species which is in agreement with the findings of studies in other tropical areas (e.g. Turner, 1996; Fischer and Lindenmayer, 2002; Arroyo-RodrÃguez and Mandujano, 2006). In fact the set of small forest patches in sites II and III had some species not recorded in continuous forest in Site I, including a higher number of conservation value species population (*Afrocarpus falcatus, Pouteria adolfi-friederici, Olea welwitschii*). This suggests that the forest patches at higher altitudes are crucial for maintaining the regional biodiversity of the Illubabor Zone. Furthermore, of the species that occurred in both larger forest blocks in Site I and smaller patches in Site II and III, most had a larger population density/plot in patches, such as for *Prunus africana, Elaeodendron buchananii, Erythrococca trichgyne* and *Apodytes dimidata* and for some species endemic to Ethiopia, *Vepris dainelii, Milletia ferrugina, Phyllanthus limmuensis* and *Maytenyus addat,* as well as afromontane endemics, *Ilex mitis, Bersama abyssinica, Olea capensis subsp macrocarpa, Canthium oligocarpum, Galineria saxfragia, Macaranga capensis* and *Ritchiea albersii*. This indicates the substantial threat to tree species conservation that would result from the loss or alteration of these small patches. Nonetheless, their capacity to act as effective habitats for conservation will greatly depend on them not becoming fragmented beyond a threshold at which they could no longer collectively support metapopulations of conservation value species (Rutledge, 2003).

The intensity of anthropogenic disturbance on sites can also be responsible for species distribution patterns that led to difference in species composition. The forests in Site I exprienced frequent human disturbance in which the timber tree species were extracted selectively (Feyera and Denchi, 2006) while trees in the understory were slashed for coffee management (Aerts et al., 2011). In the present study a high proportion of the woody plants recorded in Site I were of pioneer-type species that dominate the upper and lower canopy layers (*Milletia ferrugina*, *Albizia gummifera*, *Croton macrostachyus* and *Maesa lanceolata*) and whose establishment is promoted by forest disturbance (Hill and Curran, 2003). The composition and density of the sapling and seedling size classes was also dominated by these species, along with *Coffea arabica* actively promoted by management which was the most abundant species in the lower story (Aerts et al., 2011; Chapter 4). There are four possible reasons for this result: 1) the germination of other forest-habitat dependent species is greatly reduced by frequent substrate disturbance exposing their seed to predators; 2) the forest canopy is so severely disturbed that the understorey environment is unstable for the survival of other forest-habitat dependent species; 3) forest-habitat species are selectively slashed by farmers preventing them from attaining their full height (Aerts et al., 2011) and thus reproductive maturity or the capacity to set seed; 4) continuous human disturbance associated with coffee management may alter the forest environment to favor the growth of pioneer species (Tesfaye et al., 2010) in competition with the other forest-habitat species. A combination of these four factors may lead to the *priority effect* (Alford and Wilbur, 1985) where pioneer species are able to preempt the establishment of other species, competitively occupy the habitat and prevent the later invasion by forest-habitat species due to ongoing levels of disturbance.

Fragmented patches at higher altitude in Site II and III, in contrast, contain a mixture of species mostly limited to a higher altitudinal range (*Syzygium guineense*, *Afrocarpus falcatus* and *Pouteria adolfi-friederici*) and some with a wide altitudinal range in afromontane forests. The upper story of the patches was dominated by the first group (high altitude species). The lower story, including saplings and seedlings of the upper storey species, were dominated by forest-habitat species (*Chionanthus mildbraedii, Veperis dainelii*, *Oxyanthus speciiosus* and *Canthium oligocarpum*) and generalist species (*Syzygium guineense, Bersama abyssinica, Prunus africana* and *Pouteria adolfi-friederici*). However, the sampling of the edge habitat of these patches (Chapter 5) showed that the upper canopy was dominate by generalist species and the lower story dominated by species of pioneer types similar to Site I, suggesting that the edge effect through fragmentation and human disturbance can lead to a similar species composition to the generally disturbed/degraded coffee forests of Site I.

The lower number of woody plant species found in the smaller forest patches at high altitude compared with the larger continuous forest areas in Site I is in agreement with several studies that have found higher species richness in large continuous forest than in small forest patches (e.g. Turner, 1996; Cambin and Lachavanne, 2002; Schmitt et al., 2010). However, the individual-based species accumulation curves showed that the species richness was far from completely recorded for the forest patches despite the large sampling effort used in the present study, indicating that there were more species to be found in each of the three sites if sampling were continued. Although the curve showed an increment in species number at the end of the full sample in each site, species density (number of species per sampled plot) is still higher in the continuous forests of Site I (chapter 4; Figure 4.3) for 3200 m^2 sample plot size. The possible explanations for this difference might be that the sampling in Site I covered a greater range of habitat types than in sites II and III due to inherent differences in the environmental heterogeneity between the two areas. While the forests in Site I, predominantly under management for coffee production, were subject to high rates of disturbance, this was patchy within these large forest blocks. Because of the large block size, individual sampled locations were less isolated from other forest habitats than was the case for the sampled patches in sites II and III, making them more likely to be re-colonized after disturbance. The abundance of pioneer woody species in the sampled flora provided a set of species that could maintain high levels of species richness in these circumstances (Schmitt et al., 2010, Tesfaye et al., 2010). The other possible explanation for the higher species number in Site I is that the range of species could be restricted to this mid-altitudinal area (some of these are *Vangueria apiculata*, *Trichilia dregeana*, *Mimusops kummel*, *Deinbollia kilimandischarica*, *Manilkaria butugi* and *Pouteria altissima*).

Vegetation analysis along altitudinal gradient by Hamilton (1975) in Ugand forest also showed that the number of species decreased with increasing altitude. Though the regression analysis between altitude and species richness in this study indicated little relationship there should be an altitudinal effect at a level of individual species (Table 4.11).

Forest structure defined by tree basal area and individual tree density can be influenced by patch area (RodrÃguez and Mandujano-Arroyo, 2006), whereas Barbosa et al. (2010) found that patch size reduction due to anthropogenic habitat loss influenced tree basal area of the patches. In addition, a positive relationship between altitude and tropical forest basal area was found by Lieberman et al. (1996), Wangda and Ohsawa (2006) and Sahu et al. (2008). On the other hand, Coomes and Allen (2007), like many previous studies, showed a decline of tree diameter growth with altitude. However, in the present study, the sampled patches at higher altitude in sites II and III contained an equally high total density of individual trees and total tree basal area as the sampled blocks of forests at mid-altitude in Site I. Tree density and basal area showed high variability among the sampled patches within all three sites, which is likely to be due to variation in anthropogenic disturbance (cutting and grazing impacts on small trees, and timber harvesting impacts on large trees). The variability of basal area among patches within sites at higher altitude was lagely due to the density of large individual trees. The disturbance from tree cutting had not affected the patches equally in these areas, thus some patches still retained a considerable number of larger trees. The variability of individual tree density in patches can be due to cutting of many lower story shade trees for coffee management and trees of a range of sizes for construction purposes.

7.2.4 Response to Fragmentation of Plant species differing in life-form and habitat requirement

Species differing in life-form and habitat requirement can be affected differently by fragmentation (McIntyre and Lavorel, 1994; Annette and Martin, 2004). For example, the present study has shown differences in distribution between forest patch edge and interior of species differing in their previously-known habitat preference (Lindenmayer and Fischer, 2006; Chapter 5). In particular, species known to be associated with forest habitats (e.g. the woody species *Vepris dainelli, Chionanthis mildbraedii, Galineria saxfragia, Oxyanthus speciosus* and *Psydrax parviflora*) were rarer or absent in patch edges. This agrees with the findings of other studies of tropical forest, where fragmentation has been reported to restrict the establishment of shade tolerant species to patch interior habitat (Hill and Curran, 2003).

Changes resulting from land-use change, like encroachment into forest patches, reduction in forest habitat core area and spatially variable degradation within forest habitats due to increased grazing pressure and selective tree cutting, can all lead to quantitative changes in the distribution of shade tolerant species between edge and interior (Laurance, 2008; Barbosa et al., 2010). An important mechanism is increased light levels and temperature and reduced humidity in patch edge habitats due to canopy disturbance and lateral penetration of exterior microclimate (Laurance, 2008). As a corollary of this, the present study found that species known to be associated with (more open) non-forest habitats (e.g. the woody species *Maesa lanceolata*, *Albizia gummifera, Albizia schimperiana*, *Milletia ferruginea* and *Croton macrostachyus*) were more abundant in the edge than the interior of the forest patches (as had been found by Hill and Curran, 2003). A third functional group, generalists with wide-habitat tolerance, e.g. *Syzygium guineense*, *Prunus africana* and *Pouteria adolfi-friederici*, showed no variation in abundance between forest patch edge and interior habitat.

Despite this variation in species' relative abundance between patch edge and interior habitat no difference was detected in overall species richness of trees between the two habitats. However, when all woody species (trees, shrubs and vines) were combined species richness was higher in the edge habitat. This is attribuTable to the suitability of this habitat for a large number of shrub and vine species associated with non-forest habitats and present in the surrounding matrix.

In terms of forest structure, tree basal area was higher in the interior than the edge habitat (where there was a lower density of large trees). This may be a direct consequence of higher rates of selective tree cutting, and tree mortality in the edge habitat (Laurance, 2008; Gutierrez et al., 2008; Barbosa et al., 2010). A higher rate of tree mortality in patch edges has been attributed to a range of causes including canopy damage and tree fall caused by wind turbulence (Esseen and Renhhorn, 1998). Unlike basal area, no difference was found in overall individual tree density between edge and interior habitats. However, there was a large difference in the composition of tree density between the two habitats: as noted above, in the edge non-forest and generalist species were more abundant, whereas in the interior forest-specialist species were more abundant.

Multivariate analysis of the association of woody species distributions with individual environmental variables and fragmentation metrics showed a complex pattern. Overall the distribution of species amongst forest patches was most strongly linked to altitude and an index of forest disturbance, followed by fragmentation metrics (patch shape and edge density). However, the distribution amongst patches of individual functional groups did not show these associations (instead it appears that their distribution between edge and interior within patches (reported above) was a clearer response to fragmentation and forest disturbance).

Vascular epiphytes were found to be more susceptible to negative effects of fragmentation than were woody plants, which supports their value as indicators of variation in microclimate and human disturbance (Werner and Gradstein, 2009; Wyse and Burns, 2011). As these can be linked to forest edge conditions, epiphytes are expected to be sensitive to forest habitat fragmentation (Lobel et al., 2006). High temperatures, low humidity and lack of availability of host trees have all been found to limit survival of vascular epiphytes in forest edge habitats (Kersten et al., 2009). In the present study, as expected total species richness and individual density of vascular epiphytes was found to be lower in forest edge than interior habitats. A clear distinction was found between different vascular epiphyte functional groups in their edge/interior distribution. As anticipated, species previously known to be associated with forest-habitats were notably more abundant in the interior than the edge. Morphologically, these species generally have herbaceous stems and leaves (Appendix 5.6) which are associated with sensitivity to low humidity conditions. In contrast, species with a succulent stem and leaf morphology, as well as those with woody stems were similar in abundance between edge and interior habitats. However, the vascular epiphyte communities in interior habitats were dominated by three species: *Peperomia teteraphylla, Peperomia rotundifolia* and *Asplenium theciferum*

Epiphyte species composition and richness has also been found to be highly dependent on forest stand properties (Heizer, 1999, Hietz et al., 2006). In the present study a positive correlation was found between host tree diameter and vascular epiphyte individual density and species richness (supporting the findings of Hietz et al (2006)). Large diameter trees provide a suiTable substrate for epiphytes because of their branch structure and bark properties (Kersten, 2009), however there is also a big difference between tree species in these properties. In the present study the species *Syzygium guineense, Schefflera abyssinica*, *Ficus sur, Prunus africana*, and *Pouteria adolfi-feridricii* (all of which are widely distributed generalist species) were found to support a high individual density and species richness of forest-habitat-associated vascular epiphytes that were rare in the forest edge, as well as species such as *Peperomia tetraphylla* which are also abundant in edge habitat. The disturbance through selective tree cutting and the edge effect resulting from fragmentation will increasingly constitute a bottleneck for the persistence of vascular epiphytes in the face of ongoing habitat alteration.

Geophytic fern species richness was substantially lower in forest edge than interior habitat, in agreement with previous studies (e.g. Paciência and Prado, 2005; Silva et al., 2011). However, in contrast to Silvia et al. (2011) who found a similar density of fern individuals in edge and interior habitats, the present study found a higher geophytic fern density in the interior than the edge. There was also a noTable variation in community composition between the two habitats: 16 geophytic fern species (51.6%) had a frequency at least 2.5 times higher in interior habitats, of which four (*Asplenium formosum*, *Asplenium sandersoni*, *Thelypteris confluens* and *Conigramme africana*) were restricted to the interior, 10 species (32.3%) were approximately evenly distributed between the two habitats, just 5 species (16.1%) had a frequency at least 2.5 times higher in edge habitats, and only one species (*Pteridium aquilinum*) was found only in the edge. As expected, the species found to be more abundant in the interior habitat were predominantly those already known to be forest-specialist species, such as three species of *Asplenium*: *A. theciferum*, *A. gemmiferum* and *A. lunulatum*. However, species known to be more generalist in their distribution or associated with other habitats showed no difference in abundance between edge and interior.

Differences in frond and rhizome morphology were found between species more abundant in edge or interior habitats. Species with tufted fronds were more abundant in the interior while species with spaced fronds showed no trend of greater abundance in one habitat over the other. Species with creeping and with erect rhizomes were greatly more abundant in the interior habitat.

The higher geophytic fern individual density found in the forest interior habitat is very likely to be associated with its higher humidity (Silvia et al., 2011). Ferns are highly dependent on humidity for their sexual reproduction, since they use flagellated gametes for external fertilization. Therefore, fragmentation and disturbance of forest which reduces humidity is likely to lead to a gradual reduction in fern abundance (starting with the more humidity-sensitive forest specialist species, thus shifting community composition) due to a reduction in the rate of reproductive success.

7.2.5 Value of sacred groves in plant biodiversity conservation

The results of the present study provide strong evidence of the important contribution of sacred forests in the Gamo highlands of Ethiopia to conservation of plant biodiversity. Analysis of similarity showed a difference in species composition between sacred groves and non-sacred forests. The sacred groves were found to possess a slightly higher plant species richness (152 species in a total sample plot area of 2.24 ha) than did the non-sacred forests with which they were compared (142 species). In addition, the flora of the sacred groves contained a higher proportion of species endemic to Ethiopia (12.5%) than did the non-sacred forests (9.2%). Moreover, two tree species (*Cordia africana* and *Hagenia abyssinica*) listed as national priority species for conservation were recorded only in the sacred groves while one tree species (*Prunus africana*) included in the IUCN red list of threatened species was found to be much more abundant in the sacred groves than the non-sacred forests. A number of species were found only in the non-sacred forests and not in the sacred groves (e.g. *Aguaria saxifolia*, *Olinia rochetiana*, *Polyscias fulva*, *Schefflera volkensii*), however these are regionally commoner species of lower conservation value. Moreover species/individual accumulation curves showed an increasing trend at the limit of the full sample for all six sacred groves but only for two of the four non-sacred forests, suggesting that more complete sampling of the area would show a greater superiority of the sacred groves in species richness. This supports the findings of previous studies in other countries which also recorded higher species in sacred groves than in non-sacred forests (e.g. Ambinakudige and Sathish, 2009).

Due to the confounding of the geographical characteristics of the sites of forests in the Gamo highlands and their sacred status, the difference in species composition between the sacred groves and non-sacred forests cannot be attributed with certainty to sacred status. The distance between the two sets of sites ranges from 17 to 25 km and they differed in topographic position, which can be a major controlling factor in vegetation growth, pattern and spatial distribution in mountain areas, specifically aspect and slope (Busing et al., 1992; Dawes and Short 1994; Titshall et al., 2000). The two site areas also differed in geology and thus probably in soil type. However, there was no appreciable difference in rainfall and elevation.

The surrounding land matrix could also have an influence on species composition within the patch-like sacred groves. The sacred groves are completely isolated within an agricultural landscape (Chapter 6, Figure, 6.17) and land-use activities around them may have an impact through edge effects, or reduction in patch area where their protection is not completely effective. Assessment of fragmentation of the individual sacred grove patches showed that the area of four was reduced over 15 years, while two increased. Over the same period the shape complexity of one increased while two decreased and the remaining three showed no noTable change. The edge density of all six sacred groves increased, indicating that there was frequent disturbance around their edges which impacted on the groves. For all four non-sacred forests the area and the shape complexity decreased over the same period, while the edge density of two non-sacred forests increased and two reduced.

These results show that the non-sacred forests in the Gamo highlands are under a high level of threat of habitat loss and fragmentation. The situation for the sacred groves is more mixed. While Gamo religious leaders and community elders still adopt traditional sacred values, cultural taboos, belief systems and bylaws to protect sacred groves (as has also been observed in many other societies around the world (Bhagwat and Rutte, 2006)), in practice this is not effectively ensuring that the status of all of them is preserved. Despite the continuation of these traditional values and institutions, many of the sacred groves are under threat, reflecting the complex economic, social and cultural changes currently taking place in the Gamo highlands. Thus, the sacred groves were degraded due to grazing and some tree cutting. No active management from enrichment tree planting or restoration was observed or reported. This may be due to the associated taboo which prevents digging sacred ground by sharp metal implements.

Another challenging factor within the communities that is likely to have threatened the sacred groves is the increasing stratification of the communities into different religious groups. Most of the newly formed religious groups are dominated by employees of government agencies, with many decisions made by them without considering their impacts on the less privileged traditional group (led by the custodians of the sacred groves). This has an impact on the ownership and decision-making rights of the custodians on the sacred groves. Traditional practices associated with the sacred groves have become increasingly limited to the older generation since most members of the younger generation not willing to learn, according to the custodians. The transmission of culture to the younger generation is likely to be hampered due to educational systems that do not acknowledge local value systems, alienating youth from local culture (Claudia, 2008). This, in turn, has contributed to the undermining of traditional institutions by local government agents exposing them to a dominant religion that is seriously impacting on local value systems and conservation practices.

7.3 Conclusion

The use of a landscape approach in this study has allowed the assessment of change in the spatial structure of habitats associated with forest fragmentation at the spatial scale that is most relevant for many aspects of biodiversity conservation and enactment of policy on forest management. The results obtained, and in particular the information derived by the landscape metrics, identified the changes in forest habitat spatial structure that have taken place over the past two decades across the whole Illubabor Zone. The land-cover change analysis showed that landscapes across the zone are becoming increasingly dominated by farm and settlement land-use, and grazing at higher altitudes. At lower altitudes, while forest cover remains above 50%, this forest has become notably more fragmented. These patterns of land-use change have had a major effect on the spatial structure of the forest remnants in both altitudinal areas, resulting in smaller fragmented patches that are more isolated, less well spatially connected and exposed to a greater edge effect. This fragmentation is the result of socio-economic drivers, such as increased demand for land for crop and livestock production, and for settlement, all of which have led to deforestation. These impacts have been supplemented by socio-economically driven degradation of the remaining forest (though its dominant mechanism varies with altitude, between lower areas suited to understorey coffee production and higher areas farm expansion). The underlying causes, however, include increases in human population density, changes in the socio-economic characteristics of the resident population (Schulz et al., 2011), external economic drivers (such as investment in cash crop cultivation) and changes in effective tenure and management responsibility for forests (Tadesse, 2007; Desalegn and Yilma, 2008).

The results clearly show the value of forest patches for plant biodiversity conservation (Chapter 4). Examples of conservation priority species such as afromontane endemics, species endemic to Ethiopia, other national priority species, internationally vulnerable species and a wider set of species with a narrow habitat range restricted to forests were all found in the studied forest patches. The presence of wild coffee (in high abundance in some patches) also makes them important for the conservation of coffee genetic resource (Labouisse et al., 2008). However, these priority species are restricted to (or found in the highest abundance) in interior core areas (away from the edge zone) of the forest patches. Therefore their populations will be threatened by further fragmentation, and future survival of these species within the studied landscapes will be depend on effective conservation of the patches' core area. Contrary to what would be expected from early biogeography theory (MacArthur and Wilson, 1967), a forest does not necessarily have to be large to have a high conservation value, provided that it is well connected to sufficient other patches within the landscape to allow adequate gene flow to enable the survival of metapopulations of conservation priority species (Braunisch et al., 2010). Therefore, there is a need to promote conservation of all forest patches within a landscape, even those that currently have low biodiversity value, through protection, restoration and management. Inspection of the results of the present study showed no evidence of spatial trends of the conservation value of the species composition patches in the landscape. Therefore, it provided no basis for prioritizing particular patches/areas for conservation action on this basis. In addition, the results provided no basis to prioritize patches for conservation simply on their area as it showed that even small patches can harbour conservation priority species that were missing in some of the larger forest patches (Chapter 4, Appendix 4.2 & 4.3). Therefore, forest patches cannot be ignored in conservation policy and practice simply on the grounds that they are currently small in area. Nonetheless, current developments in theory and evidence of forest habitat networks (e.g. Quine et al., 2002; Jacquemyn et al. 2003), which were not tested in the present study, show the great potential for spatial prioritization of habitat patches, less on the basis of their current species, composition but more on the basis of their position relative to other habitat patches, the extent to which they would (as "stepping stones" increase the connectivity of other forest patches, and thus contribute to a habitat network maximizing the potential for survival of metapopulations of forest-dependent vulnerable species.

Fragmentation results in an increase in the proportion of forest area that is in the proximity of a patch edge (the "edge zone"), and the results of this study showed strong evidence of the large impact of this edge effect on plant species composition, causing differential effects between different functional groups. Species known to be most dependent on forest habitats, and those with morphological characteristics likely to give the least tolerance of desiccation were the most negatively affected. These findings indicate the importance of preventing the reduction in area of those forest patches that are currently sufficiently large to include an interior (core) area that is big enough to support populations of such vulnerable species. On the other hand, within the larger forests, the human disturbance through coffee management altered the plant species composition and diversity more than species richness. However, in line with other findings (Tadesse et al., 2008; Gove et al., 2008; and Aerts et al., 2011) these forests still have biodiversity value. Where these forests are subject to high levels of degradation (e.g. associated with coffee cultivation) there is a conservation priority to reduce degradation in at least some parts of the interior area sufficient to meet the habitat needs of conservation priority species. Coffee management practices by farmers are mainly carried out in the forest edge zone in larger patches, and this zonation gives potential for protection of the interior habitat for strict biodiversity conservation which would also protect a coffee gene pool for cultivated coffee.

The landscape of Site I, in contrast to the higher altitude areas, was characterized by a forest-dominated mosaic with a few forest species retained by farmers as shade in coffee farms outside the forests as well as some fruit trees and hedgerows managed in and around individual farms (per. Obs.). This management of different tree species for different uses can increase the number of species in the farmland (e.g. Tolera et al., 2008). Some of these forest trees maintained in the farmland harbour some vascular epiphyte species (pers. obs, and as found by,e.g., Hylander et al., 2008) because they are still functionally connected to forests in the vicinity which influence temperature and moisture in the air to create favorable conditions for some epiphytes. However, most of these epiphytes are likely to be forest-margin or wide-habitat species with less conservation value. Although conversion of forest to coffee cultivation may not affect the number of woody plant species (Tolera et al., 2008) it is likely to alter species composition with the loss of forest-dependent and vulnerable species. Therefore, protection of forest interior areas will still be required for their conservation (Cayuela et al., 2006c).

In the higher altitude areas, in contrast to the lower altitude, forest patches tend to be isolated from each other (Chapter 3). Some patches are used for coffee production in their edge zone while it is others that have tended to be most subject to deforestation for cultivation of other agricultural crops (pers. obs.). Households manage coffee cultivation in patches adjacent to their dwelling place starting from forest edge inwards up to a certain distance. The interior area beyond this intensively managed forest edge zone is used for collection of fuelwood and other non-timber forest products (NTFPs), in contrast to Site I where it is used more for collection of coffee berries from unmanaged coffee trees (Desalegn and Yilam, 2008). In most of the patches in sites II and III there are rarely any natural coffee plantations, associated with their higher altitude (Teketaye, 1999). The interior area of these patches could be protected as a conservation zone while maximizing coffee production, if necessary, in the edge zone or diversifying agroforestry systems (including coffee) in the surrounding farmland that is managed by local farmers. This should help to protect the remaining patches from degradation and complete habitat loss, provided land tenure meets the livelihood needs of all households and they have sufficient alternative sources of fuelwood, construction wood and other destructively harvested NTFPs.

Sacred groves are the forests most strongly protected by the traditional institutions of local communities in the Gamo highlands. The present study showed clearly the importance of these forests as habitats for biodiversity in the agricultural dominated landscape of this area, where most forests are now plantations of exotic species of

Eucalyptus and *Cupressus* (per.obs.). These forest patches, their flora and their associated culture are becoming increasingly threatened in this area by change in land use (expansion of farmland, grazing land) and cultural institutions (the reduction in the status and influence of the traditional belief systems and their custodians within the local communities). Measures need to be taken at a local level to empower local traditional institutions and facilitate community participation in protection and management of sacred groves. For custodians to effectively conserve and manage their sacred groves and freely exercise their traditional practice, they must have a sense of custodianship toward their forests. This develops through the diffusion of awareness within communities (especially between generations) and the empowerment of custodians to maintain control over sacred groves; as well over developmental and other political processes affecting the communities. In the face of new political, cultural and economic pressures, custodians may also need effective collaboration with non-governmental and governmental organizations that should play an active but equal role, that of a facilitator rather than a dominating ruler. Conservation NGOs in particular can play a significant role in contributing at community discussion forums, by raising awareness and introducing information about the use of sacred groves from ecological and cultural perspectives.

7.4 Recommendations

Recommendations made for forest biodiversity conservation often focus on the protection of the single largest remaining areas of forest (Turner et al., 2001). Less attention has been given to conservation and management of many small forest patches at the landscape level. There is increasing evidence that the presence of large numbers of forest patches in a landscape is of great importance in maintaining its biodiversity (Lindenmayer and Fischer, 2006). The results of the present study have shown how the presence of a large number of forest patches in the landscape of the Ilubabor Zone may be of great importance for the conservation, to date, of its forest biodiversity. These small patches contain a high proportion of the conservation priority and vulnerable species, relative to the remaining large forest areas (Chapter 4). The new national conservation policy should therefore be structured in such a way as to include the conservation of forest patches.

In conservation practice, the maintenance of large areas of continuous forest cover should be concomitant with complementary measures such as protection and management of small forest patches (Lindenmayer and Fischer, 2006). Thus the patches surrounded by an agricultural matrix should also receive conservation attention. A strong emphasis should be put on reduction of encroachment and within-habitat disturbance of the patches in the higher altitude landscape and fragmentation and disturbance of larger forest areas in the mid-altitude landscape. Patches in the higher altitude landscape are currently exposed to harsh environmental conditions in the edge zone that are likely to harm forest-dependent species. In Chapter 5, major changes in plant species richness were reported between the forest interior zone (more than 50 m from the edge) and the edge zone. Vascular epiphytes and geophytic ferns often showed a higher richness and density in the interior zone. Thus focusing forest protection on the inner zone of patches, while maintaining forest cover in the edge zone to buffer them is important for conservation of forest-dependent species. The distinct edge zone habitat, being suiTable for other species, will contribute to overall biodiversity. Within the edge zone, promoting forest restoration, e.g. through reducing disturbance and encouraging tree natural regeneration, can enhance the size and shape of forest patches, the linkages between isolated patches and, therefore, the resilience of forest habitat at the landscape scale, and metapopulations of key forest species, to external disturbances and to land degradation. Chapter 3 reported that the most important direct causes of deforestation and degradation in Illubabor Zone include the conversion of forest lands to crop agriculture and grazing land. The forces behind these activities are related to the poor state of the local economy that has been heavily influenced by recent major socio-cultural change through high rates of immigration. Therefore, conservation and management plans must address the social and economic factors that are leading to patch degradation and deforestation.

The current economically-driven threats to the sacred grove of the Gamo highlands also involve competing land uses (grazing) and over-exploitation of forest resources (tree cutting). However, here an additional important factor is that they are inextricably linked to the declining strength of the traditional community institutions that have, hitherto, served to protect the sacred groves from such threats. Given the growing influence of, e.g., new religions as well as other major cultural and social changes in the area, and lack of backing from local government agencies, it is unlikely that the traditional institutions will retain sufficient power to protect the sacred groves in the future. The taboo against digging sacred ground by hoe or any sharp metal implement, even if used to prepare ground for tree planting, remains a major constraint on restoration of forest within the sacred groves. On one hand, most of the sacred groves are lacking core forest habitat area due to their small size. On the other hand, even if the patches have enough size to contain a core area, interviews with the custodians of the sacred groves revealed that it is culturally prohibited to divide sacred grove into edge and core area, e.g. to recommend the edge zone for sustainable use and core area for strict conservation. Therefore, instead, the establishment of a forest buffer zone around the remaining sacred groves is recommended. Forest established in this buffer zone could be managed for multiple uses, e.g. through agroforestry by the farmers owning the land. Where there is not the local demand for such multiple use, but grazing pressure is severe, land around sacred groves could be fenced to facilitate forest restoration, if necessary with the use of active reclamation techniques. Extrapolating from the results obtained in the Illubabor Zone (Chapter 5) it is anticipated that the microclimatic impact of this buffer zone would improve habitat quality within the existing sacred grove for sensitive forest-dependent species, and may increase the rejuvenation of their populations through natural regeneration. In addition, analysis of the landscape context of sacred groves may indicate those for which creation of new woodland patches within the matrix with similar species composition may increase the connectivity between sacred groves and improve the potential for survival of woodland species' metapopulations.

The tenure status of some sacred groves conflicts with certain government interests since local government authorities consider them as state forests and according to the custodians, have allocated them to people who oppose the traditions upheld by the custodians, without even consulting the custodians about this decision. This would obviously undermine the authority of the traditional custodians to protect these forests. To counter this tendency in order to conserve both cultural and biological diversity would require the empowering of local institutions and custodians to maintain their traditional authority and control over the sacred groves, and thus strengthen the existing traditional rules about the use, management, and conservation of these forests. This will require full recognition of the tenure rights of the custodians over the sacred groves and acceptance of their practices by the local government authorities. Education and other advocacy approaches by governments and NGOs are required to try and improve the understanding of the ecological and cultural values of the sacred groves amongst the many religious organizations, groups and individuals that are newly active within the, hitherto, very traditional communities of the Gamo people. While it may be unrealistic to expect these new religions to become active advocates of the protection of the sacred groves, it will be important at least, to discourage them from actively undermining their conservation.

7.5 Limitation of the study

The use of images with different spatial resolution and taken at different times in the year, could lead to misleading results and interpretation. Furthermore, the low spatial resolution of Landsat images limited the analysis to a few land cover classes and, as a consequence, allowed the investigation of few spatial processes related to fragmentation.

Although the study considered many environmental factors, there are still some which could not be assessed due to limitations of data availability and research budget. For instant, analysis of soil properties would have provided further useful information on site factors influencing species composition.

This study compared patches in upper altitude landscapes with large unfragmented forest located in a lower altitude landscape. This confounding is likely to have influenced the results and interpretation. If the actual location of forests of different types within the landscape had permitted, it would have been better if both the sampled large and the small forest patches were located with the same site conditions in the same landscapes, or at least those that did not have such a major confounding variable. However, such is the spatial determination of patterns of deforestation that such a situation rarely occurs in reality.

7.6 Suggestion for future research

This study has not taken into account connectivity or isolation between patches, either the linear features such as corridors that connect patches or detailed characteristics of the matrix between patches. According to the metapopulation model (Hanski and Gilpin 1992; Hanski, 1999) smaller patches may support a viable population if they are well connected. Even though a small mapping unit size (0.5 ha) was used in the image analysis in the present study to include small habitats and linear corridors it was difficult to identify them in the map hindering analysis of connectedness between patches. Satellite imagery with finer spatial resolution should be used in the future to measure the linear connections between patches and describe the nature of the matrix land between patches.

In this study, functional grouping of species focused on plant life form, morphological characteristics and previously reported distribution of species between different habitat types. While this provided valuable preliminary information about the characteristics of species that were most negatively affected by forest fragmentation and degradation, new research is needed that uses much more sophisticated (and truly “functional”) measures of species’ functional traits.

The impact of fragmentation on annual vascular herbaceous plant species was not covered in great detail in this study due to the field work being restricted to the dry season and research in the future should consider both seasons to record the full range of species present in forest patches.

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APPENDICES

Appendix 3.1 Patch spatial characteristic of Site I during study period: Nump= Number of patch, MPS= Mean patch size (ha),MSI= Mean shape index, AWMSI= Area-weighted mean shape index, AWMPFD= Area-weighted mean patch fractal dimension, TE= Total edge (m), ED= Edge density, PSCoV= Patch size cofficient of variation, PSSD= Patch size standard deviation, TLA= Total land area, CA= class area

Site I 1986	NumP	MPS	MSI	AWMSI	AWMPFD	TE	ED	PSCoV	PSSD	TLA	CA
Forest	3297	8.2	1.4	17.1	1.5	3169799.7	66.5	3525.3	288.5	47647.7	26980.2
Grazing land	2176	1.4	1.5	3.1	1.4	1315892.9	27.6	365.7	5.2	47647.7	3114.2
Wooded grassland	14	0.7	1.4	1.6	1.4	5661.1	0.1	79.9	0.5	47647.7	9.2
Farm and settlement	6095	2.9	1.4	27.4	1.5	4413668.7	92.6	3978.4	114.6	47647.7	17550.9
Site I 2000											
Forest	1545	21.5	1.4	17.0	1.4	1979936.0	41.5	3469.7	746.0	47647.7	28005.9
Grazing land	2346	1.0	1.5	2.2	1.4	1148448.1	24.1	206.3	2.1	47647.7	3353.0
Wooded grassland	147	0.7	1.4	1.7	1.4	58897.9	1.2	122.5	0.9	47647.7	310.7
Farm and settlement	3357	3.6	1.4	12.4	1.5	2572220.1	54.0	2179.6	77.8	47647.7	15978.1
Site I 2007											
Forest	668	37.9	1.7	20.0	1.5	2487891.9	52.2	1557.1	589.8	47647.7	25300.4
Grazing land	1388	3.4	1.7	4.8	1.4	1509430.6	31.7	529.7	18.2	47647.7	4776.6
Wooded grassland	206	0.9	1.5	1.6	1.4	96842.1	2.0	91.0	0.8	47647.7	183.3
Farm and settlement	1890	9.2	1.7	25.1	1.5	3833856.5	80.5	2139.4	196.9	47647.7	17394.2

Appendix 3.2 Patch spatial characteristic of Site II during study period: Nump= Number of patch, MPS= Mean patch size (ha), MSI= Mean shape index, AWMSI= Area-weighted mean shape index, MPFD= Mean patch fractal dimension, AWMPFD= Area-weighted mean patch fractal dimension, TE= Total edge (m), ED= Edge density(m/ha), PSCoV= Patch size cofficient of variation, PSSD= Patch size standard deviation, TLA= Total land area (ha), CA= class area (ha)

Site II 1986	NumP	MPS	MSI	AWMSI	MPFD	AWMPFD	TE	ED	PSCoV	PSSD	TLA	CA
Forest	2127	6.7	1.6	6.8	1.4	1.4	2357511.7	22.9	1304.9	87.0	47647.7	14186.4
Grazing land	2983	3.3	1.6	4.8	1.4	1.4	2903747.4	60.9	537.8	17.7	47647.7	9817.3
Wooded grassland	1432	0.5	1.4	1.7	1.4	1.4	497244.1	1.4	136.8	0.7	47647.7	778.4
Farm and settlement	2839	8.1	1.4	46.6	1.4	1.5	5339647.2	112.1	3259.2	262.5	47647.7	22865.5
Site II 2000												
Forest	3380	3.7	1.5	5.3	1.4	1.4	2604878.0	49.5	1278.9	47.9	47647.7	12664.2
Grazing land	4402	1.2	1.5	2.3	1.4	1.4	2437818.6	96.7	210.1	2.5	47647.7	5287.4
Wooded grassland	144	1.0	1.4	2.0	1.4	1.4	64957.7	10.4	253.6	2.4	47647.7	137.4
farm and settlement	2497	11.8	1.3	50.8	1.4	1.5	4152571.2	87.2	4564.6	540.3	47647.7	29558.7
Site II 2007												
Forest	940	7.5	1.6	4.1	1.4	1.4	1092547.8	54.7	821.0	61.6	47647.7	7052.0
Grazing land	2168	9.4	1.7	17.4	1.4	1.5	4609847.2	51.2	1471.9	138.2	47647.7	20351.5
Wooded grassland	167	1.0	1.4	1.6	1.4	1.4	78961.1	1.7	106.2	1.0	47647.7	158.8
Farm and settlement	1823	11.0	1.7	15.6	1.4	1.5	3838254.9	80.6	1538.5	169.5	47647.7	20085.4

Appendix 3.3 Patch spatial characteristic of Site III during study period: Nump= Number of patch, MPS= Mean patch size (ha), MSI= Mean shape index, AWMSI= Area-weighted mean shape index, AWMPFD= Area-weighted mean patch fractal dimension, TE= Total edge (m), ED= Edge density (m/ha), PSCoV= Patch size cofficient of variation, PSSD= Patch size standard deviation, TLA= Total land area (ha), CA= class area (ha)

Site III 1986	NumP	MPS	MSI	AWMSI	AWMPFD	TE	ED	PSCoV	PSSD	TLA	CA
Forest	1639	6.4	1.5	6.4	1.4	1568321.6	33.1	1298.1	83.3	47647.7	10522.5
Grazing land	3211	2.9	1.6	4.0	1.4	2982438.4	62.9	456.7	13.3	47647.7	9588.9
Wooded grassland	2397	0.8	1.4	2.3	1.4	1000631.7	3.5	292.0	2.2	47647.7	1841.0
Farm and settlement	2851	9.0	1.4	27.5	1.6	5118327.9	108.0	4677.9	421.6	47647.7	25695.3
Site III 2000											
Forest	2458	5.4	1.4	6.6	1.4	1976173.9	24.8	1495.1	80.8	47647.7	13284.3
Grazing land	6653	0.9	1.4	2.7	1.4	3134964.4	94.5	340.1	3.2	47647.7	6579.3
Wooded grassland	425	0.7	1.4	1.8	1.4	163866.8	21.1	184.0	1.2	47647.7	282.6
Farm and settlement	1635	16.8	1.4	35.9	1.5	3272621.9	69.1	3355.4	564.4	47647.7	27501.5
Site III 2007											
Forest	766	12.4	1.6	5.6	1.4	1175583.0	41.7	899.6	111.9	47647.7	9530.2
Grazing land	2933	5.3	1.7	10.3	1.5	4479011.8	66.2	1043.0	54.9	47647.7	15439.6
Wooded grassland	506	1.1	1.5	1.6	1.4	262618.7	5.5	109.8	1.2	47647.7	802.3
Farm and settlement	2162	10.1	1.7	25.5	1.5	5355988.0	113.0	1573.8	159.2	47647.7	21875.5

Appendix 3.4 Landscape spatial characteristic of all study sites of each year: Nump= Number of patch, MPS= Mean patch size (ha), MSI= Mean shape index, AWMSI= Area-weighted mean shape index, SDI= Shannon diversity index, SEI= Shannon eveness index, ED= Edge density (m/ha), PSCoV= Patch size cofficient of variation, PSSD= Patch size standard deviation, TLA= Total land area (ha)

	Class	NumP	MPS	MSI	AWMSI	SDI	SEI	ED	PSCoV	PSSD	TLA
Site I											
1986	All	11582	4.11	1.40	20.00	0.87	0.63	186.87	4252.12	174.95	47647.68
2000	All	7395	6.44	1.42	15.11	0.96	0.55	120.86	5354.56	345.06	47647.68
2007	All	4152	11.48	1.66	20.29	0.76	0.69	166.36	2367.86	271.77	47647.68
Site II											
1986	All	9381	5.08	1.51	25.42	1.11	0.80	232.92	2964.89	150.59	47647.68
2000	All	10423	4.57	1.46	33.17	0.91	0.66	194.35	5817.00	265.92	47647.68
2007	All	5098	9.35	1.67	14.61	1.03	0.74	201.89	1478.53	138.19	47647.68
Site III											
1986	All	10098	4.69	1.47	37.35	1.11	0.80	225.18	4830.62	226.67	47647.68
2000	All	11171	4.24	1.44	23.05	0.97	0.70	180.39	5170.19	219.30	47647.68
2007	All	6367	7.44	1.68	16.26	1.10	0.79	237.88	1441.89	107.32	47647.68

Appendix 3.5: Core area characteristic of forest in each site during study period : Nump= Number of patch, MPS= Mean patch size (ha), AWMSI= Area-weighted mean shape index, AWMPFD= Area-weighted mean patch fractal dimension, TFA= Total forest area (ha), TCoA= Total core area (ha), %CA= Percent of Core area, CAD= Core area density, ED= Edge density (m), PSCoV= Patch size coefficient of variation, PSSD= Patch size standard deviation.

	NumP	MPS	AWMSI	AWMPFD	TFA	TCoA	% CA	CAD	ED	PSCoV	PSSD
Site I											
1986	733.0	24.5	7.6	1.4	26980	17925.7	66.4	2.9	51.0	1892.3	462.8
2000	328.0	81.0	7.3	1.4	28005.9	26553.4	94.8	1.0	34.3	1168.6	946.0
2007	622.0	24.8	14.3	1.4	25300.4	15399.4	60.9	2.5	94.1	1493.6	369.8
Site II											
1986	697.0	10.4	3.3	1.3	14186.4	7256.8	51.1	5.4	89.1	834.9	86.9
2000	667.0	8.8	3.0	1.3	12664.2	5890.7	46.5	6.1	89.7	817.5	72.2
2007	385.0	9.5	2.7	1.3	7052	3647.1	51.7	6.0	93.0	733.8	69.5
Site III											
1986	351.0	17.1	3.4	1.3	10522.5	6002.5	57.0	3.6	76.1	663.2	113.4
2000	410.0	18.8	4.2	1.3	13284.3	7720.1	58.1	3.3	74.2	712.1	134.1
2007	375.0	14.7	4.0	1.3	9530.2	5524.4	57.9	4.2	85.5	717.3	105.7

Appendix 4.1 Importance value of tree species in Site I. BA = Basal area, RDO = Relative dominance, D = Species Density, RED = Relative Density, F = Frequency, RF = Relative frequency and IVI = Importance Value Index

Species	BA	RDO	D	RED	F	RF	IVI
Millettia ferruginea	27.85	10.14	64.8	10.38	0.73	4.88	8.46
Coffea arabica	3.08	1.12	103.5	16.56	0.84	5.60	7.76
Albizia gummifera	21.72	7.91	21.1	3.38	0.42	2.80	4.70
Celts africana	25.87	9.42	5.1	0.81	0.20	1.35	3.86
Croton macrostachyus	12.71	4.63	18.8	3.00	0.44	2.91	3.51
Maesa lanceolata	8.93	3.25	39.5	6.31	0.11	0.73	3.43
Bersama abyssinica	4.59	1.67	23.4	3.75	0.47	3.11	2.84
Ehertia cymosa	5.20	1.90	19.5	3.13	0.45	3.01	2.68
Chionanthus mildbraedii	2.14	0.78	23.8	3.81	0.44	2.91	2.50
Sapium elliptcum	14.93	5.44	2.3	0.38	0.25	1.66	2.49
Maytenus undata	5.33	1.94	27.0	4.31	0.17	1.14	2.46
Trilepisium madagascariense	9.92	3.61	8.2	1.31	0.34	2.28	2.40
Pouteria adolfi-friederici	9.42	3.43	7.8	1.25	0.31	2.08	2.25
Ficus sur	9.51	3.46	6.3	1.00	0.33	2.18	2.21
Vepris dainellii	1.80	0.65	21.5	3.44	0.38	2.49	2.19
Oxyanthus speciosus	2.71	0.99	18.4	2.94	0.38	2.49	2.14
Syzygium guineense	5.34	1.94	12.5	2.00	0.33	2.18	2.04
Mimusops kummel	11.04	4.02	9.0	1.44	0.08	0.52	1.99
Cordia africana	6.70	2.44	7.4	1.19	0.30	1.97	1.87
Psychotria orophila	2.11	0.77	16.4	2.63	0.25	1.66	1.68
Macaranga capensis	4.72	1.72	7.0	1.13	0.33	2.18	1.67
Trichilia dregeana	5.51	2.01	5.9	0.94	0.28	1.87	1.60
Argomuellera macrophylla	1.05	0.38	20.7	3.31	0.13	0.83	1.51
Rothmannia urcelliformis	1.22	0.44	7.0	1.13	0.41	2.70	1.42
Ritchiea albersii	2.45	0.89	8.6	1.38	0.28	1.87	1.38
Elaeodendron buchanani	4.73	1.72	5.9	0.94	0.17	1.14	1.27
Canthium oligocarpum	2.30	0.84	7.4	1.19	0.27	1.76	1.26
Dracaena fragrans	1.12	0.41	8.2	1.31	0.27	1.76	1.16
Galineria saxifraga	0.88	0.32	7.0	1.13	0.30	1.97	1.14
Ficus vasta	7.12	2.59	0.8	0.13	0.06	0.42	1.04
Diospyros abyssinica	0.80	0.29	4.7	0.75	0.25	1.66	0.90
Cassipourea malosana	0.88	0.32	3.5	0.56	0.27	1.76	0.88
Ficus vallis-choudae	4.67	1.70	2.0	0.31	0.08	0.52	0.84
Allophyllus abyssinicus	1.80	0.66	3.9	0.63	0.19	1.25	0.84
Psydrax parviflora	0.47	0.17	3.5	0.56	0.23	1.56	0.76
Pappea capensis	1.62	0.59	4.3	0.69	0.11	0.73	0.67
Albizia grandibracteata	1.39	0.51	1.6	0.25	0.19	1.25	0.67

Vangueria apiculata	0.84	0.31	2.7	0.44	0.19	1.25	0.66
Polyscia fulva	3.28	1.19	1.6	0.25	0.08	0.52	0.65
Garcinia buchananii	1.16	0.42	2.3	0.38	0.17	1.14	0.65
Scheflera abyssinica	3.19	1.16	0.8	0.13	0.09	0.62	0.64
Prunus africana	3.13	1.14	1.6	0.25	0.08	0.52	0.64
Ficus mucuso	2.98	1.09	2.0	0.31	0.06	0.42	0.60
Euphoriba ampliphylla	3.51	1.28	2.3	0.38	0.02	0.10	0.59
Blighia unijugata	1.36	0.50	2.7	0.44	0.11	0.73	0.55
Olea welwitschii	1.60	0.58	2.7	0.44	0.09	0.62	0.55
Brucea antidysenterica	0.51	0.18	3.1	0.50	0.13	0.83	0.50
Teclea nobilis	0.32	0.12	3.1	0.50	0.13	0.83	0.48
Lepidotrichilia volkense	0.20	0.07	0.8	0.13	0.19	1.25	0.48
Sarcocephalus latifolius	0.68	0.25	3.5	0.56	0.09	0.62	0.48
Celts africana	0.39	0.14	3.5	0.56	0.11	0.73	0.48
Apodytes dimidata	1.34	0.49	1.6	0.25	0.09	0.62	0.45
Dalbergia lactea	0.48	0.17	1.6	0.25	0.14	0.93	0.45
Filicium decipiens	1.00	0.37	2.0	0.31	0.09	0.62	0.43
Alangium chinense	0.97	0.35	1.6	0.25	0.09	0.62	0.41
Olea capensis subsp macrocarpa	0.03	0.01	0.4	0.06	0.17	1.14	0.41
Flacourtia indica	0.27	0.10	1.6	0.25	0.13	0.83	0.39
Anthocleista schweinfurthii	1.52	0.55	1.2	0.19	0.06	0.42	0.39
Ficus thonningii	1.44	0.52	1.2	0.19	0.06	0.42	0.38
Pouteria altisma	0.71	0.26	2.0	0.31	0.08	0.52	0.36
Albizia schimperiana	0.75	0.27	1.2	0.19	0.09	0.62	0.36
Bridelia micrantha	0.29	0.11	1.2	0.19	0.11	0.73	0.34
Eugenia bukobensis	0.39	0.14	1.6	0.25	0.09	0.62	0.34
Ficus exasperata	0.96	0.35	0.8	0.13	0.08	0.52	0.33
Hallea rubrostipulata	1.16	0.42	1.6	0.25	0.05	0.31	0.33
Breonadia salicina	0.50	0.18	1.2	0.19	0.08	0.52	0.30
Antiaris toxicara	0.35	0.13	0.8	0.13	0.09	0.62	0.29
Morus mesozygia	1.00	0.37	0.4	0.06	0.06	0.42	0.28
Dracaena afromontana	0.75	0.27	0.8	0.13	0.05	0.31	0.24
Ekbergia capensis	0.56	0.20	0.4	0.06	0.06	0.42	0.23
Dracaena steudneri	0.09	0.03	0.8	0.13	0.08	0.52	0.22
Pittosporum viridiflorum	0.05	0.02	0.8	0.13	0.08	0.52	0.22
Fagaropsis angolensis	0.52	0.19	0.8	0.13	0.05	0.31	0.21
Celts toka	0.11	0.04	0.4	0.06	0.06	0.42	0.17
Ocotea keynensis	0.22	0.08	0.8	0.13	0.05	0.31	0.17
Nuxia congesta	0.33	0.12	0.4	0.06	0.05	0.31	0.16
Maytenus addat	0.43	0.16	0.8	0.13	0.03	0.21	0.16
Scherebra alata	0.03	0.01	0.4	0.06	0.06	0.42	0.16

Cyathea manniana	0.07	0.02	0.8	0.13	0.05	0.31	0.15
Senna petersiana	0.01	0.00	0.8	0.13	0.05	0.31	0.15
Manilkara butugi	0.37	0.14	1.2	0.19	0.02	0.10	0.14
Deinbollia kilimandischarica	0.14	0.05	0.4	0.06	0.05	0.31	0.14
Vernonia amygdalina	0.07	0.03	0.4	0.06	0.05	0.31	0.13
Grewia ferruginea	0.03	0.01	0.4	0.06	0.05	0.31	0.13
Dombeya torrida	0.30	0.11	0.4	0.06	0.03	0.21	0.13
Trema orientalis	0.40	0.15	0.4	0.06	0.02	0.10	0.10
Ficus palmata	0.06	0.02	0.4	0.06	0.03	0.21	0.10
Rhus quartiniana	0.04	0.02	0.4	0.06	0.03	0.21	0.10
Ilex mitis	0.06	0.02	0.8	0.13	0.02	0.10	0.08

Appendix 4.2 Importance value of tree species in Site II. BA = Basal area, RDO = Relative dominance, D = Species Density, RED = Relative Density, F = Frequency, RF = Relative frequency and IVI = Importance Value Index

Species	BA	RDO	D	RED	F	RF	IVI
Syzygium guineense	78.13	19.58	93.23	13.60	0.85	7.15	13.44
Croton macrostachyus	19.76	4.95	28.65	4.18	0.69	5.76	4.96
Millettia ferruginea	19.72	4.94	41.41	6.04	0.45	3.75	4.91
Albizia gummifera	31.01	7.77	23.96	3.49	0.41	3.40	4.89
Macaranga capensis	21.32	5.34	31.25	4.56	0.55	4.62	4.84
Chionanthus mildbredii	5.66	1.42	46.88	6.84	0.56	4.71	4.32
Pouteria adolfi-friederici	22.22	5.57	18.75	2.73	0.35	2.97	3.76
Trilipsium madagascariense	35.80	8.97	9.90	1.44	0.05	0.44	3.62
Vepris dainellii	4.94	1.24	34.90	5.09	0.45	3.75	3.36
Oxyanthus speciosus	4.12	1.03	31.25	4.56	0.46	3.84	3.14
Bersama abyssinica	3.71	0.93	24.22	3.53	0.55	4.62	3.03
Prunus africana	17.01	4.26	9.11	1.33	0.22	1.83	2.47
Elaeodendron buchanani	14.36	3.60	15.10	2.20	0.19	1.57	2.46
Ficus sur	15.58	3.90	4.69	0.68	0.26	2.18	2.26
Maesa lanceolata	5.15	1.29	20.05	2.92	0.28	2.36	2.19
Allophlus abyssinicus	6.56	1.64	10.94	1.60	0.40	3.32	2.19
Olea capensis	9.39	2.35	11.98	1.75	0.22	1.83	1.98
Canthium oligocarpum	6.53	1.64	10.16	1.48	0.29	2.44	1.85
Psychotria orophila	1.49	0.37	28.39	4.14	0.05	0.44	1.65
Coffea arabica	1.07	0.27	17.45	2.54	0.20	1.66	1.49
Ehertia cymosa	7.88	1.97	10.94	1.60	0.08	0.70	1.42
Teclea nobilis	0.80	0.20	11.20	1.63	0.28	2.36	1.40

Maytenus undata	1.25	0.31	9.64	1.41	0.26	2.18	1.30
Dracaena fragrans	2.01	0.50	11.20	1.63	0.21	1.74	1.29
Apodytes dimidata	4.84	1.21	4.17	0.61	0.24	2.01	1.28
Albizia schimperiana	4.22	1.06	5.47	0.80	0.23	1.92	1.26
Nuxia congesta	4.11	1.03	7.81	1.14	0.17	1.40	1.19
Cassipourea malosana	0.99	0.25	5.99	0.87	0.28	2.36	1.16
Brucea antidysenterica	0.64	0.16	6.25	0.91	0.25	2.09	1.06
Lepidotrichilia volkensii	1.57	0.39	6.77	0.99	0.21	1.74	1.04
Polycias fulva	4.82	1.21	4.43	0.65	0.14	1.13	1.00
Trichilia dregeana	1.67	0.42	6.51	0.95	0.13	1.05	0.80
Dracaena afromontana	1.66	0.42	3.13	0.46	0.17	1.40	0.76
Ritchiea albersii	0.96	0.24	3.91	0.57	0.17	1.40	0.74
Sapium ellipticum	5.63	1.41	1.56	0.23	0.06	0.52	0.72
Maytenus arbutifolia	0.54	0.14	5.73	0.84	0.13	1.05	0.67
Ekbergia capensis	2.96	0.74	1.82	0.27	0.11	0.96	0.66
Psydrax parviflora	0.66	0.17	5.73	0.84	0.11	0.96	0.65
Flacortia indica	0.66	0.17	3.39	0.49	0.15	1.22	0.63
Rothmannia urcelliformis	1.87	0.47	3.91	0.57	0.09	0.79	0.61
Galineria saxifraga	0.75	0.19	5.21	0.76	0.10	0.87	0.61
Diospyros abyssinica	1.90	0.48	3.91	0.57	0.06	0.52	0.52
Vangueria apiculata	0.22	0.06	7.81	1.14	0.04	0.35	0.51
Afrocarpus falcatus	0.78	0.20	1.82	0.27	0.13	1.05	0.50
Albizia grandibracteata	2.24	0.56	3.65	0.53	0.03	0.26	0.45
Celts africana	1.76	0.44	2.34	0.34	0.06	0.52	0.44
Garcinia buchanani	2.96	0.74	3.13	0.46	0.01	0.09	0.43
Dalbergia factea	0.28	0.07	2.08	0.30	0.09	0.79	0.39
Euphorbia ampliphylla	2.01	0.50	1.82	0.27	0.04	0.35	0.37
Ilex mitis	0.56	0.14	3.65	0.53	0.05	0.44	0.37
Dombeya torrida	1.51	0.38	2.08	0.30	0.04	0.35	0.34
Scheflera abyssinica	1.01	0.25	1.04	0.15	0.07	0.61	0.34
Pittosporum viridiflorum	0.56	0.14	2.34	0.34	0.05	0.44	0.31
Trema orientalies	1.47	0.37	2.34	0.34	0.02	0.17	0.30
Cordia africana	0.84	0.21	1.82	0.27	0.03	0.26	0.25
Eugenia bukobensis	0.83	0.21	2.60	0.38	0.01	0.09	0.22
Bridelia micrantha	1.33	0.33	1.04	0.15	0.02	0.17	0.22
Dracaena steudneri	1.38	0.35	1.30	0.19	0.01	0.09	0.21
Ficus thonningii	0.90	0.23	0.78	0.11	0.03	0.26	0.20
Blighia unijugata	1.05	0.26	1.04	0.15	0.01	0.09	0.17
Olea welwitschii	0.80	0.20	0.52	0.08	0.02	0.17	0.15
Allophylus macrobotrys	0.06	0.01	0.78	0.11	0.02	0.17	0.10
Mimusops kummel	0.44	0.11	0.26	0.04	0.01	0.09	0.08

Vernonia amygdalina	0.04	0.01	0.26	0.04	0.01	0.09	0.05
Phoeneix reclinata	0.03	0.01	0.26	0.04	0.01	0.09	0.04

Appendix 4.3 Importance value of tree species in Site III. BA = Basal area, RDO = Relative dominance, D = Species Density, RED = Relative Density, F = Frequency, RF = Relative frequency and IVI = Importance Value Index

Species	BA	RDO	D	RED	F	RF	IVI
Syzygium guineense	68.19	16.86	107.50	14.82	0.93	7.91	13.20
Pouteria adolfi-friederici	55.37	13.69	37.19	5.13	0.53	4.49	7.77
Prunus africana	41.72	10.32	18.75	2.59	0.33	2.78	5.23
Croton macrostachyus	22.86	5.65	29.69	4.09	0.66	5.66	5.14
Macaranga capensis	24.39	6.03	36.88	5.08	0.48	4.06	5.06
Bersama abyssinica	7.90	1.95	43.13	5.95	0.61	5.24	4.38
Milletia ferruginea	16.54	4.09	36.56	5.04	0.35	2.99	4.04
Albizia gummifera	16.78	4.15	26.88	3.71	0.40	3.42	3.76
Vepris dainellii	2.32	0.57	37.81	5.21	0.45	3.85	3.21
Alllophylus abyssinicus	8.03	1.99	19.38	2.67	0.58	4.91	3.19
Apodytes dimidata	12.46	3.08	15.31	2.11	0.45	3.85	3.01
Oxyanthus speciosus	5.46	1.35	30.63	4.22	0.36	3.10	2.89
Canthium oligocarpum	8.08	2.00	21.56	2.97	0.43	3.63	2.87
Afrocarpus falcatus	20.52	5.08	8.13	1.12	0.21	1.82	2.67
Coffea arabica	0.97	0.24	31.56	4.35	0.25	2.14	2.24
Chionanthus mildbraedii	2.46	0.61	22.19	3.06	0.28	2.35	2.01
Galineria saxifraga	2.53	0.63	16.56	2.28	0.34	2.88	1.93
Ficus sur	11.58	2.86	4.38	0.60	0.15	1.28	1.58
Dracaena fragrans	1.92	0.48	11.88	1.64	0.25	2.14	1.42
Brucea antidysenterica	1.16	0.29	15.00	2.07	0.21	1.82	1.39
Olea capensis subsp macrocarpa	3.52	0.87	6.88	0.95	0.26	2.24	1.35
Cassipourea malosana	1.15	0.28	8.13	1.12	0.28	2.35	1.25
Scheflera abyssinica	12.15	3.00	1.25	0.17	0.05	0.43	1.20
Maesa lanceolata	2.34	0.58	8.75	1.21	0.20	1.71	1.16
Elaeodendron buchanani	3.90	0.96	7.50	1.03	0.18	1.50	1.16
Cordia africana	8.43	2.08	4.69	0.65	0.09	0.75	1.16
Dracaena afromontanum	2.05	0.51	7.50	1.03	0.20	1.71	1.08
Psydrax parviflora	0.91	0.22	8.44	1.16	0.19	1.60	1.00
Olea welwitschii	4.84	1.20	6.88	0.95	0.06	0.53	0.89
Polycias fulva	4.92	1.22	3.44	0.47	0.11	0.96	0.88
Pavetta oliveriana	0.86	0.21	7.81	1.08	0.15	1.28	0.86
Lepidotrichilia volkensii	1.03	0.25	7.19	0.99	0.15	1.28	0.84
Trichilia dregeana	0.66	0.16	7.19	0.99	0.15	1.28	0.81

Maytenus undata	0.52	0.13	8.13	1.12	0.14	1.18	0.81
Teclea noblis	0.47	0.12	7.81	1.08	0.08	0.64	0.61
Psychotria orophila	0.35	0.09	4.38	0.60	0.13	1.07	0.59
Vernonia amygdalina	1.19	0.29	7.50	1.03	0.05	0.43	0.58
Celts africana	1.30	0.32	2.81	0.39	0.11	0.96	0.56
Ehertia cymosa	1.01	0.25	4.06	0.56	0.09	0.75	0.52
Dombeya torrida	3.76	0.93	3.75	0.52	0.01	0.11	0.52
Diospyros abyssinica	2.12	0.52	3.44	0.47	0.06	0.53	0.51
Flacortia indica	0.71	0.18	2.50	0.34	0.08	0.64	0.39
Pittosporum viridiflorum	0.45	0.11	2.81	0.39	0.08	0.64	0.38
Maytenus arbutifolia	0.18	0.04	0.94	0.13	0.10	0.85	0.34
Ritchiea albersii	0.43	0.11	2.19	0.30	0.05	0.43	0.28
Ekbergia capensis	1.26	0.31	1.25	0.17	0.04	0.32	0.27
Euphorbia ampliphylla	1.80	0.45	1.25	0.17	0.01	0.11	0.24
Sapium ellipticun	1.02	0.25	0.94	0.13	0.04	0.32	0.23
Trilepisium madagascariense	0.97	0.24	0.94	0.13	0.04	0.32	0.23
Nuxia congesta	0.46	0.11	1.56	0.22	0.04	0.32	0.22
Ilex mitis	0.58	0.14	2.81	0.39	0.01	0.11	0.21
Cyathea manniana	0.33	0.08	1.88	0.26	0.03	0.21	0.18
Ficus sycomorus	1.22	0.30	0.63	0.09	0.01	0.11	0.17
Maytenus addat	0.27	0.07	1.25	0.17	0.03	0.21	0.15
Ficus thonningii	1.13	0.28	0.31	0.04	0.01	0.11	0.14
Ficus lutea	1.04	0.26	0.31	0.04	0.01	0.11	0.14
Celts zenkeri	0.85	0.21	0.31	0.04	0.01	0.11	0.12
Hallea rubrostipulata	0.23	0.06	0.63	0.09	0.03	0.21	0.12
Garcinia buchananii	0.75	0.19	0.31	0.04	0.01	0.11	0.11
Dracaena steudmeri	0.26	0.06	0.94	0.13	0.01	0.11	0.10
Albizia schimperiana	0.48	0.12	0.31	0.04	0.01	0.11	0.09
Antiaris toxicaria	0.45	0.11	0.31	0.04	0.01	0.11	0.09
Eugenia bukobensis	0.23	0.06	0.31	0.04	0.01	0.11	0.07
Alangium chin	0.15	0.04	0.31	0.04	0.01	0.11	0.06
Bridelia micrantha	0.12	0.03	0.31	0.04	0.01	0.11	0.06
Dovyalis abyssinica	0.11	0.03	0.31	0.04	0.01	0.11	0.06
Scheribera alata	0.08	0.02	0.31	0.04	0.01	0.11	0.06
Rothmannia urcelliformis	0.06	0.01	0.31	0.04	0.01	0.11	0.05
Grewia ferrugina	0.06	0.01	0.31	0.04	0.01	0.11	0.05
Vanguria apiculata	0.04	0.01	0.31	0.04	0.01	0.11	0.05

Appendix 4.4 Regenerating species seedling and sapling density in each site (I, II and III)

Species (Site I)	density	Species (Site II)	density	Species (Site III)	density
Coffea arabica	1483	*Oxyanthus speciosus*	561	*Chionanthus mildbraedii*	487
Albizia gummifera	778	*Pouteria adolfi-friederici*	531	*Bersama abyssinica*	336
Millettia ferruginea	719	*Chionanthus mildbraedii*	510	*Coffea arabica*	324
Maesa lanceolata	388	*Vepris dainellii*	453	*Syzygium guineense*	312
Syzygium guineense	339	*Prunus africana*	310	*Veperis dainelii*	311
Bersama abyssinica	337	*Syzygium guineense*	310	*Albizia gummifera*	307
Psychotria orphila	334	*Croton macrostachyus*	219	*Pouteria adolfi-friederici*	288
Croton macrostachyus	326	*Maesa lanceolata*	166	*Dracaena fragrans*	272
Dracaena steudneri	310	*Albizia gummifera*	163	*Olea capensis subs macrocarpa*	171
Veperis dainellii	303	*Bersama abyssinica*	150	*Oxanthus speciosus*	156
Trilepisium madagascariense	296	*Coffea arabica*	148	*Syzygium guineense*	153
Ritchiea albersii	275	*Olea capensis*	130	*Polyscia fulva*	115
Pouteria adolfi-friederici	270	*Macaranga capensis*	104	*Croton macrostachyus*	100
Cordia africana	262	*Apodytes dimidata*	87	*Macaranga capensis*	95
Chionanthus mildbraedii	256	*Canthium oligocarpum*	75	*Maesa lanceolata*	79
Oxyanthus speciosus	212	*Milletia ferruginea*	70	*Maytenus undata*	75
Macaranga capensis	166	*Allophylus abyssinicus*	68	*Prunus africana*	73
Argomuellera macrophylla	160	*Pavetta abyssinica*	60	*Apodytes dimidata*	72
Cassiopurea malosana	159	*Lepidotrichilia*	54	*Allophylus abyssinicus*	69
Ficus palmata	154	*Celts africana*	44	*Trichilia dregeana*	65
Trichilia dregeana	145	*Galineria saxfraga*	35	*Canthium oligocarpum*	62
Maytenus undata	143	*Brucei antidysentrica*	30	*Millettia ferruginea*	62
Allophyllus abyssinicus	139	*Cassipurea malosana*	27	*Brucei antidysenterica*	55
Sapium ellipticum	130	*Ilex mitis*	23	*Elaerodendron buchanani*	52
Dracaena fragrans	127	*Albizia schemperiana*	19	*Cassipourea malosana*	49
Rothmannia urcelliformis	126	*Dalbergia lactea*	19	*Galineria saxfraga*	46
Flacortia indica	101	*Maytenus undata*	18	*Lepidotrichilia volkensii*	46
Teclea nobilis	99	*Dracaena stuedeneri*	15	*Afrocarpus falcatus*	39
Canthium oligocarpum	89	*Diospyros abyssinicus*	14	*Teclea noblis*	38
Dalbergia factea	82	*Ficus sur*	12	*Nuxia congesta*	35
Brucea antidysenterica	73	*Polyscias fulva*	11	*Flacourtia indica*	32
Diospyros abyssinica	72	*Rothmannia urcelliformis*	9	*Cassipurea malosana*	32
Maytenus arbutifolia	70	*Euphorbia sp*	8	*Albizia schimperiana*	30
Nuxia congesta	55	*Ritchiea albersii*	8	*Pittosporum vridifolium*	30
Ilex mitis	48	*Trilipsium madagascariense*	8	*Chanthium oligocarpa*	25
Lepidotrichilia volkensii	48	*Ehertia cymosa*	6	*Ehertia cymosa*	20

Albizia schimperiana	47	*Sapium ellipticum*	6	*Ilex mitis*	17
Prunus africana	46	*Dracaena afromontana*	5	*Euphorbia sp*	9
Antiaris toxicaria	44	*Ekbergia capensis*	5	*Psydrax parviflora*	8
Ficus lutea	43	*Ealerodenderon buchnanii*	5	*Rothmannia urcelliformis*	8
Ehertia cymosa	41	*Mimusops kummel*	5	*Scheflera abyssinica*	7
Polyscias fluva	40	*Nuxia congesta*	5	*Celts africana*	5
Pavetta abyssinica	35	*Psydrax parviflora*	5	*Ritchiea albersii*	5
Dracaena afromontana	28	*Teclea noblis*	5	*Sapium ellepticum*	5
Celts africana	26	*Cordia africana*	3	*Ficus sur*	4
Scheflera abyssinica	26	*Schflera abyssinica*	2	*Dracaena stuednerii*	3
Albizia grandibracteata	24	*Flacourtia indica*	1		
Aningeria altissima	24	*Phoniex reclinata*	1		
Apodytes dimidata	22	*Afrocarpus falcatus*	1		
Bridelia micrantha	22				
Celts toka	17				
Vangueria apiculata	16				
Pittosporum viridiflorum	15				
Galineria saxifraga	13				
Olea capensis subsp macrocarpa	12				
Anthocleista schweinfurthii	11				
Ocotea keynensis	10				
Manilkara butugi	9				
Olea welwitschii	9				
Vernonia amygdalina	8				
Pappea capensis	7				
Chanthium lactescens	6				
Ficus vallis-choudae	6				
Grewia ferruginea	6				
Psydrax parviflora	6				
Dombeya torrida	5				
Garcinia buchanani	5				
Phoniex reclinata	5				
Elaeodendron buchananii	3				
Ficus sur	3				
Ficus vasta	3				
Morus mesozygia	3				
Mumisops kummel	3				
Euphorbia sp	2				
Trilipsium madagascariense	2				

Eugenia bukobensis	1
Ficus mucuso	1
Ficus ovata	1
Hallea rubrostipulata	1
Rhus quartinaiana	1
Sarcocephlus latifolius	1
Trema orientalis	1

Appendix 4.5 The total species and mean alpha diversity of species at each sampled forest fragment

Site I		
Site Name	Total species	Mean Fisher's alpha
Metu_zuria	95	68.68±13.1
Gore_zuria	82	40.85±4.9
Burusa	75	35.96±3.2
Baljo	70	33.51±2.6
Uka	69	31.50±2.2
Masha	61	29.82±1.9
Leka	62	28.51±1.7
Gaba	52	27.48±1.6
Site II		
Sibu1	65	41.38±8.1
Sibu2	62	24.19±3.2
Markafa	62	21.76±2.2
Tulu_Boka	59	20.87±1.8
Yaroo	56	20.13±1.6
Lag_chancho	55	19.72±1.4
Gobe	50	19.51±1.3
Malate	48	19.33±1.3
Tulu_Mako	44	19.24±1.2
Jorgo	44	19.12±1.2
Qolo_Warabesa	39	19.03±1.1
Bongi	27	18.83±1.1
Site II		
Issya	72	37.67±7.3
Qotora1	44	22.29±2.9
Qotora2	46	20.71±2.1
Koda	54	20.02±1.9
Simber	55	19.67±1.5
Dike	56	19.33±1.4

Busa_jireen	51	19.25±1.3
Buyina	58	19.19±1.2
Jamia	65	19.22±1.2
Jireen	47	19.15±1.4

Appendix 4.6 Plant species list of 30 sampled forest patches

Acanthaceae

Acanthus eminens C.B.Clarke
Acanthopale pubescens (Lindau) C.B.Cl.
Justicia betonica L
Justcia schimperiana
Thunbergia alata Boj. ex. Sims

Alangiaceae

Alangium chinense (Lour.) Harms

Amaranthaceae

Cyathula uncinulata
Sericostachys scandens Gilg & Lopr

Anacardiaceae

Rhus ruspolii Engl.
Rhus glutinosa A. Rich
Rhus quartiniana A.Rich

Apocynaceae

Landolphia buchananii (Hall. F.) Stapf
Oncinotis tenuiloba Stapf

Aquifoliaceae

Ilex mitis (L.) Radlk.

Araliaceae

Polyscias fulva (Heirn) Harms
Schefflera abyssinica (Hochst ex A. Rich) Harms
Schefflera volkensii (Engl.) Harms
Schefflera myriantha (Bak.) Drake

Arecaceae

Phoenix reclinata Jacq.

Asclepiadaceae

Blyttia fruticulosum
Ceropegia cufodontis Chiov
Dregea schimperi (Decne.) Bullock
Pentarrhinum inspidum E. Mey
Pergularia daemia (Forssk.) Chiov subsp. daemia
Periploca linearifolia Quert.-Dill. & A. Rich.

Lauraceae

Cassytha filiformis L.
Ocotea keynensis

Liliaceae

Simlax aspera L
Sparmannia ricinocarpa (Eck. & Zeyh.)

Loganiaceae

Anthocleista schweinfurthii Gilg.
Strychnos mitis S. Moore

Malvaceae

Abutilon cecilli N.E.Br
Abutilon longicuspe Hochst. Ex A. Rich.
Hibiscus ludwigii Eckl. & Zeyh
Hibiscus micranthus L.f.
Hibiscus sp2
Pavonia urens Cav.
Sida ternata L.f
Sida collina Schlechtend

Maranthaceae

Maranthochloa leucantha (A. Rich) Munro

Meliaceae

Eckebergia capensis Saprrm.
Lepidotrichilia volkensii (Gürke) Leroy
Trichilia dregeana Sond
Turraea holstii Guerke

Melianthaceae

Bersama abyssinica Fresen.

Menispermaceae

Cissampelos pareira L-Abuta
Cissampleos torulosa E. Mey. ex Harv
Stephania abyssinica (Dillion ex A. Rich.) Walp.
Tiliachora funifera Oliv.
Tiliachora troupinii Cufod

Moraceae

Secamone parvifolia (Oliv.) Bullock
Secamone punctulata Decne
Asparagaceae
Asparagus officinalis
Asparagus africana
Asteraceae
Aspilia mossambicensis
Helichrysum splendidum (Thunb.) Less
Laggera pterodonta (DC.) Sch, Bip. Ex Oli
Microglossa pyrifolia (Lam.) Kuntze
Solanecio gigas(Vatke) C.Jeffrey
Solanecio mannii (Hook.f.) C.Jeffrey
Vernonia auriculifera Hiern.
Vernonia glabra (Steetz) Vatke
Vernonia myriantha Hook. f.
Vernonia urticifolia A. Rich.
Vernonia ruepellii Sch. Bip. Ex Walp.
Vernonia amygdalina Del
Buddlejiaceae
Buddleja polystachya Fresen
Nuxia congesta R.Br. ex Fresen.
Nuxia oppositifolia (Hochst.) Benth-CJB
Boraginaceae
Cordia africana Lam.
Ehertia cymosa Thonn.
Campanulaceae
Lobelia gibberoa Hemsl.
Capparidaceae
Ritchiea albersii Gilg
Celasteraceae
Carissa span L
Hippocratea africana (Wild) Loes
Hippocratea pallens Plancho ex Oliver
Hippocratea goetzei Loes
Elaeodendron buchananii (Loes) Loes
Maytenus gracilipes (Welw. ex Oliv.) Exell
Maytenus obscura (A Rich) Cuf
Maytenus adat (Loes.) Sebsebe
Maytenus arbutifolia (A. Rich.) Wilczek
Maytenus undata (Thunb.) Blakelock
Combretaceae

Antiaris toxicara
Ficus capreaefolia Del.
Ficus exasperata Vahl
Ficus lutea Vahl
Ficus mucuso Ficalho
Ficus ovata
Ficus palmata
Ficus sur Forssk.
Ficus sycomorus L.
Ficus thonningii Blume
Ficus vallis-choudae Del.
Ficus vasta Forssk.
Morus mesozygia Stapf
Trilepisium madagascariense DC
Myrsinaceae
Embelia schimperi Vatke
Myrsine africana L.
Maesa lanceolata Forssk.
Myrtaceae
Eugenia bukobensis Engl.
Syzygium guineense (Willd.) DC. ssp. afromontanum
Oleaceae
Jasminum abyssinicum Hochst. ex DC.
Chionanthus mildbraedii (Gilg &Scellenb.) Stearn.
Olea capensis subsp macrocarpa (C.H.Wright) I. Verd
Olea welwitschii (Knobl.) Gilg & Schellenb
Schrebera alata (Hochst.) Welw.
Opiliaceae
Opilia campestris Engl.
Phytolaccaceae
Phytolacca dodecandra L´Herit.
Piperaceae
Piper capense L.f. var. capense
Pittosporaceae
Pittosporum viridiflorum Sim.
Podocarpaceae
Podocarpus falcatus (Thunb.) Mirb
Ranunculaceae
Clematis longicauda Steud. Ex A.Rich.
Clematis simensis Fresen.
Clematis hirsuta Perr. & Guill

Combretum paniculatum Vent.
Convolvulaceae
Ipomoea cairica (L.) Sweet
Ipomoea hochstetteri House
Ipomoea obscura (L.) Ker-Gawl.
Ipomoea tenuirostris Steud. Ex Choisy
Cucurbitaceae
Kedrostis foetidissima (Jacq.) Cogn.
Lagenaria abyssinica (Hook.f.) C.Jeffrey
Momordica foetida Schumach
Peponium vogelii (Hook.f.) Engle.
Sicyos polyacanthus Cong.
Zeheneria scabra (Linn. f.) Sond.
Cyatheaceae
Cyathea manniana
Dioscoreaceae
Dioscorea bulbifera L.
Dracaenaceae
Dracaena afromontana Mildbr.
Dracaena fragrans (L) Ker-Gawl
Dracaena steudneri Engl
Ebenaceae
Diospyros abyssinica (Hiern) F. White
Euphorbiaceae
Tragia brevipes Pax
Acalypha marissima
Acalypha ornata
Acalypha psilostachya
Acalypha racemosa
Acalypha acrogyna
Argomuellera macrophylla Pax
Bridelia micrantha (Hochs) Baill
Clutia abyssinica Jaub. & Spach
Croton macrostachyus Del.
Erythrococcaa trichogyne (Muell Arg) Prain (det A Radcliffe-Smith)
Euphorbia ampliphylla Pax
Euphorbia acandilabrium
Macaranga capensis var kilimandscharica (Pax) Friis & Gilbert
Phyllanthus limmuensis Cuf
Rhamnaceae
Gouania longispicata Engl
Helinus mystacinus (Ait.) E. Mey. ex Steud.
Rhamnus prinoides L`Herit.
Rhamnus staddo A. Rich.
Scutia myrtina (Burm. f.) Kurz
Rhizophoraceae
Cassipourea malosana (Bak.) Alston
Rosaceae
Rubus steudneri Schweinf.
Rubus apetalus Poir.
Prunus africana (Hook.f.) Kalkm.
Rubiaceae
Breonadia salicina (Vahl) Hepper &Wood
Canthium oligocarpum Hiern
Canthium sp
Coffea arabica L.
Crossopteryx febrifuga (Afz. Ex G.Don) Benth.
Galiniera saxifraga (Hochst.) Bridson
Hallea rubrostipulata (K.Schum.) J.-F.Leroy
Keetia gueinzii (Sond.) Bridson
Pavetta abyssinica Fresen
Pavetta oliveriana Hiern
Pentas lanceolata (Forssk.)
Psydrax schimperiana (A. Rich.) Bridson
Rytigynia neglecta (Hiern) Robyns
Oxyanthus lepidus
Oxanthus speciosus sbsp. Stenocarpus (K.Schum.) Bridson
Psychotria orophila Petit
Psydrax parviflora (Afz.) Bridson
Rothmannia urcelliformis (Hiern) Robyns
Sarcocephalus latifolius (Sm.) Bruce.
Vangueria apiculata K. Schum
Rutaceae
Clausena anisata (Willd.) Benth.
Fagaropsis angolensis (Engl.) Dale
Teclea nobilis Del.
Vepris dainelli (Pichi-Sermolli) Kokwaro
Sapindaceae
Allophyllus abyssinicus (Hochst.) Radlkofer

Phyllanthus ovalifolius Forssk
Ricinus communis L.
Sapium ellipticum (Hochst) Pax.

Fabaceae

Albizia grandibracteata. T
Albizia gummifera (J.F.Gmel.) C.A.Sm.
Albizia schimperiana Oliv.
Calpurnia aurea (Ait.) Benth.
Crotalaria emarginella
Dolichos sericeus E. Mey
Dalbergia lactea Vatke
Glycine wightii (Wight & Arn.) Verdc.
Indigofera arrecta Hochst ex A. Rich.
Kotschya sp
Macrotyloma axillare (E. Mey.) Verdc
Milletia ferruginea (Hochst) Baker
Pterolobium stellatum (Forssk.) Brenan
Rhynchosia minima (L.) DC.
Senna petersiana (Bolle) Lock
Tephrosia linearis (Willd.) Pers
Tephrosia villosa (L.) Pers.
Vigna parkeri Bak.

Flacourtaceae

Dovyalis abyssinica (A. Rich) Warb.
Flacourtia indica (Burm.f.) Merr.

Guttiferae

Garcinia buchananii Baker

Hypericaceae

Hypericum quartinianum A. Rich.

Icacinaceae

Apodytes dimidata, Arn.

Lamiaceae

Clerodendron myricoides (Hochst.) R. Br.ex Vatke.
Leonotis ocymifolia (Burm.f.) Iwarsson
Leucas glabrata (Vahl) R.Br.
Leucas martinicensis (Jacq.) R. Br.
Ocimum lamiifolium (Hochst. ex Bent.) DC.
Ocimum suave Willd.
Plectranthus assurgens (Baker) J.K. Morton
Plectranthus sylvestris Gürke
Premna schimperi Engl.
Pycnostachys abyssinica Fresen.

Allophylus macrobotrys (Gilg
Blighia unijugata
Dododaea angustifolia
Deinbollia kilimandscharica Taub.
Filicium decipiens (Wight &Am) Thwaites
Lepisanthes senegalensis (Juss. ex Poir.) Leenh.
Pappea capensis Eckl. & Zeyh.
Paullinia pinnata (L)

Sapotaceae

Manilkara butugi Chiov.
Mimusops kummel Bruce ex A.DC.
Pouteria adolfi-friederici
Pouteria altissima (A.Chev.) Aubrev. & Pellegr

Simaroubaceae

Brucea antidysenterica J.F. Mill.

Solanaceae

Discopodium penninervium Hochst.
Solanum incanum L.
Solaniun indicum L
Solanium anguivi Lam

Stericuliaceae

Dombya torrida (J.F. Gmel.) P. Bamps

Tiliaceae

Grewia ferruginea Hochst. ex A. Rich.

Ulmaceae

Celtis africana Burm. F.
Celts gompholla Bak
Celtis toka (Forssk.) Hepper & wood
Celts zenkeri Endl
Trema orientalis (L) Blume

Urticaceae

Elatostema monticolum Hook. F.
Urerea hypselodendron (A. Rich.) Wedd.
Urerea trinervis (Hochst. Ex Krauss) Friis & immelman

Verbenaceae

Lantana trifolia L
Lippia adoensis Hochst. ex Walp.

Vitaceae

Cissus quadrangularis L.
Cissus petiolata Hook. f.
Cissus quadrangularis Linn
Cyphostemma adenocaule (Stued. ex A. Rich.) Desc.oings ex Wild & R.B.

Pycnostachys eminii Gürke	*Cyphostemma cyphoetalum (Fresen.) Desc. ex Wild & R.B.* *Cyphostemma rivae (Gilg) Desc.oings* *Rhoicissus tridentata (L. f.) Wild & R.B. Drumm.Willd & Drummond*

Appendix 5.1 The importance value of tree species in the edge habitat. BA = Basal area, RDO = Relative dominance, D = Species Density, RED = Relative Density, F = Frequency, RF = Relative frequency and IVI = Importance Value Index

			Edge				
Species	BA	RDO	D	RDE	F	RF	IVI
Syzygium guineense	53.0	11.7	4.5	10.5	0.67	6.3	9.5
Croton macrostachyus	37.9	8.4	4.0	9.3	0.74	6.9	8.2
Macaranga capensis	34.9	7.7	3.4	7.9	0.62	5.7	7.1
Maesa lanceolata	31.4	6.9	5.0	11.7	0.65	6.0	8.2
Prunus africana	21.6	4.8	0.8	1.8	0.27	2.5	3.0
Ficus sur	20.7	4.6	0.5	1.2	0.31	2.9	2.9
Pouteria adolfi-friederici	20.2	4.5	1.0	2.3	0.22	2.1	2.9
Nuxia congesta	19.2	4.2	1.6	3.6	0.35	3.2	3.7
Albizia gummifera	19.1	4.2	1.1	2.6	0.33	3.1	3.3
Milletia ferrugina	19.0	4.2	1.9	4.4	0.40	3.7	4.1
Sapium elpticum	18.3	4.0	0.3	0.8	0.13	1.2	2.0
Albizia schimperiana	15.6	3.4	1.0	2.4	0.46	4.3	3.4
Maytenus arbutifolia	10.1	2.2	0.9	2.2	0.35	3.2	2.5
Allophlus abyssinicus	9.6	2.1	1.1	2.5	0.34	3.2	2.6
Bersama abyssinica	9.5	2.1	2.5	5.9	0.58	5.4	4.5
Apodytes dimidata	8.9	2.0	0.5	1.1	0.24	2.3	1.8
Acacia abyssinica	8.2	1.8	0.5	1.1	0.19	1.7	1.6
Ekbergia capensis	7.9	1.7	0.1	0.1	0.02	0.2	0.7
Afrocarpus falcatus	7.0	1.5	0.2	0.5	0.11	1.0	1.0
Celtis africana	6.0	1.3	0.2	0.5	0.10	0.9	0.9
Rhus glutinosa	5.8	1.3	1.6	3.6	0.35	3.2	2.7
Elaeodendron buchanani	5.2	1.1	0.2	0.4	0.06	0.5	0.7
Polycias fulva	3.6	0.8	0.1	0.2	0.05	0.5	0.5
Canthium oligocarpum	3.6	0.8	0.5	1.1	0.15	1.4	1.1
Schefleria abyssinica	3.6	0.8	0.1	0.3	0.04	0.4	0.5
Flacortia idica	3.3	0.7	0.3	0.7	0.11	1.0	0.8
Vernonia amygdalina	3.0	0.7	0.7	1.7	0.34	3.2	1.8
Euphorbia candilabrium	2.5	0.6	0.1	0.2	0.04	0.4	0.4
Dombeya torrida	2.4	0.5	0.1	0.2	0.07	0.6	0.5
Ficus thonningii	2.3	0.5	0.1	0.2	0.03	0.3	0.4
Olea capensis subsps	2.3	0.5	0.3	0.7	0.10	1.0	0.7

macrocarpa							
Cupressus lucitanica	2.2	0.5	0.4	1.0	0.06	0.6	0.7
Trema orientalis	2.1	0.5	0.2	0.5	0.05	0.5	0.5
Cassipourea malosana	2.0	0.5	0.1	0.3	0.06	0.5	0.4
Ehertia cymosa	1.9	0.4	0.3	0.6	0.10	0.9	0.7
Galineria saxifraga	1.8	0.4	0.4	1.0	0.11	1.0	0.8
Cordia africana	1.8	0.4	0.1	0.3	0.04	0.4	0.4
Dracaena afromontana	1.7	0.4	0.2	0.5	0.01	0.1	0.3
Vepris dainellii	1.4	0.3	0.6	1.4	0.07	0.6	0.8
Maytenus addat	1.3	0.3	0.1	0.3	0.02	0.2	0.3
Maytenus undata	1.3	0.3	0.4	0.8	0.19	1.8	1.0
Albizia grandibracteata	1.2	0.3	0.1	0.2	0.03	0.3	0.3
Brucea antidysenterica	1.1	0.2	0.6	1.4	0.25	2.3	1.3
Dracaena steudneri	1.1	0.2	0.1	0.3	0.03	0.3	0.3
Eucalyptus camaldulensis	1.0	0.2	0.2	0.5	0.02	0.2	0.3
Euphorbia ampliphylla	0.9	0.2	0.1	0.1	0.00	0.0	0.1
Chionanthus mildbraedii	0.9	0.2	0.4	0.8	0.12	1.1	0.7
Lepidotrichlia volkensii	0.8	0.2	0.3	0.6	0.04	0.4	0.4
Pittosporum viridiflorum	0.8	0.2	0.2	0.4	0.06	0.5	0.4
Ficus vasta	0.7	0.2	0.0	0.1	0.02	0.2	0.1
Olea welwitschii	0.7	0.2	0.0	0.1	0.01	0.1	0.1
Gardenia ternfolia	0.7	0.1	0.1	0.2	0.01	0.1	0.2
Rhus ruspolii	0.6	0.1	0.0	0.1	0.06	0.5	0.2
Ritchiea albersii	0.6	0.1	0.1	0.3	0.06	0.5	0.3
Psychotria orophila	0.5	0.1	0.2	0.5	0.06	0.6	0.4
Garcinia buchananii	0.5	0.1	0.0	0.1	0.01	0.1	0.1
Oxyanthus speciosus	0.5	0.1	0.2	0.5	0.06	0.5	0.4
Celts philipensis	0.5	0.1	0.0	0.0	0.01	0.1	0.1
Bridelia micrantha	0.4	0.1	0.1	0.2	0.04	0.4	0.2
Coffea arabica	0.4	0.1	0.3	0.7	0.06	0.6	0.5
Psydrax parviflora	0.4	0.1	0.1	0.3	0.03	0.3	0.2
Stereospermum kunthianum	0.4	0.1	0.0	0.1	0.01	0.1	0.1
Entada abyssinica	0.4	0.1	0.1	0.2	0.03	0.3	0.2
Blighia unijugata	0.4	0.1	0.0	0.1	0.01	0.1	0.1
Fagaropsis angolensis	0.3	0.1	0.0	0.1	0.01	0.1	0.1
Allophylus macrobotrys	0.3	0.1	0.1	0.2	0.02	0.2	0.1
Buddleja plostachya	0.3	0.1	0.1	0.2	0.03	0.3	0.2
Teclea nobilis	0.3	0.1	0.2	0.4	0.03	0.3	0.2
Grewia ferrugina	0.3	0.1	0.2	0.4	0.08	0.7	0.4
Combretum collinum	0.3	0.1	0.0	0.0	0.02	0.2	0.1
Trichilia dregeana	0.2	0.0	0.3	0.8	0.08	0.7	0.5
Maytenus obscura	0.2	0.0	0.1	0.1	0.01	0.1	0.1

Ilex mitis	0.2	0.0	0.0	0.1	0.01	0.1	0.1
Diospyros abyssinica	0.2	0.0	0.1	0.1	0.01	0.1	0.1
Ficus lutea	0.2	0.0	0.1	0.2	0.01	0.1	0.1
Senna petersiana	0.1	0.0	0.1	0.1	0.01	0.1	0.1
Dalbergia lactea	0.1	0.0	0.0	0.1	0.01	0.1	0.1
Celts toka	0.1	0.0	0.0	0.0	0.01	0.1	0.0
Phoneix reclinata	0.1	0.0	0.0	0.1	0.01	0.1	0.0
Dracaena fragrans	0.1	0.0	0.1	0.1	0.11	1.0	0.4
Dodonea angustifolia	0.0	0.0	0.0	0.0	0.01	0.1	0.0
Ficus vallis-choudae	0.0	0.0	0.0	0.0	0.01	0.1	0.0
Dombeya abyssinica	0.0	0.0	0.0	0.0	0.01	0.1	0.0
Canthium giordanii	0.0	0.0	0.0	0.0	0.01	0.1	0.0
Celtis zenkeri	0.0	0.0	0.0	0.0	0.01	0.1	0.0

Appendix 5.2 The importance value of tree species in the interior habitat. BA = Basal area, RDO = Relative dominance, D = Species Density, RED = Relative Density, F = Frequency, RF = Relative frequency and IVI = Importance Value Index

Species	BA	RDO	D	RDE	F	RF	IVI
Syzygium guineense	139.5	21.6	111	16.7	0.9	7.4	15.2
Croton macrostachyus	70.5	10.9	29	4.4	0.5	3.7	6.3
Macaranga capensis	42.9	6.6	36	5.4	0.6	4.5	5.5
Bersama abyssinica	38.1	5.9	12	1.8	0.3	2.1	3.3
Vepris dainellii	34.3	5.3	24	3.6	0.4	2.9	4.0
Chionanthus mildbraedii	32.7	5.1	24	3.7	0.6	5.1	4.6
Allophlus abyssinicus	26.9	4.2	37	5.5	0.4	3.4	4.4
Pouteria adolfi-friederici	21.3	3.3	6	0.9	0.2	1.3	1.8
Oxyanthus speciosus	20.3	3.1	4	0.6	0.5	3.6	2.5
Milletia ferruginea	16.4	2.5	9	1.3	0.4	2.8	2.2
Canthium oligocarpum	16.1	2.5	13	1.9	0.2	1.4	1.9
Albizia gummifera	13.6	2.1	17	2.6	0.4	3.1	2.6
Apodytes dimidata	13.2	2.0	1	0.2	0.1	0.7	1.0
Cassipurea malosana	12.9	2.0	12	1.8	0.3	2.3	2.0
Ficus sur	11.9	1.8	14	2.2	0.5	3.8	2.6
Olea capensis sps macrocarpa	10.8	1.7	1	0.2	0.0	0.3	0.7
Prunus africana	8.3	1.3	33	5.0	0.5	3.6	3.3
Dracaena fragrans	7.4	1.1	39	5.8	0.5	3.8	3.6
Galineria saxifraga	6.7	1.0	24	3.6	0.6	4.4	3.0
Brucea antidysenterica	6.6	1.0	2	0.2	0.1	1.1	0.8
Maytenus undata	6.6	1.0	17	2.6	0.5	4.2	2.6
Coffee arabica	6.5	1.0	2	0.3	0.1	1.1	0.8

Maesa lanceolata	5.6	0.9	4	0.6	0.0	0.2	0.6
Dracaena afromontana	4.6	0.7	6	0.9	0.1	1.1	0.9
Lepidotrichlia volkensii	4.3	0.7	9	1.4	0.2	1.6	1.2
Elaeodendron buchanani	4.2	0.7	2	0.3	0.1	0.8	0.6
Psydrax parviflora	4.0	0.6	12	1.8	0.2	1.9	1.4
Teclea nobilis	3.9	0.6	3	0.4	0.1	0.9	0.7
Psychotria orophila	3.7	0.6	3	0.4	0.0	0.4	0.5
Afrocarpus falcatus	3.2	0.5	3	0.5	0.1	0.5	0.5
Trichilia dregeana	3.1	0.5	2	0.3	0.0	0.4	0.4
Nuxia congesta	2.9	0.4	12	1.7	0.2	1.9	1.4
Polycias fulva	2.7	0.4	2	0.3	0.1	0.7	0.5
Albizia schimperiana	2.6	0.4	5	0.8	0.2	1.5	0.9
Flacortia indica	2.3	0.3	8	1.1	0.3	2.5	1.3
Maytenus arbutifolia	2.1	0.3	7	1.0	0.1	1.1	0.8
Ekbergia capensis	2.0	0.3	1	0.1	0.0	0.2	0.2
Celts africana	2.0	0.3	7	1.1	0.2	1.5	1.0
Schefleria abyssinica	1.7	0.3	11	1.6	0.2	1.8	1.2
Ritchiea albersii	1.7	0.3	2	0.3	0.1	0.4	0.3
Sapium ellipticun	1.7	0.3	11	1.6	0.2	1.3	1.1
Pittosporum viridiflorum	1.6	0.2	10	1.5	0.2	1.8	1.2
Diospyros abyssinica	1.5	0.2	1	0.2	0.0	0.2	0.2
Rothmannia urcelliformis	1.2	0.2	3	0.5	0.1	0.9	0.5
Ilex mitis	1.2	0.2	0	0.1	0.0	0.1	0.1
Dombeya torrida	1.1	0.2	4	0.6	0.1	0.4	0.4
Euphorbia ampliphylla	1.1	0.2	11	1.7	0.2	1.3	1.1
Dalbergia lactea	1.1	0.2	7	1.0	0.2	1.4	0.9
Ehretia cymosa	1.0	0.2	0	0.0	0.0	0.1	0.1
Trilipsium madagascariense	1.0	0.2	14	2.1	0.2	1.7	1.3
Olea welwitschii	1.0	0.1	1	0.1	0.0	0.1	0.1
Euphorbia candilabrium	0.9	0.1	2	0.3	0.0	0.3	0.2
Ficus thonningii	0.9	0.1	3	0.4	0.1	0.6	0.4
Vernonia amygdalina	0.9	0.1	2	0.4	0.1	0.6	0.4
Albizia grandibracteata	0.4	0.1	0	0.0	0.0	0.1	0.1
Cyathea manniana	0.4	0.1	0	0.1	0.0	0.1	0.1
Mimusops kummel	0.3	0.0	0	0.1	0.0	0.1	0.1
Bridelia micrantha	0.3	0.0	2	0.3	0.1	0.8	0.4
Allophylus macrobotrys	0.3	0.0	1	0.2	0.0	0.2	0.1
Cordia africana	0.2	0.0	1	0.1	0.0	0.1	0.1
Garcinia buchananii	0.2	0.0	1	0.1	0.0	0.3	0.1
Ficus palmata	0.2	0.0	0	0.1	0.0	0.2	0.1
Ficus exasperata	0.2	0.0	1	0.1	0.0	0.1	0.1

Dracaena steudneri	0.1	0.0	0	0.0	0.0	0.1	0.0
Ocotea keynensis	0.1	0.0	0	0.0	0.0	0.1	0.0
Trema orientalis	0.1	0.0	0	0.0	0.0	0.1	0.0
Pouteria altissima	0.0	0.0	0	0.1	0.0	0.1	0.0
Ficus lutea	0.0	0.0	0	0.0	0.0	0.1	0.0
Eugenia bukobensis	2.0	0.3	1	0.1	0.0	0.1	0.2
Blighia unijugata	0.7	0.1	1	0.2	0.0	0.1	0.1
Ficus sycomomorus	0.6	0.1	1	0.1	0.0	0.1	0.1
Breonadia salicina	1.1	0.2	1	0.1	0.0	0.1	0.1
Fagaropsis angolensis	0.4	0.1	0	0.1	0.0	0.1	0.1
Schrebera alata	0.9	0.1	0	0.1	0.0	0.1	0.1
Canthium giordanii	0.8	0.1	1	0.1	0.0	0.1	0.1
Phoneix reclinata	0.8	0.1	1	0.1	0.0	0.1	0.1

Appendix 5.3 Tree basal area and density and species richness and species richness of total woody species (TWS, i.e. tree, shrub and vines), tree species and average tree height of each patch in edge and interior habitats of 18 forest patches in Illubabor Zone south-west Ethiopia.

patches code	Tree basal area (m^2 ha^{-1})		Tree density (ha^{-1})		Tree species richness		TWS species richness		Mean height (m)	
	Interior	Edge	Interior	Edge	Interior	Edge	Interior	Edge	Interior	Edge
SIB1	102.8±2.1	90.3±1.7	475±7.6	350±5.6	28	29	65	79	13.6	8.9
SIB2	105.3±1.7	84.1±0.7	512.5±3.5	625±3.2	32	45	48	63	14.5	10.5
MAF	97.5±3.5	63.8±1.4	418.8±5.5	725±13.0	26	18	44	47	16.8	11.5
ISS	92.5±1.9	69.4±0.9	503.1±8.4	475±5.20	25	32	63	73	16.5	10.8
QOT1	100.3±3.4	80±1.0	509.4±7.1	528.1±6.4	18	28	45	90	15.6	13.5
QOT2	101.6±1.3	86.6±0.7	456.3±2.5	578.3±4.4	34	35	46	74	13.7	10.8
KOD	97.5±1.4	90±0.7	696.9±6.9	640.6±4.5	32	35	54	85	15.5	12
SIM	75.9±1.2	77.2±1.0	734.4±12.	606.3±7.7	29	32	54	78	12.6	12.4
DIK	101.9±1.6	73.4±0.9	878.1±7.8	609.4±8	30	29	54	74	16.7	13.6
JIR	84.4±2.1	68.4±0.8	565.6±8.9	662.5±8.3	24	28	46	75	16.6	10.5
BUS_J	80.9±2.1	60.9±0.9	668.8±8.8	559.4±6.5	20	23	50	66	14.7	12.8
TUL_B	93.8±1.7	59.1±1.1	628.1±6.4	503.1±6.3	35	25	62	66	12.4	10.7
YAR	113.1±2.8	39.7±0.4	593.8±8.1	440.6±3.9	27	35	62	74	15.5	10.6
LAG_H	102.5±2.2	73.4±0.9	768.8±15.8	496.9±5.7	20	32	39	67	13.6	8.6
GOB	96.9±1.9	90±28.4	759.4±9.4	500±3.6	32	45	50	83	14.5	11.5
MAL	90.3±1.6	101.7±1.3	650±9.5	700±9.0	28	24	44	45	15.6	11.5
TUL_M	97.8±1.9	91.9±1.7	740.6±1.7	556.3±8.3	29	28	56	50	15.0	8.7
JOR	107.8±1.4	91.3±1.1	631.3±4.3	512.5±2.7	36	42	55	59	16.8	12.8
Mean					28	31	52	69		

SIB1=Sibu1, SIB2=Sibu2, MAF= Markafa, ISS= Issya, QOT1= Qotora1, QOT2= Qotora 2, KOD= Koda, SIM= Simbir, DIK= Dike, JIR= Jireen, BUS_J= Busa-Jireen, TUL_B= Tulu-Boka, YAR= Yaro, LAG_H= Laga_Chancho, GOB= Gobe, MAL= Malate, TUL_M= Tulu_Mako, JOR= Jorgo

Appendix 5.4 The species score of each axis (axis 1 and axis 2) for tree species.

	Axis 1	Axis 2			
Albizia grandibracteata	-2.0090	0.4605	*Garcinia buchananii*	-0.7776	-3.0356
Albizia gummifera	-0.2751	0.6674	*Grewia ferruginea*	2.2459	-1.5232
Albizia schimperiana	0.2476	1.5048	*Hallea rubro*	3.0090	-2.2363
Allophlus abyssinicus	0.4499	0.5302	*Hypericum revolutum*	-1.5202	3.5323
Allophylus macrobotrys	-2.1024	-0.7371	*Ilex mitis*	-0.7237	1.4723
Apodytes dimidata	0.9443	-0.3881	*Lepidotrichlia volkense*	-0.5651	-1.1891
Bersama abyssinica	-0.1670	-0.0879	*Macaranga capensis*	-0.0505	0.4082
Blighia unijugata	1.4353	-1.5714	*Maesa lanceolata*	-0.3715	0.7422
Breonadia salicina	1.5843	-2.8019	*Maytenus arbutifolia*	-0.2353	0.0441
Bridelia micrantha	-0.4487	5.6420	*Maytenus undata*	-0.7495	-0.0631
Brucea antidysenterica	-0.7780	0.7390	*Milletia ferruginea*	-0.7028	0.2974
Budlleja polystachya	-1.5202	3.5323	*Mimusops kummel*	0.6842	3.6060
Canthium oligocarpa	-0.0764	-1.5545	*Nuxia congesta*	-1.0380	1.3535
Cassipurea malosana	0.0499	-0.4192	*Ocotea keynensis*	-3.1395	-3.2693
Celts africana	-1.5114	-1.3748	*Olea capensis*	0.6679	-0.6208
Chionanthus mildbraedii	-0.6366	-0.1657	*Olea welwitschii*	-0.7776	-3.0356
Coffee arabica	-0.0121	2.6160	*Oxyanthus speciosus*	-0.1888	-0.6959
Cordia africana	1.4432	2.2474	*Pappea capensis*	3.2769	2.2737
Croton macrostachyus	-0.1249	0.1403	*Phoneix reclinata*	-1.9036	-0.2160
Cyathea manniana	-0.9043	1.5531	*Pittosporum viridiflorum*	-0.3798	0.1171
Dalbergia lactea	-1.0117	1.8351	*Afrocarpus falcatus*	0.5343	-1.6767
Diospyros abyssinica	-1.4014	-1.5986	*Polycias fulva*	-0.0621	-0.8072
Dombeya torrida	-1.0147	-0.2358	*Pouteria adolfi-friederici*	1.6776	-0.8595
Dovyalis abyssinica	2.3786	-2.1672	*Pouteria altisna*	-1.9664	-1.0097
Dracaena afromontana	0.2360	0.8062	*Prunus africana*	0.5960	-1.2688
Dracaena fragrans	0.4206	-0.7620	*Psychotria orophila*	-0.8801	-1.1035
Dracaena steu	2.1548	1.1714	*Psydrax parviflora*	1.3260	0.6263
Ehretia cymosa	0.5924	-1.5827	*Rhus ruspoli*	-1.2784	-0.0425
Ekbergia capensis	-1.6905	-0.6314	*Rhus glutinosa*	-1.7746	0.3929
Elaeodendron buchanani	0.2227	-0.1606	*Ritchiea albersii*	0.0714	0.6708
Entada abyssinica	-1.2784	-0.0425	*Rothmannia urcelliformis*	1.9838	1.2806
Eugenia bukobensis	2.1548	1.1714	*Sapium ellipticun*	-1.1928	2.1949
Euphorbia candelabrum	-1.3877	-1.0455	*Schefleria abyssinica*	2.0064	-0.2547
Euphorbia ampliphylla	0.9146	-1.8244	*Schrebera alata*	2.3455	-2.1682
Fagaropsis angolensis	-3.1395	-3.2693	*Senna petersiana*	-1.5620	0.7581
Ficus exasperata	-0.7776	-3.0356	*Syzygium guineense*	-0.5939	0.3955
Ficus lutea	4.0534	2.9208	*Teclea noblis*	-1.7945	-0.1494
Ficus palmata	-2.2090	-1.6559	*Trema orientalis*	-1.4514	2.3699

Ficus sur	-0.1200	0.0548	*Trichilia dregeana*	-0.0078	-0.6135
Ficus sycomorus	2.0405	-0.9744	*Trilipsium madagascariense*	2.1888	1.5746
Ficus thommingii	-0.3621	3.3202	*Vangueria apiculata*	2.6824	0.5318
Ficus vasta	2.3786	-2.1672	*Vepris dainelli*	2.0460	-0.4180
Flacortia indica	-1.0080	-0.0595	*Vernonia amygdalina*	2.9652	0.7591
Galineria saxifraga	0.1926	-0.9099			

Appendix 5.5 Full name of species abbreviations used in Cannonical Correspondance Ananysis

Abbreviation	Full name of species	Abbreviation	Full name of species
Alb_ gr	*Albizia grandibracteata*	*Gar_bu*	*Garcinia buchananii*
Alb_gu	*Albizia gummifera*	*Gre_fe*	*Grewia ferruginea*
Alb_sc	*Albizia schimperiana*	*Hal_ru*	*Hallea rubro*
All_ ab	*Allophlus abyssinicus*	*Hyp_re*	*Hypericum revolutum*
All_ma	*Allophylus macrobotrys*	*Ile_ mi*	*Ilex mitis*
Apo_ di	*Apodytes dimidata*	*Lep_vo*	*Lepidotrichlia volkense*
Ber_ab	*Bersama abyssinica*	*Mac_ca*	*Macaranga capensis*
Bli_ un	*Blighia unijugata*	*Mae_la*	*Maesa lanceolata*
Bre_ sa	*Breonadia salicina*	*May_ar*	*Maytenus arbutifolia*
Bri_mi	*Bridelia micrantha*	*May_un*	*Maytenus undata*
Bru_an	*Brucea antidysenterica*	*Mil_fe*	*Milletia ferruginea*
Bud_po	*Budlleja polystachya*	*Mim_ku*	*Mimusops kummel*
Can_ol	*Canthium oligocarpa*	*Nux_co*	*Nuxia congesta*
Cas_ma	*Cassipurea malosana*	*Oco_ke*	*Ocotea keynensis*
Cel_af	*Celts africana*	*Ole_ca*	*Olea capensis*
Chi_mi	*Chionanthus mildbraedii*	*Ole_we*	*Olea welwitschii*
Cof_ar	*Coffee arabica*	*Oxy_sp*	*Oxyanthus speciosus*
Cor_af	*Cordia africana*	*Pap_ca*	*Pappea capensis*
Cro_ma	*Croton macrostachyus*	*Pho_re*	*Phoneix reclinata*
Cya_ma	*Cyathea manniana*	*Pit_vi*	*Pittosporum viridiflorum*
Dal_la	*Dalbergia lactea*	*Afr_fa*	*Afrocarpus falcatus*
Dio_ab	*Diospyros abyssinica*	*Pol_fu*	*Polycias fulva*
Dom_to	*Dombeya torrida*	*Pou_ad*	*Pouteria adolfi-friederici*
Dov_ ab	*Dovyalis abyssinica*	*Pou_al*	*Pouteria altisna*
Dra_af	*Dracaena afromontana*	*Pru_af*	*Prunus africana*
Dra_fr	*Dracaena fragrans*	*Psy_or*	*Psychotria orophila*
Dra_st	*Dracaena steu*	*Psy_pa*	*Psydrax parviflora*
Her_cy	*Ehretia cymosa*	*Rhu_ru*	*Rhus ruspoli*
Ekb_ca	*Ekbergia capensis*	*Rhu_gl*	*Rhus glutinosa*
Ela_bu	*Elaeodendron buchanani*	*Rit_al*	*Ritchiea albersii*
Ent_ab	*Entada abyssinica*	*Rot_ur*	*Rothmannia urcelliformis*

Eug_bu	*Eugenia bukobensis*	*Sap_el*	*Sapium ellipticun*
Eup_ca	*Euphorbia candelabrum*	*Sch_ab*	*Schefleria abyssinica*
Eup_am	*Euphorbia ampliphylla*	*Sch_al*	*Schrebera alata*
Fag_an	*Fagaropsis angolensis*	*Sen_pe*	*Senna petersiana*
Fic_ex	*Ficus exasperata*	*Syz_gu*	*Syzygium guineense*
Fic_lu	*Ficus lutea*	*Tec_no*	*Teclea noblis*
Fic_pa	*Ficus palmata*	*Tre_or*	*Trema orientalis*
Fic_su	*Ficus sur*	*Tri_dr*	*Trichilia dregeana*
Fic_sy	*Ficus sycomorus*	*Tri_ma*	*Trilipsium madagascariense*
Fic_th	*Ficus thommingii*	*Van_ap*	*Vangueria apiculata*
Fic_va	*Ficus vasta*	*Vep_da*	*Vepris dainelli*
Fla_in	*Flacortia indica*	*Ver_am*	*Vernonia amygdalina*
Gal_sa	*Galineria saxifraga*		

Appendix 5.6 The habitat characteristic, frequency distribution, leaf and stem morphology of epiphyte species between edge and interior.

Species	**Habitat**	**Frequency**		**Morphology**	
		Interior	edge	Leaf texture	Stem
Aerangis brachycarpa	Dense shade on the lower trunk	34	0	Leathery	Woody
Aerangis thomsonii	Deep shade on lower trunk	17	0	leathery	Woody
Ancistrorhynchus metteniae	Evergreen forest (shaded forest)	39	0	Leathery	Herbaceous
Angraecopsis holochila	In forest	14	1	Herbaceous	Pseudobulbs
Angraecopsis parviflora	Evergreen forest and riverine forest	13	4	Fleshy	Semi-woody
Ansellia africana	Open woodland	12	9	Herbaceous	Pseudobulbs/ woody
Arthropteris monocarpa	Deep shade and moist forest	78	3	Herbaceous	Herbaceous
Arthropteris orientalis	Open woodland	11	8	Herbaceous	Herbaceous
Asplenium aethiopicum	Under the shade	42	13	Herbaceous	Herbaceous
Asplenium hypomelas	Dense shade in moist forest	24	0	Herbaceous	Herbaceous
Asplenium sandersonii	Moist forest	75	0	Herbaceous	Herbaceous
Asplenium theciferum	Shaded forest	103	14	Herbaceous	Herbaceous
Bolusiella iridifolia	Forest among lichens and on rocks in river	21	0	Herbaceous	Herbaceous
Bulbophyllum josephi	Submonane forest and woodland	20	12	Herbaceous	Pseudobulbs
Bulbophyllum lupulinum	Forest and riverine forest	18	0	Herbaceous	Pseudobulbs
Bulbophyllum maximum	Open woodland and riverine forest	15	9	leathery	Pseudobulbs
Bulbophyllum sandersoni	Woodland and forest	6	4	leathery	Pseudobulbs
Calyptrochilum christyanum	Forest, wooded grassland	16	45	Fleshy	Woody

Canarina eminii	Hang on tree trunk in dense forest	5	1	Herbaceous	Semi-woody
Corymboriks corymbis	Rainforest/base of trees (deeply shaded	13	0	Herbaceous	Semi-leathery
Culcasia falcifolia	In shade evergreen forest	12	8	Leathery	Semi-woody
Cyrtorchis arcuata	Shaded forest, bushland and rocks in forest	9	45	Herbaceous	Semi-woody
Diaphananthe adoxa	Upland evergreen forest	5	0	Herbaceous	Semi-woody
Diaphananthe fragrantissima	Forest, bushland and rocky	21	30	Leatheey	Semi-woody
Diaphananthe rohrii	Montane forest (deep shade)	10	0	Herbaceous	Herbaceous
Diaphananthe tenuicalcar	In forest	40	10	Herbaceous	Herbaceous
Drynaria volkensii	In forest and forest margin	85	23	Herbaceous	Herbaceous
Elaphoglossum acrostidoides	Moist montane forest	14	0	Herbaceous	Herbaceous
Elaphoglossum deckeni	Moist montane forest	23	0	Herbaceous	Herbaceous
Elaphoglossum lastii	Moist montane forest	7	0	Herbaceous	Herbaceous
Huperzia dacrydioides	Shaded evergreen moist forest	5	2	Leathery	Semi-woody
Lepisorus excavatus	Shade often on rocks	3	2	Herbaceous	Herbaceous
Liparis abyssinica	Shaded forest	5	0	fleshy	Herbaceous
Loxogramme abyssinica	Under the shade on rocks	5	0	Herbaceous	Herbaceous
Microcoelia globulosa .	Margin of evergreen forest, riverine forest, secondary forest	42	40	Leathery	Herbaceous
Peperomia rotundifolia	In forest and forest margin on tree trunk	122	26	Leathry	Succulent
Peperomia abyssinica	In forest and forest margin on tree trunk	10	5	Leathry	Succulent
Peperomia tetraphylla	In forest and forest margin on tree trunk	123	28	Leathry	Succulent
Pleopeltis macrocarpa	In deep shade	29	2	Herbaceous	Herbaceous
Polystachya bennettiana	Open woodland and riverine forest	25	1	Herbaceous	Semi-woody
Polystachya cultriformis	under the shade of rainforest	5	0	Leathry	Pseudobulbs
Polystachya eurychila	Riverine forest/wet rock	5	4	Herbaceous	Fleshy
Polystachya fusiformis	Rainforest/wet rocks	18	0	Herbaceous	Pseudobulbs
Polystachya lindblomii	In dense forest and forest Margin	8	5	Herbaceous	Pseudobulbs
Polystachya steudneri	Woodland/dryish scrub	37	4	Herbaceous	Pseudobulbs
Polystachya tessellata	Rainforest (deeply shaded)	3	0	Herbaceous	Pseudobulbs
Pyrrosia schimperiana	Shaded area in forest on tree trunk	10	0	Herbaceous	Herbaceous
Thelypteris confluens	Moist shaded forest, swampy area	6	0	Herbaceous	Herbaceous
Tridactyle bicaudata	Montane forest and riverine forest	8	0	Semi-leathery	Semi-woody
Vittaria volkensii	Moist forest on tree trunk	10	0	semi-Leathery	Herbaceous

Appendix 5.7 The habitat characteristic, rhizome type, frond arrangement and frequency distribution of fern species in edge and interior

Species	Rhizome	Frond type	Shade tolerant	Frequency	
				Interior	Edge
Adiantum lunulatum	Erect	Spaced	Forest margin	28	60
Arthropteris orientalis	Erect	Spaced	Forest margin	39	65
Athyrium filix-femia	Erect	Spaced	Forest margin	25	65
Cheilanthes farinosa	Erect	Spaced	Forest margin	6	102
Dicranopteris linearis	Creeping	Spaced	Forest margin	47	51
Doryopteris concolor	Creeping	Spaced	Forest margin, disturbed area	23	65
Hypolepis sparsisora	Erect	spaced	Forest margin	32	60
Pteridium aquilinus	Erect	Spaced	Forest margin	0	43
Arthropteris monocarpa	Creeping	Spaced	In the forest, under shade	31	13
Asplenium aethiopicum	Creeping	Spaced	Forest habitat, underdeep shade	31	56
Asplenium anisophyllum	Erect	Tuffted	Deep shade, Shade tolerant	57	22
Asplenium erectum	Erect	Tufted	Forest-habitat	25	9
Asplenium formosum	Erect	Tufted	Forest-habitat	65	0
Asplenium gemmiferum	Erect	Tuffted	Forest-habitat	107	9
Asplenium hypomelas	Erect	Tufted	Forest-habitat	91	48
Asplenium lunulatum	Erect	Tufted	Forest-habitat	105	65
Asplenium monanthes	Erect	Tufted	Forest-habitat	36	51
Asplenium sandersoni	Rhizome	Tofted	Forest-habitat	67	0
Asplenium theciferum	Erect	Tufted	Forest-habitat	109	36
Blotiella glabra	Creeping	Spaced	Forest-habitat	58	38
Conigramme africana	Erect	Spaced	Forest-habitat	60	0
Drynaria volkensii	Creeping	Tufted	Forest-habitat	57	24
Dryopteris athamantica	Creeping	Tufted	Forest-habitat	49	75
Loxogramma abyssinica	Erect	Tufted	Forest-habitat	21	5
Marsilia minuta	Creeping	NA	Forest-habitat	87	24
Microlepia speluncae	Creeping	Spaced	Forest-habitat	63	50
Pleopeltis macrocarpa	creeping	spaced	Forest-habitat	48	16
Tectaria gemmifera	Erect	Spaced	Forest-habitat	52	32
Marattia fraxinea	Creeping	Tufted	Forest-habitat	64	54
Thelypteris confluens	Creeping	Spaced	Forest-habitat	48	0
Asplenium sp	creeping	spaced	Forest-habitat	21	0

Appendix 5.8 Habitat characteristics of woody species

Species	Habitat Chracteristics	Functional groups
Justcia schimperiana	Understorey shrub, open area, forest margin, secondary forest	Wide-habitat species
Justicia betonica	Understorey of evergreen forest, forest margin, clearings, recently opend forest	Wide-habitat species
Acalypha racemosa	Understory of riverine forest, secondary forest, more open area, dry habitat	Wide-habitat species
Clausena anisata	Montane forest, forest margin, understorey shrub in moist forest, common in secondary forest	Wide-habitat species
Clutia abyssinica	Along margins of evergreen forest, riverine forest and high rainfall woodland and clearings	Wide-habitat species
Hibiscus dongolensis	Marshy grassland and secondary forest	Forest-margin species
Hibiscus ludwigii	Afrocarpus forest, on edge, upland bushland	Forest-margin species
Hibiscus macranthus	Forest edge, woodland and secondary forest	Forest-margin species
Hilleria latifolia	Forest margin	Forest-margin species
Hypericum quartinianum	Rocky place, gullery and river bank decidous woodland	Wide-habitat species
Kotschya (tiro)	Forest margins, grassland swampy area	Forest-margin species
Phytolacca dodecandera	Forest margin, open area and cultivated land	Forest-margin species
Plectranthus ornatus	semi-shade	Forest-margin species
Plectranthus punctatus	Montane gassland, open areas and forest margin	Forest-margin species
Pterolobium setllatum	Riverine thickts and rocky area	Forest-margin species
Pycnostachys abyssinicus	Forest margin, life fence, open area	Forest-margin species
Pycnostachys eminii	Forest margin, fallowland, hedgerows	Forest-margin species
Rhamnus Prinoides	Upland forest, usually on edge, riverine forest, secondary forest	Forest-margin species
Rhamnus Stado	Upland forest at edge, woodland and grassland	Forest-margin species
Rhus quartianana	Evergreen thickets and along river and streams	Forest-margin species
Rubus apetalus	Open forest area or disturbed, forest margin	Forest-margin species
Rubus steu	Disturbed forest area, forest margin, woodland and secondary forest	Forest-margin species
Sparmania macrocarpa	Edge of montane forest, scrub land scrub grassland	Forest-margin species
Solanacieo mannii	Open forest area,forest margin, rocky area in high rainfall area	Forest-margin species
Solanecio gigas	In disturbed forest, forest margin, open area, in high rainfall area	Forest-margin species
Solanium schimperiana	Forest margin, degraded area and woodland	Forest-margin species
Abutilon martinianum	Open forest area, forest margin and woodland	Forest-margin species
Acalypha marissima	Open or forested slope	Forest-margin species
Acalypha ornata A.Rich.	Riverine forest, more open rocky slopes and wooded grassland	Forest-margin species

Acanthus eminens	Open area, forest margin	Forest-margin species
Calpurnia urea	Along evergreen forest margin, riverine forest and disturbed area	Forest-margin species
Crotalaria rosenii	Open area inside forest and undershade	Forest-margin species
Crotalria milbraedii	Montane forest margins, bushlands, along roadside	Forest-margin species
Discoppodium	Forest margin, disturbed area and secondary forest	Forest-margin species
Dodonaea angustifolia	edge of upland forest, upland bushland, grassland and secondary forest	Forest-margin species
Euphorbia schimperiana	Forest margin, secondary forest	Forest-margin species
Ficus capre	Component of ripariana forest and drier area	Forest-margin species
Maytenus obscura	Forest margin, disturbed area and secondary forest	Forest-margin species
Microglossa pyrifolia	At the edge of drier forest, secondary forest	Forest-margin species
Myrsine africana	Margin of evergreen forest, woodland and rocky areas	Forest-margin species
Ocimum lamifolia	Forest margin, open area, grazing land and bushland thickts	Forest-margin species
Pavonia urens	Edge, path and clearing in upland forest, riverine forest ruderal	Forest-margin species
Pentas lanceolata	Open deforested area, woodland	Forest-margin species
Vernonia auriculifera	Forest margin, grassland	Forest-margin species
Vernonia glabra	High rain fall woodland, riverine forest	Forest-margin species
Vernonia hochstetteri	Wooded land, open area	Forest-margin species
Vernonia myriantha	Forest margin, grassland	Forest-margin species
Vernonia urticifolia	Disturbed area, forest margin, woodland	Forest-margin species
Acanthopale pubescens	Shaded forest area, in the understorey of evergreen forest, often in large numbers	Forest-habitat species
Acalypha psilostachya Hochst.	Moist montane forest, allong valley botoms, forest margins	Forest-habitat species
Acalypha brachystachya	Riverine and montane forest in deep shade	Forest-habitat species
Coffee arabica	Shaded forest area as understorey shrub	Forest-habitat species
Embelia schimperi	Upland thickt, prinary forest as understorey shrub	Forest-habitat species
Erythrococca trichogyne	Moist evergreen forest	Forest-habitat species
Maytenus gracilipes	Understorey shrub and secondary forest	Forest-habitat species
Oxanthus le	understory of shrub of evergreen forest	Forest-habitat species
Pavetta abyssinica	Understorey shrub in high canopy forest	Forest-habitat species
Pavetta oliveriana	Understorey shrub in high canopy forest and thickts	Forest-habitat species
Solanium uv	Understorey forest shrub	Forest-habitat species
Abutilon cecillii	Secondar scrub after former montane forest	Forest-habitat species
Clerodendrum myricoides	Forest margin, woodland and grassland	Forest-margin species
Crotalaria emarginella		

Hibiscus calyphyllus		
Malva verticillata		
Phyllanthus limmuensis	Under forewst shade, forest margin	
Phyllanthus ovalifolius		Forest-habitat species
Sparmannia ricinocarpa		
Albizia gummifera	In lowland and upland rain-forest, riverine forest, and in open habitats near forests. It occasionally appears as a pioneer species in forests and in thickets.	Wide-habitat species
Albizia schimperiana	In Evergreen forest and margin of forest	Wide-habitat species
Allophylus abyssinicus	In forest, edge and riverine forest	Wide-habitat species
Allophylus macrobotray	Rainforest, riverine forest and woodland	Wide-habitat species
Apodytes dimidata	Upland rain forest, secondary forest and riverine forest	Forest-habitat species
Bersama abyssinica	In fores, edge, cultivated land	Wide-habitat species
Brucea antidysenterica	Montane forest, evergreen forest, forest margin, secondary forest and montane grassland	Wide-habitat species
Celts africana	Dry and moist evergreen forest, riverine forest and dry rocky hills	Wide-habitat species
Dombeya torrida	Montane forest, shaded and margin	Wide-habitat species
Dracaena afromontana	Montane forest, forest margin, cultivated land	Wide-habitat species
Elaeodendron buchanani	Evergreen forest and thickets, forest fringing swamps, deciduous woodland, grassland	Wide-habitat species
Euphorbia ampliphylla	Moist montane forest, life fence	Wide-habitat species
Ficus ovata	Riverine forest, upland rainforest and evergreen bushland	Wide-habitat species
Ficus sur	Riverine forest, upland rainforest and secondary bushland	Wide-habitat species
Ficus thonningii	Rainforest, dry evergreen forest and riverine forest	Wide-habitat species
Hallea rubro	moist forest, forest margin	Wide-habitat species
Ilex mitis	Riverine, Montane forest, moist woodland	Wide-habitat species
Manilkara bu	Inside forest, forest margin, riverine forest	Wide-habitat species
Millettia ferruginea	In montane forest, culticated land, coffee managed forest, forest margin	Wide-habitat species
Nuxia congesta	A pioneering species, in forest, woodland or grassland, often in rocky places	Wide-habitat species
Olea capensis L. ssp macrocarpa	Forest, forest margin and secondary forest	Wide-habitat species
Olea welwitschii	In forest and forest margin	Wide-habitat species
Pittosporum viridiflorum	Forest, forest margin, cultivated land	Wide-habitat species
Polycias fulva	In montane forest, forest margin, piomeer in clearings	Wide-habitat species
Pouteria adolfi-friederici	Componenet of afromontane forest, in forest, forest margin	Wide-habitat species

Prunus africana	High rain forest, riverine forest, forest margin	Wide-habitat species
Ritchiea albersii	Understorey shrub of evergreen forest, forest margin and dense woodland	Wide-habitat species
Rytigina neglecta	Understorey shrub and forest edge	Wide-habitat species
Sapium ellipticun	Along streens in area of deciduous woodland, margin of moist forest and occasionaly forest component	Wide-habitat species
Schefleria abyssinica	Upland rain forest, secondary forest and woodland	Wide-habitat species
Schrebera alata	Forest, forest margin and secondary forest	Wide-habitat species
Syzygium guineense	Componenet of afromontane green forest, in forest, forest margin	Wide-habitat species
Teclea noblis	Dry evergreen bushland, woodland and moist montane forest	Wide-habitat species
Afrocarpus falcatus	Component of upland forest, forest margin and as a solitrary tree in cultivated	Forest-margin species
Alangium chinense	Pioneer species in cleared area and upland rain forest	Forest-margin species
Anthocleista schweinfurthii	In forest and forest margin, moist woodland	Forest-margin species
Bridelia micrantha	Riverine forest margin	Forest-margin species
Buddleja polystachya	Forest margin, cultuvatedland, life fence, woodland	Forest-margin species
Cordia africana	Forest margin, sometines inside forest as coffee shade, woodland, grassland	Forest-margin species
Croton macrostachyus	Forest margin, secondary woodland, disturbed areas and along the road	Forest-margin species
Deinbollia kilimandisch	Lowland mixed forest	Forest-margin species
Dovyalis aby	Highland forest margin, riverine forest and montane grassland	Forest-margin species
Dracaena steudneri	Margin of evergreen forest	Forest-margin species
Ehretia cymosa	Secondary forest, forest margin and grassland	Forest-margin species
Ekbergia capensis	In montane forest	Forest-margin species
Eugenia bukobensis	Sand-dune thicket and sometines in forest and grassland	Forest-margin species
Euphorbia candilabrium	Secondary foret, forest margin and grassland, cultivated area	Forest-margin species
Ficus exasperata	Secondary scrub and forest edge	Forest-margin species
Ficus palmata	Riverine forest, upland rainforest and evergreen bushland	Forest-margin species
Ficus vallis-choudae	Riverine forest or scrub	Forest-margin species
Flacortia indica	Forest edge, woodland and secondary forest	Forest-margin species
Grewia ferruginea	Gullery forest, along rivers and open woodland	Forest-margin species
Maesa lanceolata	Forest margin of forest (pioneer species)	Forest-margin species
Maytenus addat	Endemic to afromontane forest, forest margin	Forest-margin species

Maytenus arbutifolia	Forest margin, fallowland, sometimes under forest at the edge	Forest-margin species
Maytenus undata	Along the margin of forest and woodland	Forest-margin species
Mimusops kummel	Riverine forest, forest margin, ripariana forest	Forest-margin species
Nuxia opositfolia	Riverine forest, and rocky areas and forest margin	Forest-margin species
Opilia campestris	Woodland	Forest-margin species
Pappea capensis	Wooded land, edge of forests	Forest-margin species
Phoneix reclinata	Riverine forest, forest margin, woodland, swampy area	Forest-margin species
Rhus ruspolii	Margins of montane forest and woodland	Forest-margin species
Senna petersiana	Forest margin and grassland	Forest-margin species
Trema orientalis	Riverine forest, pioneer in clearings, forest edge	Forest-margin species
Trichilia emitica	Savanna woodland and ripariana woodland	Forest-margin species
Vangueria apiculata	Forest margin, woodland and grassland	Forest-margin species
Vernonia amy	Forest margin, cultivated land, woodland	Forest-margin species
Ficus lutea	Riverine forest, upland rainforest and evergreen bushland	Forest-habitat species
Antiaris toxicara	Lowland forest	Forest-habitat species
Blighia unijugata	Evergreen lowland forest, upland forests and shade tree in coffee forest	Forest-habitat species
Breonadia salicina	riverine forest	Forest-habitat species
Cassipurea malosana	Upland rainforest and dry upland forest	Forest-habitat species
Celts philippensis	Wet forest, understorey forest, secondary association	Forest-habitat species
Celts toka	Riverine forest	Forest-habitat species
Celts zenkeri	Wet evergreen forest and seni-deciduous woodland	Forest-habitat species
Chionanthus mildbraedii	Understorey forest shrub, deeply shaded	Forest-habitat species
Cyathea manniana	Along river and streams in deep shade	Forest-habitat species
Dalbergia lactea	In montane forest, forest margin	Forest-habitat species
Diospyros abyssinica	In evergreen forest,	Forest-habitat species
Dracaena fragrans	Understorey shrub of evergreen forest. Deep shade	Forest-habitat species
Fagaropsis angolensis	Upland rainforest (Afrocarpus forest)	Forest-habitat species
Galineria saxifraga	Inside forest as understorey small tree	Forest-habitat species
Garcinia buchananii	Riverine forest	Forest-habitat species
Lepidotrichlia volkensii	Understory of moist forest	Forest-habitat species
Morus mesozygia	Humid forest	Forest-habitat species
Ocotea keynensis	Upland forest,	Forest-habitat species
Oxyanthus speciosus	Inside forest as understorey small tree	Forest-habitat species
Pouteria altisma	Lowland montane forest and forest margin	Forest-habitat species
Psychotria orophila	Under canopy of wet montane forest and becoming common at margins	Forest-habitat species

Psydrax parviflora	Under canopy of wet montane forest and becoming common at margins	Forest-habitat species
Rothmannia urcelliformis	Understorey of evergreen forest	Forest-habitat species
Sarcocephalus latifolius	Small tree as understorey of forest/montane	Forest-habitat species
Trilipsium madagascariense	South-west Ethiopia Upland rainforest	Forest-habitat species
Vepris dainelli	Understorey of moist montane forest	Forest-habitat species
Ficus mucuso	Lowland rainforest and coffee plantation	Forest-habitat species
Ficus sycomorus	riverine forest	Forest-habitat species
Filicium decipiens	Dense forest and riverine forest	Forest-habitat species
Trichilia dregeana	Mid-altitude rain forest	Forest-habitat species
Canthium oligocarpa	Inside forest, forest margin, riverine forest	Wide-habitat species
Canthium sp	Inside forest	Wide-habitat species
Macaranga capensis	Under canopy of wet montane forest and becoming common at margins	Wide-habitat species
Rhus glutinosa	Forest margin and woodland	Forest-margin species

Appendix 5.9 Habitat characteristics of some herbaceous species

Species	Habitat Characteristics	Functional groups
Achyranthes aspera	Widespread, hedges, thickets, shaded habitat, bushland and riverine forest	Wide-habitat species
Achyrospermum schimperi	Partial shade, forest, forest margins, bushland, coffee plantation	Wide-habitat species
Acmella caulirhiza	Wet place, grassy slopes, forest floors, along stream banks	Wide-habitat
Ageratum conyzoides	Wetter area, shade area, eroded soils, grassland, along waterways disturbed land, grazing field	Forest-margin/Wide-habitat species
Ajuga intigrifolia	Wetter areas, disturbed grassland, along road sides.	Forest-margin/Wide-habitat species
Alchemilla cryptantha	Moist shade forest, wet grassland, along stream	Forest-habitat species
Alchemilla rothii	Wet area, montane forest, bamboo ticket, evergreen bushland	Forest-habitat species
Arisaema flavum	Open area, under shade, disturbed places	Forest-margin/Wide-habitat species
Asystasia gangetica	Forest, margin, partial shade, roadside, streamside	Forest-margin/Wide-habitat species
Bidens pilosa	Under forest, open place, road side, disturbed area	Forest-margin/Wide-habitat species
Cardamine trichocarpa	Open area, under trees, disturbed area	Forest-margin/Wide-habitat species
Centella asiatica	Under shade, swampy area, open land under trees	Forest-habitat species

Kalanchoe petitiana	Open evergreen, forest margin, under trees, disturbed area	Forest-margin/Wide-habitat species
Cyperus esculenta	In a forest, forest margin, shade areas	Forest-habitat species
Didymodoxa cafra	Under shade, in forest, wetland, river bank	Forest-habitat species
Hypoestes forskaoli	Roadside, grassland, open woodland, disturbed montane forest	Forest-margin/Wide-habitat species
Hypoestes triflora	Under the shade of forest, woodland roadside, waterways	Forest-habitat species
Justicia heterocarpa	Woodland, undershade, degraded acacia woodland	Forest-margin/wide-habitat species
Justicia striata	Open area, disturbed land,	Forest-margin/wide-habitat species
Pilea tetraphylla	Open area, forest margin, disturbed area	Forest-margin/wide-habitat species
Piper capense	Under shade of montane forest	Forest-habitat species
Spilanthes costata	Under forest shade, wet area	Forest-habitat species

Appendix 5.10 Shrub species ≥0.50 average abundance in edge and interior habitats of 18 forest patches (from SIMPER analysis)

Interior	Average Abundance	Average abundance	Average Dissimilarity	% Contribution
Species	Edge	Interior		
Clausena anisata	0.67	0.61	1.41	1.85
Rytigina neglecta	0.72	0.56	1.47	1.93
Maytenus gracilipes	0.44	0.56	1.49	1.95
Erythrococcaa trichogyne	0.5	0.5	1.48	1.94
Cyathula uncinulata	0.44	0.5	1.45	1.91
Solanecio mannii	0.28	0.5	1.46	1.91
Pavetta oliveriana	0.22	0.5	1.45	1.91
	0.47±0.07	0.53±0.02		
Edge				
Species	Edge	Interior		
Rytigina neglecta	0.72	0.56	1.47	1.93
Maytenus gracilipes	0.72	0.17	1.79	2.35
Clausena anisata	0.67	0.61	1.41	1.85
Vernonia auriculifera	0.61	0.39	1.49	1.95
Lippia adoensis	0.61	0.06	1.45	1.91
Acanthus eminens	0.56	0.28	1.53	2
Clerodendrum myricoides	0.56	0.11	1.44	1.88
Ocimum lamifolia	0.56	0.06	1.34	1.75
Lantana trifolia	0.56	0	1.55	2.04

Erythrococcaa trichogyne	0.5	0.5	1.48	1.94
Pavetta abyssinica	0.5	0.28	1.46	1.92
Hibiscus micranthus	0.5	0.28	1.35	1.77
Calpurnia urea	0.5	0.17	1.35	1.77
	0.58±0.2	0.27±0.06		

Appendix 5.11 Vine species with ≥0.50 average abundance in edge and interior habitats of 18 forest patches (from SIMPER analysis)

Interior	Average Abundance	Average abundance	Average Dissimilarity	% Contribution
Species	Edge	Interior		
Simlax aspera	0.06	0.56	2.15	2.7
Ipomoea carica	0.56	0.54	2.15	2.69
Hippocratea goetzei	0.33	0.52	2.11	2.65
Dioscorea bulbifera	0.11	0.5	1.95	2.44
Tiliachora troupinii	0.44	0.5	2.02	2.53
	0.3±0.09	0.52±0.01		
Edge				
Species	Edge	Interior		
Clematis hirsuta	0.78	0.22	2.83	3.54
Ipomoea carica	0.56	0.5	2.15	2.69
Cyphostemma adenocaule	0.53	0.17	2.11	2.64
Ipomoea tenuirostris	0.51	0.17	2.08	2.6
Helinus mystacinus	0.51	0.11	2.15	2.7
Clematis sinensis	0.5	0.17	1.95	2.45
Gouania longispicata	0.5	0.11	1.95	2.44
	0.56±0.04	0.21±0.05		

Appendix 5.12 Angiosperm herb species with ≥ 0.50 average abundance in interior and edge habitats of 18 forest patches

Interior	Average Abundance	Average abundance	Average Dissimilarity	% Contribution
Species	Edge	Interior		
Didymodoxa cafra	0.23	1	1.37	2.49
Centella asiatica	0.47	0.95	0.95	1.73
Cyperus esculenta	0.35	0.93	1.11	2.03
Pilea tetraphylla	0.45	0.78	0.93	1.69
Bidens pilosa	0.92	0.76	0.52	0.94
Justicia striata	0.64	0.73	0.78	1.43

Ageratum conyzoides	0.86	0.71	0.63	1.14
Achyranthes aspera	0.95	0.68	0.61	1.11
Justicia heterocarpa	0.53	0.66	0.87	1.59
Acmella caulirhiza	0.54	0.65	0.87	1.59
Piper capense	0.06	0.65	1.09	1.98
Achyrospermum schimperi	0.65	0.64	0.83	1.51
Hypoestes forskaoli	0.47	0.64	0.91	1.65
Alchemilla cryptantha	0.42	0.6	0.92	1.68
Alchemilla rothii	0.55	0.58	0.88	1.6
Cardamine trichocarpa	0.31	0.56	0.93	1.69
Ajuga intigrifolia	0.44	0.51	0.89	1.62
Hypoestes triflora	0.47	0.5	0.89	1.62
	0.52±0.05	0.69±0.05		

Edge

Species	Edge	Interior		
Achyranthes aspera	0.95	0.68	0.61	1.11
Bidens pilosa	0.92	0.76	0.52	0.94
Ageratum conyzoides	0.86	0.71	0.63	1.14
Arisaema flavum	0.68	0.44	0.93	1.69
Achyrospermum schimperi	0.65	0.64	0.83	1.51
Justicia striata	0.64	0.73	0.78	1.43
Spilanthes costata	0.57	0.38	0.91	1.67
Asystasia gangetica	0.57	0.04	1	1.82
Alchemilla rothii	0.55	0.58	0.88	1.6
Acmella caulirhiza	0.54	0.65	0.87	1.59
Apium sp	0.54	0.28	0.92	1.67
Justicia heterocarpa	0.53	0.66	0.87	1.59
Chalanchoe petitiana	0.51	0.27	0.88	1.61
	0.65±0.04	0.52±0.06		

Appendix 5.13 Densityof vascular epiphytes in interior and edge habitats of 18 forest patches in Illubabor Zone of south-west Ethiopia: Species richness, SR; density of host trees (bearing epiphytes) (no. 400 m-2), hD; mean DBH of host trees (cm), mDBH; density of epiphyte individuals or clumps (no. 400 m-2), epD and attributes of the patches associated with fragmentation and environment (are, shape, edge density, rainfall, altitude and D. score = disturbance).

Site	Interior/SR	Edge /SR	Interior hD	Edge hD	Interior m DBH	Edge mDBH	Interior epD	Edge epD	Area of patches	D. score interior	D. score edge	Shape index	Edge density	Altitude	Rainfall
Sibu1	26	17	60	46	66	94.3	123	51	37	3	3.1	2.3	137.1	1942	1576
Sibu2	24	7	45	19	71.6	82.2	93	35	52	2.4	2	2.5	125.7	2198	1576
Markafa	23	14	43	21	41.4	124	62	39	55	3	2.3	3.2	152.4	1918	1576
Issya	34	15	66	15	67.3	151.4	130	85	109	4	1.4	4.9	162.6	2197	1928
Qotora1	29	12	68	20	117.4	106	158	67	2000	2	1	8.4	62.9	2179	1928
Qotora2	33	11	64	14	125.5	65	146	42	164	1.1	3.3	2.7	76.3	2312	1928
Koda	36	16	70	14	102.3	82.8	145	30	1958	0	4	7.9	63.3	2296	1780
Simbir	34	12	51	9	109.3	110.4	138	35	248.7	3.1	3.1	4.9	111.8	2268	1780
Dike	29	16	48	18	116.8	118	118	36	656	1	3	5.6	78	2422	1811
Jireen	27	19	45	17	124.4	67.3	107	33	52	1.3	4	2.7	132	2356	1811
Busa-Jireen	31	15	56	10	63.5	104.5	106	63	151	3	3.3	3.6	105	2156	1811
Tulu-Boka	33	16	60	9	91.2	89.2	121	32	304	2	4	4.7	96.2	2188	1662
Yaro	29	15	60	12	84.3	62.8	147	24	42.8	2.3	4	2.4	128.9	2141	1828
Laga-Chancho	27	11	55	8	76	55.2	99	32	85.2	2	4	2.1	110	1951	1576
Gobe	28	14	49	20	65	70	82	43	620	3	4	7.8	111.6	2005	1576
Malate	29	15	56	15	99.3	67.7	113	32	174	1.4	4	3.7	99.8	2044	1828
Tulu-Mako	26	15	54	10	77.2	67.7	98	33	165.6	0	4	2.9	81.2	2319	1828
Jorgo	22	15	45	18	63.2	69.2	75	38	1301	1	3	3.1	30.7	2192	1662

Appendix 5.14 The density of geophytic fern species (numbers of individuals per 400 m^2 plot) in interior and edge habitats of 18 forest patches and attributes of the patches associated with fragmentation and environment (area, D. score= disturbance score, shape, edge density, rainfall and altitude).

Patch	Interior SR	Edge SR	Density/plot Interior	Density/plot edge	Area of patches	D. score interior	D. score edge	Shape index	Edge density	Altitude	Rainfall
Sibu1	28	25	82	71	37	2.7	2.5	2.3	137.1	1942	1576
Sibu2	37	23	85	51	52	2.5	3.5	2.5	125.7	2198	1576
Markafa	29	25	101	71	55	3	3.2	3.2	152.4	1918	1576
Issya	34	24	110	64	109	1.5	3.5	4.9	162.6	2197	1928
Qotora1	33	22	114	49	2000	2	3.8	8.4	62.9	2179	1928
Qotora2	27	20	103	55	164	1	2.9	2.7	76.3	2312	1928
Koda	33	27	115	76	1958	0	2.2	7.9	63.3	2296	1780
Simbir	26	26	127	55	248.7	3	3.7	4.9	111.8	2268	1780
Dike	36	23	137	82	656	1.5	1.5	5.6	78	2422	1811
Jireen	25	21	134	88	52	1.5	1.5	2.7	132	2356	1811
Busa-Jireen	28	21	107	53	151	2.5	3.5	3.6	105	2156	1811
Tulu_Boka	29	21	121	60	304	2	2.8	4.7	96.2	2188	1662
Yaro	23	18	130	62	42.8	2	2.7	2.4	128.9	2141	1828
Laga-Chancho	26	17	61	34	85.2	3.5	4	2.1	110	1951	1576
Gobe	27	20	117	48	620	2	4	7.8	111.6	2005	1576
Malate	29	22	78	60	174	3.3	2.6	3.7	99.8	2044	1828
Tulu_Mako	31	25	77	43	165.6	2.7	4	2.9	81.2	2319	1828
Jorgo	29	17	82	54	1301	3.5	3.7	3.1	30.7	2192	1662

Appendix 5.15 Density of epiphytes under each functional groups

Functional groups	Density				
Forest-habitat species	Interior	Edge	**Woody as well as pseudobulbus stem**	Interior	Edge
Aerangis brachycarpa	7	0	*Aerangis brachycarpa*	7	0
Aerangis thomsonii	8	0	*Aerangis thomsonii*	8	0
Ancistrorhynchus metteniae	6	0	*Angraecopsis holochila*	2	2
Angraecopsis holochila	2	2	*Angraecopsis parviflora*	9	7
Angraecopsis parviflora	9	7	*Ansellia africana*	51	12
Arthropteris monocarpa	9	9	*Bulbophyllum josephi*	39	11
Asplenium aethiopicum	87	73	*Bulbophyllum lupulinum*	6	0
Asplenium hypomelas	9	0	*Bulbophyllum maximum*	39	17
Asplenium sandersonii	87	5	*Bulbophyllum sandersoni*	5	5
Asplenium theciferum	182	87	*Calyptrochilum christyanum*	67	12
Bolusiella iridifolia	55	0	*Canarina eminii*	1	1
Bulbophyllum lupulinum	6	0	*Corymboriks corymbis*	8	0
Canarina eminii	1	1	*Culcasia falcifolia*	31	12
Corymboriks corymbis	8	0	*Cyrtorchis arcuata*	67	11
Culcasia falcifolia	31	12	*Diaphananthe adoxa*	10	0
Diaphananthe adoxa	10	0	*Diaphananthe fragrantissima*	5	4
Diaphananthe rohrii	56	0	*Huperzia dacrydioides*	2	2
Diaphananthe tenuicalcar	83	14	*Polystachya bennettiana*	1	16
Drynaria volkensii	92	67	*Polystachya cultriformis*	34	0
Elaphoglossum acrostidoides	117	0	*Polystachya eurychila*	4	4
Elaphoglossum deckeni	32	0	*Polystachya fusiformis*	4	0
Elaphoglossum lastii	23	0	*Polystachya lindblomii*	21	9
Huperzia dacrydioides	2	2	*Polystachya steudneri*	4	4
Lepisorus excavatus	2	2	*Polystachya tessellata*	3	0
Liparis abyssinica	1	0	*Tridactyle bicaudata*	5	0
Loxogramme abyssinica	11	0		17.3±8.1	5.2± 2.2
Peperomia rotundifolia	178	101	**Herbaceous leaf**		
Peperomia abyssinica	101	41	*Ansellia africana*	51	12
Peperomia tetraphylla	234	127	*Arthropteris monocarpa*	9	9
Pleopeltis macrocarpa	78	54	*Arthropteris orientalis*	77	25
Polystachya cultriformis	34	0	*Asplenium aethiopicum*	87	73
Polystachya eurychila	4	4	*Asplenium hypomelas*	9	0
Polystachya fusiformis	4	0	*Asplenium sandersonii*	87	0
Polystachya lindblomii	21	9	*Asplenium theciferum*	182	87
Polystachya tessellata	3	0	*Bolusiella iridifolia*	55	0
Pyrrosia schimperiana	4	0	*Bulbophyllum josephi*	39	11
Thelypteris confluens	10	0	*Bulbophyllum lupulinum*	6	0
Tridactyle bicaudata	5	0	*Canarina eminii*	1	1
Vittaria volkensii	3	0	*Corymboriks corymbis*	8	0

	41.4±17.9	15.8±10.1	*Diaphananthe adoxa*	10	0
Wide-habitat species			*Diaphananthe rohrii*	56	0
Ansellia africana	51	12	*Diaphananthe tenuicalcar*	83	14
Arthropteris orientalis	77	25	*Drynaria volkensii*	92	67
Bulbophyllum josephi	39	11	*Elaphoglossum acrostidoides*	117	0
Bulbophyllum maximum	39	17	*Elaphoglossum lastii*	23	0
Bulbophyllum sandersoni	5	5	*Lepisorus excavatus*	2	2
Diaphananthe fragrantissima	5	4	*Loxogramme abyssinica*	11	0
Microcoelia globulosa	91	21	*Pleopeltis macrocarpa*	78	54
Polystachya bennettiana	1	16	*Polystachya bennettiana*	1	16
Polystachya steudneri	4	4	*Polystachya cultriformis*	34	0
	34.7±22.0	12.8±4.9	*Polystachya eurychila*	4	4
Leathery leaves and Succulent stem			*Polystachya fusiformis*	4	0
Aerangis brachycarpa	7	0	*Polystachya lindblomii*	21	9
Aerangis thomsonii	8	0	*Polystachya steudneri*	4	4
Ancistrorhynchus metteniae	6	0	*Polystachya tessellata*	3	0
Angraecopsis parviflora	9	7	*Pyrrosia schimperiana*	4	0
Bulbophyllum maximum	39	17	*Thelypteris confluens*	10	0
Bulbophyllum sandersoni	5	5		38.9±15.9	12.9 ±8.6
Calyptrochilum christyanum	67	12			
Cyrtorchis arcuata	67	11			
Diaphananthe fragrantissima	5	4			
Huperzia dacrydioides	2	2			
Liparis abyssinica	1	0			
Microcoelia globulosa	91	21			
Peperomia abyssinica	101	41			
Peperomia rotundifolia	178	101			
Peperomia tetraphylla	234	127			
Polystachya tessellata	3	0			
Tridactyle bicaudata	5	0			
Vittaria volkensii	3	0			
	46.2±31.2	19.3±16.8			
Herbaceous stem					
Ancistrorhynchus metteniae	6	0			
Arthropteris monocarpa	9	9			
Arthropteris orientalis	77	25			
Asplenium aethiopicum	87	73			
Asplenium hypomelas	9	0			
Asplenium sandersonii	87	0			
Asplenium theciferum	182	87			
Bolusiella iridifolia	55	0			
Diaphananthe rohrii	56	0			

Diaphananthe tenuicalcar	83	14
Drynaria volkensii	92	67
Elaphoglossum acrostidoides	117	0
Elaphoglossum deckeni	32	0
Elaphoglossum lastii	23	0
Huperzia dacrydioides	2	2
Lepisorus excavatus	2	2
Liparis abyssinica	1	0
Loxogramme abyssinica	11	0
Microcoelia globulosa	91	21
Pyrrosia schimperiana	4	0
Thelypteris confluens	10	0
Vittaria volkensii	3	0
	47.2±20.5	13.6±11.0

5.16 Density of geophytic fern under each functional groups (Forest-habitat species, Forest-margin species, Spaced frond species, Tufted frond species, erect and creeping rhizome species. (Mean±SE)

Forest-habitat species	Interior	Edge	**Spaced frond type**		
Asplenium formosum	63	21	*Adiantum lunulatum*	37	21
Asplenium gemmiferum	137	15	*Arthropteris monocarpa*	33	14
Asplenium hypomelas	87	54	*Arthropteris orientalis*	58	81
Asplenium lunulatum	70	59	*Athyrium filix-femia*	61	44
Asplenium monanthes	98	47	*Blotiella glabra*	120	64
Asplenium sandersoni	61	37	*Cheilanthes farinosa*	38	3
Asplenium theciferum	72	61	*Conigramme africana*	63	10
Blotiella glabra	120	64	*Dicranopteris linearis*	41	25
Conigramme africana	63	10	*Doryopteris concolor*	46	32
Drynaria volkensii	115	79	*Hypolepis sparsisora*	60	14
Dryopteris athamantica	42	38	*Microlepia speluncae*	50	34
Loxogramma abyssinica	21	0	*Pleopeltis macrocarpa*	25	40
Marsilia minuta	42	26	*Pteridium aquilinus*	0	115
Microlepia speluncae	50	34	*Thelypteris confluens*	26	0
Pleopeltis macrocarpa	25	40	**Mean(±SE)**	47±14.3	35.5±16.9
Marattia fraxinea	39	46	**Tufted frond type**		
Thelypteris confluens	26	0	*Asplenium anisophyllum*	70	22
Mean (±SE)	66.5±16.5	37.1±10.9	*Asplenium erectum*	102	0
Forest- margin			*Asplenium formosum*	63	0
Adiantum lunulatum	37	21	*Asplenium gemmiferum*	137	15
Arthropteris orientalis	58	81	*Asplenium hypomelas*	87	54
Athyrium filix-femia	61	44	*Asplenium lunulatum*	70	59
Cheilanthes farinosa	38	3	*Asplenium monanthes*	98	47
Dicranopteris linearis	41	25	*Asplenium sandersoni*	61	37
Doryopteris concolor	46	32	*Asplenium theciferum*	72	61

Hypolepis sparsisora	60	14	*Drynaria volkensii*	115	79
Pteridium aquilinus	0	115	*Dryopteris athamantica*	42	38
Mean (±SE)	42.6±23.1	41.9±26.2	*Loxogramma abyssinica*	21	0
Erect Rhizome type			*Marattia fraxinea*	39	46
Adiantum lunulatum	37	21	**Mean (±SE)**	75.2±17.6	35.2±14.1
Arthropteris orientalis	58	81	**Creeping Rhizome type**		
Asplenium erectum	102	0	*Arthropteris monocarpa*	33	14
Asplenium formosum	63	0	*Blotiella glabra*	120	64
Asplenium gemmiferum	137	15	*Dicranopteris linearis*	41	25
Asplenium hypomelas	87	54	*Doryopteris concolor*	46	32
Asplenium lunulatum	70	59	*Drynaria volkensii*	115	79
Asplenium monanthes	98	47	*Dryopteris athamantica*	42	38
Asplenium sandersoni	61	37	*Marattia fraxinea*	39	46
Asplenium theciferum	72	61	*Marsilia minuta*	42	26
Athyrium filix-femia	61	44	*Microlepia speluncae*	50	34
Cheilanthes farinosa	38	3	*Pleopeltis macrocarpa*	25	40
Conigramme africana	63	10	*Thelypteris confluens*	26	0
Hypolepis sparsisora	60	14	**Mean (±SE)**	52.6±19.5	36.2±12.9
Loxogramma abyssinica	21	0			
Pteridium aquilinus	0	115			
Mean (±SE)	64.3±15.9	35.1±16.4			

Appendix 5.17 Mean score of species tree functional groups (forest-habitat, wide-habitat and forest-margin species) on CCA axis one and axis two of the environmental variables (axis 1 = altitude, rainfall, edge disturbance and patch shape, axis 2= edge density and interior disturbance)

Forest-habitat species	Axis 1	Axis 2		Axis 1	Axis 2
Canthium oligocaprpa	0.12	1.50	**Widely-distributed species**		
Cassipurea malosana	0.04	0.43	*Allophylus abyssinicus*	0.46	0.56
Chionanthus mildbraedii	0.51	0.15	*Bersama abyssinica*	0.11	0.07
Dracaena fragrans	0.60	0.69	*Croton macrostachyus*	0.15	0.13
Galinera saxifraga	0.32	0.88	*Pouteria adolfi-friederici*	1.54	0.85
Oxyanthus speciosus	0.21	0.72	*Syzygium guineense*	0.60	0.38
Vepris dainelli	1.89	0.39	**Mean (±SE)**	0.57±0.21	0.55±0.19
Olea welw	0.45	3.00			
Mean (±SE)	0.47±0.19	0.91±0.29			
Forest-margin species	Axis 1	Axis 2			
Albizia gummifera	0.39	0.62			
Albizia schimperiana	0.01	1.44			
Maesa lanceolata	0.65	0.62			
Milletia ferrugina	0.82	0.23			
Nuxia congesta	1.35	1.20			
Rhus quartinana	1.01	0.00			
Mean (±SE)	0.70±0.09	0.69±0.23			

Appendix 5.18 Plant species list of edge and interior of 18 forest patches

Acanthaceae
Thunbergia alata Boj. ex. Sims
Asystasia gangetica (L.)T. Anders
Brillantaisis madagascariensis Lindau
Dicliptera maculata Nees
Dicliptera verticillatta (Forssk.) C. Christens
Hypoestes forskaolii (Vahl.) Roem. & Schult
Hypoestes triflora (Roem. Et Schult
Isoglossa laxa Oliv.
Justcia sp
Justicia betonica Linn
Justicia heterocarpa T. Anderson
Justicia striata (Klotzsch) Bullock
Mellera lobulata S. Moore
Phaulopsis imbricata (Forssk.) Sweet
Acanthopale pubescens (Lindau) C.B.Cl.
Acanthus eminens C.B.Clarke
Justcia declipteriodes Lindau
Justcia schimperiana
Adiantaceae
Cheilanthes farinosa (Forssk.) Kaulf
Amaranthaceae
Sericostachys scandens Gilg & Lopr
Achyranthes aspera L.
Celosia trigyna L
Cyathula polycephala Baker
Gomphrena celosiodesMart.
Cyathula uncinulata (Schrad.) Schinz
Amaryllidaceae
Scadoxus multiflorus (Martyn) Raf.
Anacardiaceae
Rhus ruspolii Engl.
Rhus glutinosa A. Rich
Rhus quartiniana A.Rich
Rhus vulgare Meikle
Apiaceae
Apium sp
Centella asiatica L.
Hydrocotyle mannii Hook. F.
Oenanthe palustris (Chiov.) C. Norman
Pimpinella sp.

Hypericum peplidifolium A. Rch
Hypericum quartinianum A. Rich.
Icacinaceae
Apodytes dimidata, Arn.
Lamaripsidaceae
Elaphoglossum acrostidoides (Hook. & Grev.) Schelpe
Elaphoglossum deckeni (Huhn) C. Chr
Elaphoglossum lastii (Baker) C. Chr
Lamiaceae
Achyrospermum schimperi (Hochst. Ex Briq) Perkins ex Mildbr
Ajuga integrifolia Buch.-Ham
Plectranthus punctatus L' Herit
Plectranthus silvestris Gurke
Sanicula elata Buch-Ham
Satrureja paradoxa Vatke
Satureia abyssinica (Benth.) Briq.
Leonotis ocymifolia (Burm.f.) Iwarsson
Ocimum lamiifolium (Hochst. ex Bent.) DC.
Ocimum suave Willd.
Plectranthus assurgens (Baker) J.K. Morton
Premna schimperi Engl.
Pycnostachys abyssinica Fresen.
Pycnostachys eminii Gürke
Lauraceae
Cassytha filiformis L.
Ocotea keynensis
Liliaceae
Simlax aspera L
Sparmannia ricinocarpa
Lycopodiaceae
Huperzia dacrydioides (Baker) Pic. Serm
Malvaceae
Sida collina Schlechtend
Sida ternata L.f
Wissadula rostrata (Schumach. & Thonn.) Hook. F
Abutilon cecillii
Abutilon longicuspe
Hibiscus dongolensis
Hibiscus ludwigii Eckl. & Zeyh
Hibiscus micranthus L.f.
Malva verticillata

Torilis arvensis (Huds.) Link	*Pavonia urens Cav.*
Apocynaceae	*Grewia ferruginea Hochst. ex A. Rich.*
Landolphia buchananii (Hall. F.) Stapf	**Maranthaceae**
Oncinotis tenuiloba Stapf	*Maranthochloa leucantha (A. Rich) Munro*
Aquifoliaceae	*Marattia fraxinea Sm*
Ilex mitis (L.) Radlk.	**Marsileaceae**
Araceae	*Marsilea minuta L.*
Culcasia falcifolia Engl.	**Meliaceae**
Amorphophallus abyssinicus (A. Rich.) N. E. Br.	*Eckebergia capensis Saprrm.*
Amorphophallus gallaensis (Engl.) N. E. Br.	*Lepidotrichilia volkensii (Gürke) Leroy*
Arisaema flavum (Forssk.) Schott	*Trichilia dregeana Sond*
Arisaema schimperanum Schott	*Trichilia emitica*
Araliaceae	**Melianthaceae**
Polyscias fulva (Heirn) Harms	*Bersama abyssinica Fresen.*
Schefflera abyssinica (Hochst ex A. Rich) Harms	*Cissampelos pareira L-Abuta*
Schefflera volkensii (Engl.) Harms	*Cissampleos torulosa E. Mey. ex Harv*
Phoenix reclinata Jacq.	*Stephania abyssinica (Dillion ex A. Rich.) Walp.*
Asclepiadaceae	*Tiliachora troupinii Cufod*
Blyttia fruticulosum Decne	**Moraceae**
Ceropegia cufodontis Chiov	*Dorestenia soerensenii*
Pentarrhinum abyssinicum	*Ficus capreaefolia Del.*
Pentarrhinum inspidum E. Mey	*Antiaris toxicara Lesch*
Pergularia daemia (Forssk.) Chiov subsp. daemia	*Ficus exasperata Vahl*
Periploca linearifolia Quert.-Dill. & A. Rich.	*Ficus lutea Vahl*
Secamone parvifolia (Oliv.) Bullock	*Ficus mucuso Ficalho*
Secamone punctulata Decne	*Ficus ovata Vahl.*
Gomphocarpus fruticosus (L.)	*Ficus palmata Forsk*
Asparagaceae	*Ficus sur Forssk.*
Asparagus africana Liana	*Ficus sycomorus L.*
Asparagus officinalis L.	*Ficus thonningii Blume*
Aspidiaceae	*Ficus vallis-choudae Del.*
Tectaria gemmifera (Fee) Alston	*Ficus vasta Forssk.*
Aspleniaceae	*Trilepisium madagascariense DC*
Asplenium abyssinicum Fee	**Musaceae**
Asplenium aethiopicum (Burm.f.) Bech	*Ensete ventricosum Maurelii*
Asplenium anisophyllum Kunze	**Myrsinaceae**
Asplenium erectum (Bory ex Willd	*Embelia schimperi Vatke*
Asplenium formosum Wild	*Myrsine africana L.*
Asplenium gemmiferum Schrad	*Maesa lanceolata Forssk.*
Asplenium hypomelas Kuhn	**Myrtaceae**
Asplenium lunulatum Sw	*Eucalyptus camaldulensis Dehnh*
Asplenium monanthes L.	*Eucalyptus globulus Labill*

Asplenium sandersoni H.K.

Asplenium theciferum (Humb., Bonpl. & Kunth

Asteraceae

Mikaniopsis clematoides (Rch. Bip ex . A. Rich.) Mile-Redth

Acmella caulirhiza Del.

Ageratum conyzoides L.

Anthemis tigreensis (J. Gay ex A. Rich

Bidens pilosa L

Carduus nyassanus (S. Moore) R. E. Fr.

Cirsium dender Friis

Crassocephalum montuosum (S. Moore) Milne-Redh

Crepis rueppellii Sch. Bip

Emilia abyssinica (Sch.Bip. Ex A.Rich) C. Jeffrey

Galinsoga parviflora Cav

Galinsoga qaudriradiata Ruiz & Pav

Gutenbergia rueppellii Sch. Bip

Lactuca inermis Forssk.

Spilanthes costata Benth

Taraxcum sp

Aspilia africana (Pers.) C. D. Adams

Aspilia mossambicensis (Oliv.) Wild

Bothriocline schimperi Oliv. & Hiern ex Benth.

Helichrysum splendidum (Thunb.) Less

Laggera pterodonta (DC.) Sch, Bip. Ex Oli

Microglossa pyrifolia (Lam.) Kuntze

Solanecio gigas(Vatke) C.Jeffrey

Solanecio mannii (Hook.f.) C.Jeffrey

Vernonia adoensis Sch. Bip. Ex Walp.

Vernonia auriculifera Hiern.

Vernonia biafrae Oliv. & Hiern

Vernonia dalettiensis

Vernonia glabra (Steetz) Vatke

Vernonia hochstetteri Sch-Bip

Vernonia myriantha Hook. f.

Vernonia ruepellii Sch. Bip. Ex Walp.

Vernonia thomsoniana Oliv. & Hiern

Vernonia turbinata

Vernonia urticifolia A. Rich.

Vernonia amygdalina Del

Bidens ghedoensis Mesfin

Baddlejaceae

Eugenia bukobensis Engl.

Syzygium guineense (Willd.) DC. ssp. afromontanum F. White

Nyctaginaceae

Boehavia diffusa

Oleaceae

Jasminum abyssinicum Hochst. ex DC.

Chionanthus mildbraedii (Gilg &Scellenb.) Stearn.

Olea capensis subsp macrocarpa

Olea welwitschii (Knobl.) Gilg & Schellenb

Schrebera alata (Hochst.) Welw.

Oleandraceae

Arthropteris orientalis (J.F. Gmel.) Posth

Adiantum lunulatum (Burm.f.

Opiliaceae

Opilia campestris Engl.

Orchidaceae

Aerangis brachycarpa A. Rich

Aerangis thomsonii (Rolfe) Schltr.

Ancistrorhynchus metteniae (Kraenzl.) Summerh

Angraecopsis holochila Summerh

Angraecopsis parviflora (Thouars) Schltr

Ansellia africana Lindl.

Arthropteris monocarpa (Cordem.) C. Chr

Bulbophyllum josephi (Kuntze) Summerh

Bulbophyllum lupulinum Lindl

Bulbophyllum maximum (Lindl.) Richb.f.

Bulbophyllum sandersoni (Hook.f.

Calyptrochilum christyanum (Rchb.f.) Summerh

Corymborkis corymbis Thouars

Cyrtorchis arcuata (Lindl) Schltr

Diaphananthe adoxa F.N.Rasm

Diaphananthe fragrantissima (Rchb.f.) Schltr

Diaphananthe rohrii (Rchb.f.) Summerh

Diaphananthe tenuicalcar Summerh

Liparis abyssinica A. Rich

Microcoelia globulosa (Hochst.) L.

Polystachya bennettiana (Rchb. F.

Polystachya cultriformis (Thouars) Spreng

Polystachya eurychila Summerh

Polystachya fusiformis (Thouars) Lindl.

Polystachya lindblomii Schltr

Nuxia congesta R.Br. ex Fresen.

Nuxia opositfolia

Balsaminaceae

Impatiens hochstetteri Warb

Bignoniaceae

Stereospermum kunthianum Cham

Boraginaceae

Cordia africana Lam.

Ehertia cymosa Thonn.

Brassicaceae

Cardamine africana L

Cardamine parviflora L

Cardamine trichocarpa Hochst. Ex A. Rich

Buddlejiaceae

Buddleja polystachya Fresen

Campanulaceae

Canarina abyssinica Engl.

Canarina eminii Asch. Ex Schweinf

Lobelia gibberoa Hemsl.

Cannaceae

Canna indica L

Capparidaceae

Gynandropsis gynandra

Ritchiea albersii Gilg

Caryophylaceae

Drymaria cordata (Linn) Willd

Celasteraceae

Hippocratea africana (Wild) Loes

Hippocratea goetzei Loes

Hippocratea pallens Plancho ex Oliver

Carissa spinarium Linn

Maytenus gracilipes (Welw. ex Oliv.) Exell

Maytenus obscura (A Rich) Cuf

Elaeodendron buchananii (Loes) Loes

Maytenus adat (Loes.) Sebsebe

Maytenus arbutifolia (A. Rich.) Wilczek

Maytenus undata (Thunb.) Blakelock

Combretaceae

Combretum collinum fresen

Combretum paniculatum Vent.

Commelinaceae

Aneilema beniniense (P. Beauv) Kunth

Commeliana diffusa

Commelina foliacea Chiov

Polystachya steudneri Reichb.f.

Polystachya tessellata (Jacq.) Garay & H.R.Sweet

Tridactyle bicaudata (Lindl.) Schltr

Eulophia guineensis

Eulophia guineensis Lindl

Habenaria cornuta

Habenaria cornuta Lindl.

Habenaria humilior Rchb.f

Habenaria schimperiana A. Rich

Nervilia bicarinata (Bl.) Schltr.

Oeceoclades saundersiana Rchb. F.Garay & P. Taylor

Polystachya rivae Schweinf.

Oxalidaceae

Oxalis corniculata L.

Oxalis latifolia H. B. K.

Oxalis trifolia

Phytolaccaceae

Hilleria latifolia (Lam.) H. Walter

Phytolacca dodecandra L´Herit.

Piperaceae

Peperomia rotundifolia (L.) Kunth

Peperomia abyssinica Miq

Peperomia tetraphylla (G. Frost) Hook. & Arn

Peperomia molleri C. DC.

Piper umbellatum Linn

Piper capense L.f. var. capense

Pittosporaceae

Pittosporum viridiflorum Sim.

Plantaginaceae

Plantago lanceolata L

Podocarpaceae

Podocarpus falcatus (Thunb.) Mirb

Polygalaceae

Polygala sp

Polygonaceae

Persicaria setosula (A. Rich.) K. L. Wilson

Rumex abyssinicus Jacq

Rumex nepalensis Spreng

Polypodiaceae

Lepisorus excavatus (Willd) Ching

Pyrrosia schimperiana (Kuhn) Alston

Conigramme africana

Drynaria volkensii Hieron

Loxogramme abyssinica (Baker) M. G. Price

Commelina latifolia (Hochest. Ex A. Rich

Cynotis barbata D. Don

Pollia condensata C. B. CI.

Pollia mannii C. B. CI.

Convolvulaceae

Ipomoea abyssinica

Ipomoea cairica (L.) Sweet

Ipomoea hochstetteri House

Ipomoea obscura (L.) Ker-Gawl.

Ipomoea tenuirostris Steud. Ex Choisy

Costaceae

Costus afer Ker-Gawl

Crassulaceae

Crassula alsinoides (Hook. F.) Engl

Kalanchoe densiflora Rolfe

Kalanchoe sp

Kalanchoe sp

Cucurbitaceae

Kedrostis foetidissima (Jacq.) Cogn.

Lagenaria abyssinica (Hook.f.) C.Jeffrey

Momordica foetida Schumach

Peponium vogelii (Hook.f.) Engle.

Sicyos polyacanthus Cong.

Zeheneria scabra (Linn. f.) Sond.

Cupressaceae

Cupressus lusitanica Miller

Cyatheaceae

Cyathea manniana

Cyperaceae

Carex sp

Cyperus esculenta

Kyllinga sp

Dennstaedtiaceae

Blotiella glabra (Bory) R. M. Tryon

Hypolepis sparsisora (Schrad.) Kuhn

Microlepia speluncae (L.) Moore

Pteridium aquilinum (L.) Kuhn

Dichondraceae

Dichondra repens (Forst. & Forst.

Dioscoreaceae

Dioscorea bulbifera L.

Tacca leontopetaloides (L.) Kuntze

Dracaenaceae

Dracaena afromontana Mildbr.

Pleopeltis macrocarpa (Willd.) Kaulf

Pteridaceae

Doryopteris concolor (Langsd. & Fisch.) Kuhn

Ranunculaceae

Clematis hirsuta Perr. & Guill.

Clematis longicauda Steud

Clematis simensis Fres.

Ranunculus multifidus Forssk.

Rhamnaceae

Gouania longispicata Engl

Helinus mystacinus (Ait.) E. Mey. ex Steud.

Scutia myrtina (Burm. f.) Kurz

Rhamnus prinoides L`Herit.

Rhamnus staddo A. Rich.

Rhizophoraceae

Cassipourea malosana (Bak.) Alston

Rosaceae

Alchemilla cryptantha A. Rich

Alchemilla rothii Oliv.

Rubus apetalus Poir.

Rubus steudneri Schweinf.

Prunus africana (Hook.f.) Kalkm.

Rubiaceae

Keetia gueinzii (Sond.) Bridson

Geophila repens (Linn) I.M. Johnston

Crossopteryx febrifuga (Afz. Ex G. Don) Benth

Oxyanthus lepidus

Pavetta abyssinica Fresen

Pavetta oliveriana Hiern

Pentas lanceolata Forssk

Rytigynia neglecta (Hiern) Robyns

Breonadia salicina (Vahl) Hepper &Wood

Canthium oligocarpum Hiern

Canthium sp

Coffea arabica L.

Galiniera saxifraga (Hochst.) Bridson

Gardenia ternifolia Schumach & Thonn

Oxyanthus speciosus sbsp. Stenocarpus (K.Schum.) Bridson

Psychotria orophila Petit

Psydrax schimperiana (A. Rich.) Bridson

Rothmannia urcelliformis (Hiern) Robyns

Sarcocephalus latifolius (Sm.) Bruce.

Vangueria apiculata K. Schum

Rutaceae

Dracaena fragrans (L) Ker-Gawl

Dracaena steudneri Engl

Drynariaceae

Drynaria volkensii Hieron

Dryopteridaceae

Athyrium filix-femia

Dryopteris athamantica (Kunze) Kunze

Ebenaceae

Diospyros abyssinica (Hiern) F. White

Euphorbiaceae

Euphorbia depauperata Hochest. Ex A. Rich

Euphorbia schimperiana Scheele

Phyllanthus sp

Tragia brevipes Pax

Acalypha acrogyna Pax

Acalypha marissima M.G. Gilbert

Acalypha ornate Hochst. Ex A. Rich

Acalypha psilostachya Hochst. Ex A. Rich

Acalypha racemosa Baill

Clutia abyssinica Jaub. & Spach.

Erythrococcaa trichogyne (Muell Arg) Prain (det A Radcliffe-Smith)

Phyllanthus limmuensis Cuf

Phyllanthus ovalifolius Forssk

Argomuellera macrophylla Pax

Bridelia micrantha (Hochs) Baill

Croton macrostachyus Del.

Euphorbia acandilabrium

Euphorbia ampliphylla Pax

Macaranga capensis var kilimandscharica (Pax) Friis & Gilbert

Sapium ellipticum (Hochst) Pax.

Fabaceae

Dolichos sericeus E. Mey

Lablab purpureus (L) Sweet

Clitoria ternata L.

Desmodium hirtum Guill. & Perr

Desmodium repandum (Vahl) DC

Thalictrum rhynchocarpum

Vermifrux abyssinica (A. Rich) Gillett

Calpurnia aurea (Ait.) Benth.

Crotalaria emarginella

Kotschya sp

Pterolobium stellatum (Forssk.) Brenan

Acacia abyssinica

Clausena anisata (Willd.) Benth.

Fagaropsis angolensis (Engl.) Dale

Teclea nobilis Del.

Vepris dainelli (Pichi-Sermolli) Kokwaro

Sapindaceae

Paullinia pinnata (L)

Dododaea angustifolia

Allophyllus abyssinicus (Hochst.) Radlkofer

Allophylus macrobotrys (Gilg)

Blighia unijugata

Pappea capensis Eckl. & Zeyh.

Sapotaceae

Mimusops kummel Bruce ex A.DC.

Pouteria adolfi-friederici

Simaroubaceae

Brucea antidysenterica J.F. Mill.

Solanaceae

Physalis peruviana Linn

Solanium nigrum

Discopodium penninervium Hochst.

Solanium anguivi Lam

Solanium schimperiana

Solaniun indicum L

Stericuliaceae

Dombya torrida (J.F. Gmel.) P. Bamps

Thelypteridaceae

Thelypteris confluens (Thunb.) C.V. Morton

Tiliaceae

Corchorus trilocularis L.

Triumfetta annua L.

Triumfetta rhomboidea (Trimfetta Buee)

Ulmaceae

Celtis africana Burm. F.

Celtis toka (Forssk.) Hepper & wood

Celts zenkeri Endl.

Trema orientalis (L) Blume

Urticaceae

Urerea hypselodendron (A. Rich.) Wedd.

Urerea trinervis (Hochst. Ex Krauss) Friis & immelman

Droguetia iners Schweinf.

Pilea rivularis Wedd

Pilea tetraphylla Steud. Blume

Urtica simensis steudel

Albizia grandibracteata. T	**Verbenaceae**
Albizia gummifera (J.F.Gmel.) C.A.Sm.	*Clerodendron myricoides (Hochst.) R. Br.ex Vatke.*
Albizia schimperiana Oliv.	*Lantana trifolia L*
Dalbergia lactea Vatke	*Lippia adoensis Hochst. ex Walp.*
Entada abyssinica Steud. ex A. Rich.	**Vitaceae**
Milletia ferruginea (Hochst) Baker	*Cissus petiolata Hook. f.*
Senna petersiana (Bolle) Lock	*Cissus quadrangularis Linn*
Flacourtaceae	*Cyphostemma cyphoetalum (Fresen.) Desc. ex Wild & R.B.*
Dovyalis abyssinica (A. Rich) Warb.	*Rhoicissus tridentata (L. f.) Wild & R.B. Drumm.Willd & Drummond*
Flacourtia indica (Burm.f.) Merr.	**Vittariaceae**
Geraniaceae	*Vittaria volkensii Heiron*
Geranium ocellatum Cambess.	**Zingibraceae**
Gleicheniaceae	*Aframomum corrorima (Braun) P.C.M*
Dicranopteris linearis (Burm. F.) Underw	*Aframomum zambesiacum K. Schum*
Guttiferae	**Zygophyllaceae**
Garcinia buchananii Baker	*Tribulus terrestris Linn.*
Hypericaceae	*Didymodoxa cafra (Thunb) Friis & Wilmot*

Appendix 6.1 Plant species list of six sacred groves.

Acanthaceae	*Leucas martinicensis R. Br.*
Acanthus eminens C.B.Clarke	*Lippia adoensis Hochst. ex Walp.*
Acanthopale pubescens (Lindau) C.B.Cl	*Ocimum lamiifolium* Hochst. ex Benth
Justicia schimperiana Hochst. Ex A. Rich	*Plectranthus assurgens (Baker) J.K. Morton*
Hypoestes forskaolii (Vahl.) Roem. & Schult	*Premna schimperi Engl.*
Justicia diclipteroides Lindau	*Pycnostachys abyssinica* Fresen.
Thunbergia alata Bojer ex Sims	*Pycnostachys eminii Gürke*
Alismataceae	*Satrureja paradoxa Vatke*
Alisma plantago-aquatica	**Lobelaceae**
Amarantaceae	*Lobelia gibberoa Hemsl*
Achyranthes aspera L.	**Malvaceae**
Apiaceae	*Abutilon longicuspe Hochst. Ex A. Rich*
Oenothera fruticosa A. Gray.	*Hibiscus dongolensis Delile*
Apocynaceae	*Pavonia urens Cav*
Carissa spinarum L.	*Hibiscus macranthus* Hochst A ex Rich
Aquifoliaceae	*Bersama abyssinica Fresen.*
Ilex mitis (L.) Radlk.	*Ekebergia capensis Saprrm.*
Araceae	*Lepidotrichilia volkensii (Gürke) Leroy*
Arisaema schimperiana Schott	**Moraceae**
Araliaceae	*Ficus palmata Will*
Schefflera abyssinica (Hochst ex A. Rich) Harms	*Ficus sur Forssk.*
Schefflera volkensii (Engl.) Harms	*Ficus thonningii Blume*
Arecaceae	*Ficus vasta Forssk.*

Phoenix reclinata Jacq.

Asclepiadiaceae

Pentarrhinum abyssinicum Decne

Pergularia daemia (Forssk.) Chiov.

Pentarrhinum abyssinicum

Pentarrhinum inspidum E. Mey

Asparagaceae

Asparagus racemosus Willd.

Aspleniaceae

Asplenium sandersonii Hook

Asteraceae

Bidens ghedoensis Mesfin

Bothriocline schimperi Oliv. & Hiern ex Benth.

Carduus nyassanus (S. Moore) R. E. Fr.

Anthemis tigreensis A. Rich

Aspilia africana (Pers.) C. Adams

Galinsoga parviflora Cav

Helichrysum splendidum (Thunb.) Less

Vernonia amygdalina Del

Vernonia auriculifera Hiern.

Vernonia biafrae Oliv. & Hiern

Vernonia hochstetteri Schultz Bip.

Balsaminaceae

Impatiens rothii Hook.f.

Impatiens hochstetteri Warb

Boraginaceae

Cordia africana Lam.

Buddlejaceae

Nuxia congesta R. Br. Ex Fresen.

Caparidaceae

Ritchiea albersii Gilg

Celastraceae

Hippocratea africana (Wild) Loes

Hippocratea goetzei Loes

Maytenus adat (Loes.) Sebsebe

Maytenus arbutifolia (A. Rich.) Wilczek

Maytenus gracilipes (Welw. ex Oliv.) Exell

Commelinaceae

Commelina diffusa Burm. f.

Convolvulaceae

Dichondra repens (J.R.Forst. & G.Frost

Crassulaceae

Myrsinaceae

Embelia schimperi Vatke

Maesa lanceolata Forssk.

Myrsine africana L.

Myrtaceae

Eucalyptus globulus Labill

Syzygium guineense (Willd.) DC. ssp. afromontanum F. White

Nyctaginaceae

Boerhavia diffusa

Oleaceae

Chionanthus mildbraedii (Gilg &Scellenb.) Stearn.

Jasminum abyssinicum Hochst. ex DC.

Olea capensis subsp macrocarpa (C.H.Wright) I. Verd

Olea welwitschii (Knobl.) Gilg & Schellenb

Schrebera alata (Hochst.) Welw.

Orchidaceae

Aerangis brachycarpa

Oxalidaceae

Oxalis corniculata L.

Phytolaccaceae

Phytolacca dodecandra L´Herit.

Piperaceae

Peperomia abyssinica Miq

Piper capense L.f.

Pittosporaceae

Pittosporum viridiflorum Sim.

Poaceae

Panicum hochstetteri Steud.

Hyparrhenia rufa (Nees) Stapf

Oplismenus compositus (L.) P. Beauv

Polygalaceae

Polygala steudneri Chordat

Rumex abyssinicus Jacq

Rumex nepalensis Spreng.

Drynaria volkensii Hieron

Ranunculaceae

Clematis hirsuta Perr. & Guill.

Clematis simensis Fres.

Ranunculus multifidus Forssk.

Rhamnaceae

Rhamnus prinoides L`Herit.

Rhamnus staddo A. Rich.

kalanchoe densiflora Dolfe
Cucurbitaceae
Lagenaria abyssinica (Hook.f.) C.Jeffrey
Momordica foetida Schumach.
Cupressaceae
Cupresus lucitanica
Cyperaceae
Cyperus esculenta
Dracaenaceae
Dracaena afromontana Mildbr.
Dracaena steudneri Engl.
Euphorbiaceae
Croton macrostachyus Del.
Euphorbia candelabrum Trem. Ex Kotschy
Euphorbia platyphyllos L.
Macaranga capensis var kilimandscharica (Pax) Friis & Gilbert
Tragia cinerea (Pax) M.G.Gilbert & Radcl.-Sm.
Acalypha racemosa Baill
Fabaceae
Acacia brevispica Harms
Albizia gummifera (J.F.Gmel.) C.A.Sm.
Albizia schimperiana Oliv.
Clitoria ternatea L.
Crotalaria emarginella Vatke
Desmodium repandum (Vahl) DC
Dolichos sericeus E. Mey.
Erythrina brucei Schweinf.
Millettia ferruginea (Hochst.) Bark.
Paracalyx somalorum (Vierh.) Ali
Vermifrux abyssinica (A. Rich.) J.B. Gillett
Flacourtaceae
Dovyalis abyssinica (A. Rich) Warb.
Flacourtia indica (Burm.f.) Merr.
Geraniaceae
Geranium ocellatum Cambess.
Icacinaceae
Apodytes dimidiata E. Mey. ex Arn.
Lamiaceae
Lantana trifolia L
Leonotis ocymifolia (Burm. f.) Iwarsson
Rosaceae
Alchemilla kiwuensis Angl.
Hagenia abyssinica
Prunus africana (Hook.f.) Kalkm.
Rubus steudneri Schweinf.
Rubiaceae
Canthium oligocarpum Hiern
Coffea arabica L.
Galiniera saxifraga (Hochst.) Bridson
Oxyanthus lepidus S. Moore
Pavetta abyssinica Fresen
Pentas lanceolata Forssk
Rothmannia urcelliformis (Schweinf. ex Hiern) Bullock ex Robyns
Rytigynia neglecta (Hiern) Robyns
Rutaceae
Clausena anisata (Willd.) Benth.
Vepris dainelli (Pichi-Sermolli) Kokwaro
Sapindaceae
Allophyllus abyssinicus (Hochst.) Radlkofer
Simaroubaceae
Brucea antidysenterica J.F. Mill.
Solanaceae
Discopodium penninervium Hochst.
Tiliaceae
Dombeya torrida (J.F.Gmel.) Bamps
Sparmannia ricinocarpa (Eckl. & Zeyh.) Kuntze
Ulmaceae
Celtis africana Burm. F.
Trema orientalis (L.) Blume
Urticaceae
Droguetia iners Schweinf.
Elatostema monticolum Hook.f.
Girardinia diversifolia (Link.) Friis
Pilea tetraphyla (Steud.) Blume
Urerea hypselodendron (A. Rich.) Wedd.
Urtica simensis Hochst. ex A.Rich
Verbenaceae
Clerodendron myricoides (Hochst.) R. Br.ex Vatke.
Vitaceae
Cyphostemma cyphoetalum (Fresen.) Desc. ex Wild & R.B.

Appendix 6.2 species list of four non-sacred forests

Acanthaceae

Acanthus eminens C.B.Clarke

Acanthopale pubescens (Lindau) C.B.Cl

Hypoestes forskaolii (Vahl.) Roem. & Schult

Hypoestes triflora (Roem. Et Schult

Justicia heterocarpa T. Anderson

Justicia striata (Klotzsch) Bullock

Thunbergia alata Bojer ex Sims

Adiantaceae

Cheilanthes farinosa (Forssk.) Kaulf

Alismataceae

Alisma plantago-aquatica

Amarantaceae

Achyranthes aspera L.

Cyathula uncinulata (Schrad.) Schinz

Anacardiaceae

Rhus glutinosa A. Rich

Aquifoliaceae

Ilex mitis (L.) Radlk.

Araceae

Arisaema schimperiana Schott

Araliaceae

Polyscias fulva (Heirn) Harms

Schefflera abyssinica (Hochst ex A. Rich) Harms

Schefflera volkensii (Engl.) Harms

Arecaceae

Phoenix reclinata Jacq.

Asclepiediaceae

Pentarrhinum abyssinicum

Pentarrhinum inspidum E. Mey

Asparagaceae

Asparagus racemosus Willd

Asparagus africanus Lam.

Aspleniaceae

Asplenium lunulatum Sw

Asplenium sandersoni H.K.

Asplenium erectum (Bory ex Willd

Asteraceae

Ageratum conyzoides L.

Acmella caulirhiza Del.

Bidens ghedoensis Mesfin

Bothriocline schimperi Oliv. & Hiern ex Benth.

Lobelaceae

Lobelia gibberoa Hemsl

Malvaceae

Abutilon longicuspe Hochst. Ex A. Rich

Hibiscus berberidifolius A. Rich

Melastomataceae

Dissotis canescens (Graham) Hook.f.

Meliaceae

Bersama abyssinica Fresen.

Ekebergia capensis Saprrm.

Lepidotrichilia volkensii (Gürke) Leroy

Menispermaceae

Stephania cyanantha (Welw. Ex Hiern

Moraceae

Dorestenia soerensenii

Ficus sur Forssk.

Myricaceae

Myrica salicifolia Hochst. Ex A. Rich

Embelia schimperi Vatke

Maesa lanceolata Forssk.

Myrsine africana L.

Myrtaceae

Syzygium guineense (Willd.) DC. ssp. afromontanum F. White

Nyctaginaceae

Boerhavia diffusa

Oleaceae

Chionanthus mildbraedii (Gilg &Scellenb.) Stearn.

Jasminum abyssinicum Hochst. ex DC.

Jasminum grandiflorum L.

Olea capensis subsp macrocarpa (C.H.Wright) I. Verd

Schrebera alata (Hochst.) Welw.

Oleandraceae

Arthropteris monocarpa (Cordem.) C.Chr

Oliniaceae

Olinia rochetiana A. Jussieu

Onagraceae

Oenothera fruticosa A. Gray.

Orchidaceae

Epipactis africana Rendle

Phytolacaceae

Phytolacca dodecandra L´Herit.

Piperaceae

Carduus nyassanus (S. Moore) R. E. Fr.
Crepis rueppellii Sch. Bip
Haplocarpha schimperi (Sch.Bip.) Beaverd
Helichrysum splendidum (Thunb.) Less
Laggera pterodonta (DC.) Sch, Bip. Ex Oli
Solanecio mannii (Hook.f.) C. Jeffrey
Solanecio gigas(Vatke) C.Jeffrey
Vernonia amygdalina Del
Vernonia auriculifera Hiern.
Vernonia biafrae Oliv. & Hiern
Balsaminaceae
Impatiens rothii Hook.f.
Buddlajaceae
Nuxia congesta R. Br. Ex Fresen.
Caparidaceae
Ritchiea albersii Gilg
Celastraceae
Hippocratea africana (Wild) Loes
Maytenus adat (Loes.) Sebsebe
Maytenus arbutifolia (A. Rich.) Wilczek
Maytenus undata (Thunb.) Blakelock
Maytenus gracilipes (Welw. ex Oliv.) Exell
Commelinaceae
Commeliana diffusa
Cynotis barbata D. Don
Crassulaceae
Kalanchoe petitiana A. Rich
Cucurbitaceae
Kedrostis foetidissima (Jacq.) Cogn.
Cyperaceae
Cyperus esculenta
Dracaenaceae
Dracaena afromontana Mildbr.
Dracaena fragrans (L) Ker-Gawl
Ericaceae
Erica arborea L.
Euphorbiaceae
Acalypha racemosa Baill
Clutia abyssinica Jaub. & Spach.
Croton macrostachyus Del.
Euphorbia platyphyllos L.
Macaranga capensis var kilimandscharica (Pax) Friis & Gilbert
Fabaceae
Acacia brevispica Harms
Albizia schimperiana Oliv.
Crotalaria emarginella Vatke

Peperomia rotundifolia (L.) Kunth
Peperomia abyssinica Miq
Peperomia tetraphylla (G. Frost) Hook. & Arn
Pittosporaceae
Pittosporum viridiflorum Sim.
Poaceae
Harundinaria alpina (K. Schum)
Hyparrhenia rufa (Nees) Stapf
Oplismenus compositus (L.) P. Beauv
Polygalaceae
Polygala steudneri Chordat
Drynaria volkensii Hieron
Rananculaceae
Clematis hirsuta Perr. & Guill.
Clematis simensis Fres.
Rhamnaceae
Rhamnus prinoides L`Herit.
Rosaceae
Prunus africana (Hook.f.) Kalkm.
Rubus apetalus Poir.
Rubus steudneri Schweinf.
Rubiaceae
Canthium oligocarpum Hiern
Galiniera saxifraga (Hochst.) Bridson
Oxyanthus lepidus S. Moore
Pavetta abyssinica Fresen
Pentas lanceolata Forssk
Rytigynia neglecta (Hiern) Robyns
Rutaceae
Clausena anisata (Willd.) Benth.
Teclea nobilis Del.
Vepris dainelli (Pichi-Sermolli) Kokwaro
Salicaceae
Salix subserrata Willd
Sapindaceae
Allophyllus abyssinicus (Hochst.) Radlkofer
Sapotaceae
Pouteria adolfi-friederici (Engl.) Baehni
Simaroubaceae
Brucea antidysenterica J.F. Mill.
Solanaceae
Discopodium penninervium Hochst.
Physalis peruviana Linn
Tectariaceae
Tectaria gemmifera (Fee) Alston
Ulmaceae

Desmodium repandum (Vahl) DC
Flacourtiaceae
Dovyalis abyssinica (A. Rich) Warb.
Flacourtia indica (Burm.f.) Merr.
Geraniaceae
Geranium ocellatum Cambess.
Hypericaceae
Hypericum peplidifolium A. Rch
Hypericum quartinianum A. Rich.
Icacinaceae
Apodytes dimidiata E. Mey. ex Arn.
Lamiaceae
Plectranthus assurgens (Baker) J.K. Morton
Plectranthus caninus Roth
Pycnostachys eminii Gürke
Satrureja paradoxa Vatke
Trema orientalis (L.) Blume
Celtis africana Burm. F.
Urticaceae
Droguetia iners Schweinf.
Pilea rivularis Wedd
Elatostema monticolum Hook.f.
Pilea tetraphyla (Steud.) Blume
Urerea hypselodendron (A. Rich.) Wedd.
Verbenaceae
Clerodendron myricoides (Hochst.) R. Br.ex Vatke.
Lippia adoensis Hochst. ex Walp.
Vitaceae
Cissus petiolata Hook. f.
Cyphostemma cyphoetalum (Fresen.) Desc. ex Wild & R.B.

Appendix 6.3 Variance and Eigenvalues of a CCA of plant species in sacred and non-sacred forests

Axis	Eigenvalue	%
1	0.28163	35.59
2	0.24824	31.38
3	0.15132	19.13
4	0.11001	13.9

Appendix 6.4 Biocultural diversity value of sacred groves - questionnaries

I. Questionnaires for general sacred sites

1. What sacred sites mean by you or other local elders?__________________________________
____
____

2. What kind of sacred sites you have (common, clan, family et c.)?________________________
____
____

3. How many types of sacred sites in this area?_______________________________________
____
____
____

4. Are these sites that have split into good and bad spiritual energy?________________________
____
____

5. What is the physical location of sacred sites?_______________________________________
    a) Where are sacred sites generally situated_______________________________________
    b) Why are they situated there?____
    c) What are the explanations for selection of the sacred site location? (beauty and biodiversity, landscape, presence of stones, trees of unusual form or can you explain)____
    d) What is the purpose of the sacred sites?___________________________________
    e) Are there any prohibitions on visiting the sacred sites?_________________________

6. Where did you gain the knowledge?____
____

7. Is the knowledge specific to certain group of community members?____________________
____
____

8. How often you visit sacred sites? who visit them?___________________________________
____

II. Questionnaires focused on sacred groves

9. Why do these sacred groves became sacred? are they because of the biological aspect or religious beliefs of the people?
____
____

10. Does sacred groves carry a biological importance with respect to the biological diversity conservation?____
____
____

11. Are there any rituals connected to visiting the sacred groves? What kind.________________
____

12. What are the days of prayer in a year in or around sacred grove?____
____

13. Any connection to legends, and myths? what kind?_________________________________
____

14. Are there materials within the sacred forests that serves to differentiate it from ordinary world?____

____

15. Are these sacred groves eternally sacred or occasionally sacred?______________________

____

16. How sacred groves are categorized? (forest under the control of the traditional religious leader, forests of great sacred value for sacrificial and religious ceremonies, burial ground)____

____

17. What are traditional religious practices in sacred groves?___________________________

____

18. What are the traditions (taboos and beliefs) associated with sacred groves?_____________

____

19. What will happen if one ignore or transgress these traditions?_______________________

____

20. Are sacred groves are belongs to headman of the villages, or extended family or owned collectively by the villagers?____

____

21. What is the value system (social order) of the community made it possible to keep the sacred groves? ____

____

22. How access and management of the sacred groves is governed?______________________

____

23. What institutions have been or continue to regulate access and management of sacred groves and its resources?____

____

24. What are traditional conservation practices? How people feel in these sacred forest? Have the way people felt changed over the years?_______________________________________

____

25. What within a sacred forest is being valued? the trees? the land? the organisms in that forest? forest spirit? ____

____

26. Is there any symbolic significance attached to different species in sacred forest? what are the species?

____

____

27. Have belief systems led to the protection of the biodiversity in sacred forest?____________

____

____

28. Do you have a concept of species? Is it particular trees that are of value?______________

____

____

29. Have peoples attitude to sacred forest changed over the last 20 years? why have those changes occurred? ____

30. What general change have you observed to the peoples attitude toward sacred forest, example between young and old generations? _______________________________________

____

____

31. What are the threats to the sacred forests and the belief system?______________________

____

____

32. What is the status of this traditional religion at the moment?_________________________

____

____

33. How the knowledge have passed from elders to the youngsters? ______________________

____

____

34. How did land-use change influence the spiritual perception of sacred groves?___________

____

35. Have you ever made enrichment planting in sacred groves to restore the degraded areas from grazing?____

36. What kind of plant you prefer for planting in sacred groves?_________________________

____