Vegetation ecology and plant diversity of Hugumburda-Gratkhassu National Forest Priority Area, Northeastern Ethiopia: habitat fragmentation in time and space

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Abstract

Vegetation ecology and plant diversity of Hugumburda-Gratkhassu National Forest Priority Area, Northeastern Ethiopia: habitat fragmentation in time and space

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This study was conducted with the objectives of analyzing and understanding the species composition, diversity, structure and regeneration dynamics of Hugumburda-Gratkhassu NFPA in relation to land use\land cover and habitat fragmentation. A total of 77 quadrats were sampled using stratified sampling with optimum allocation method. In the quadrats, data on abundance, height and DBH of woody plant species as well as landscape variables were recorded. Within the main quadrat, five 2 m x 2 m subplots were laid to collect data on seedlings, saplings and herbaceous species. In addition, in each subplot, soil samples were collected and soil texture, electrical conductivity and pH were analyzed. In order to investigate edge effect, 17 forest patches were selected and sampling points for vegetation and environmental variables were established at 0, 50, 100, 150 and 200 m from the edge. A total of 83 quadrats were selected for this study. Furthermore, Satellite images were acquired and analyzed by using remote sensing technology and GIS mapping to generate information on the temporal and spatial changes in land use\land cover types and habitat fragmentation. A total of 326 plant species representing 238 genera and 88 families were recorded; shrub and herb species are dominant. Cluster analysis resulted in five different plant communities and this result was supported by the ordination result. CCA result showed altitude was the main environmental variable in determining the plant communities. Species richness, diversity and evenness varied among the plant communities and along the edge gradient. Analysis

of population structure and regeneration status of the two Forest remnants revealed various patterns and generally dominated by small sized individuals. In addition, the results of image analysis revealed that disturbance was the major driving force for fragmentation and patchiness of the remnant natural vegetation and thus exposing plant species to various environmental effects at edge habitats. Consequently, the forest edge habitats were found to possess lower species richness, diversity, tree density and basal area compared to the inner habitats. Therefore, to adequately conserve and ensure species survival and maintain species diversity in the study area, these forest remnants and edge habitats should be maintained and well managed.

Key words/phrases: dry afro-montane forest, edge effect, Hugumburda-Gratkhassu, habitat fragmentation, land use\land cover, plant community, National Forest Priority Area, species richness, vegetation ecology

DEDICATION

This thesis is dedicated to the inhabitants of Hugumburda and Gratkhassu area who maintained their rich indigenous ecological knowledge and conservation practices for generations despite the tremendous environmental and socio-political changes that took place over the years. Had not been to the conservation patronage of the local people, these forest remnants would have gone long time ago.

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LIST OF ACRONYMS

BA: Basal Area
CA: Class Area
CCA: Canonical Correspondence Analysis
CSA: Central Statistical Agency
DBH: Diameter at Breast Height
DEM: Digital Elevation Model
EC: Electrical Conductivity
EEPFE: Environmental Economics Policy Forum for Ethiopia
EFAP: Ethiopian Forestry Action Program
EMA: Ethiopian Mapping Agency
ENVI: Environment for Visualizing Images
EPA: Environmental Protection Authority
ETH: The National Herbarium of Ethiopia
EWCA: Ethiopian Wild Life Conservation Authority
EWNHS: Ethiopia Wildlife and Natural History Society
FAO: Food and Agriculture Organization
FEE: Flora of Ethiopia and Eritrea
FRAC: Fractal Dimension
FRAGSTATS: The Software used for Calculating Landscape Pattern Metrics
GIS: Geographical Information System
GPS: Global Positioning System
IBC: Institute of Biodiversity Conservation
IUCN: International Union for the Conservation of Nature and Natural Resources
LULC: Landuse and Landcover
MEA: Millennium Ecosystem Assessment
NBSAP: National Biodiversity Strategy and Action Plan
NFPA: National Forest Priority Areas
NMSA: National Meteorological Service Agency
NMDS Nonmetric Multi-Dimensional Scaling
NP: Number of Patches
SFCDD: State Forests Conservation and Development Department
TM: Thematic Mapper
WU: Welo Floristic Region
WCMC: World Conservation Monitoring Center

CHAPTER ONE

1. INTRODUCTION

1.1. Background

Biodiversity describes the diversity and variability within and between species, habitats and ecosystems. Wilson (1992) defined biodiversity as "The variety of organisms considered at all levels, from genetic variants belonging to the same species through arrays of species to arrays of genera, families and still higher taxonomic level; includes the variety of ecosystems, which comprise both the communities of organisms within particular habitats and the physical conditions under which they live". Apart from its intrinsic value as part of the world's natural heritage, biodiversity is important for many other reasons (Gardner *et al.,* 2009). In particular, it plays a vital role in maintaining life support systems and ecological processes. It also has economic, educational, cultural and aesthetic values. However, there is an alarm at the scale of loss of biodiversity around the world, and the impact this will have on the way ecosystems function and ultimately on the future health and well-being of the human population. For all these reasons, conservation of biodiversity needs to be at the forefront of any development activity.

Understanding diversity, distribution and extent of the importance of vegetation is a basis for designing and implementing a sound management and utilization systems in a sustainable manner (Han *et al*., 2011). Although natural forests play important roles in providing a number of services, they are fast declining due to excessive anthropogenic influences (Dhillion *et al.,* 2003). Worldwide, forests have been fragmented into small patches, and forest structure and species composition have been influenced due to

fragmentation, edge effect and habitat loss. Deforestation, habitat fragmentation and edge effects that arise due to the conversion of forests to other agricultural land-use types, and over-utilization of forest resources to satisfy the timber, food, medicine and energy requirement of the ever-increasing human population are mentioned as major challenges to global resources (FAO, 2010).

Land-use patterns and disturbance regimes are two of the dynamic agents that can have profound effects on the abundance, distribution and diversity of plants. The need for conservation and management of forest and natural resources requires human to influence the uses of land and the sizes and distributions of populations to maintain specific habitat conditions. Identification of the primary agents of change and their extent and magnitude are necessary to maintain forested ecosystems and their components. An understanding of the background rates and causes of change can help to guide conservation efforts on many scales (Codjoe, 2007). Therefore, a study on land use\land cover plays an important role in global environmental change and sustainability, including response to climate change, effects on ecosystem structure and function, species and genetic diversity, water and energy balance and agro-ecological potential.

Similarly, habitat fragmentation and edge effects are considered the major threats to plant diversity (Laurance *et al*., 2006), increasing isolation of habitats and endangering species of plants (Gardner *et al.,* 2009) and on a variety of population and community dynamic processes (Honnay *et al.,* 2010). The effect is more severe in tropical forests where diversity is high and forests are being removed and fragmented at an increasing rate (Laurance *et al.,* 2006). Forest degradation in Sub-Saharan Africa, for instance, has

widely taken place because people gain short-term economic benefits from the forest-related economic activities (Mogaka *et al.*, 2001). In the same way, accelerated deforestation and habitat fragmentation to satisfy the food and energy requirements of the increasing population are major environmental concerns in Ethiopia (Friis *et al.,* 2001).

Ethiopia is an important regional center for biological diversity due to its wide range of altitude, its great geographical diversity with high and rugged mountains, flat-topped plateaus and deep gorges, incised river valleys and rolling plains (Ensermu Kelbessa *et al.*, 1992; Zerihun Woldu, 1999). This diversity in relief makes the country unique in Africa, and helped the emergence of wide ranges of habitats that are suitable for the evolution and survival of various plant and animal species. As a result, the country is regarded as one of the most important countries in Africa with respect to endemism of plant and animal species (WCMC, 1992). Evolutionary history has also played key roles with regard to the extant plant diversity of Ethiopia (Adane Assefa *et al*., 2007; Mamo Kebede *et al*., 2007; Popp *et al*., 2008; Gehrke, 2011).

The flora of Ethiopia is very diverse and comprises about 6000 species of higher plants, of which about 10% are endemic (Vivero *et al*., 2006). However, the rich biodiversity of the country is under serious threat due to anthropogenic disturbances (IBC, 2009; Reusing, 2000). A substantial proportion (40%) of the Ethiopian highlands were once believed to have been covered by forests having wide coverage than at present, but have gradually been cleared (Friis, 1992; EFAP, 1998). By the early 1950s, high forests were reduced to 16% of the total land area, 8% in the 1960s, 4% in 1970s, by 1989 to about

2.7% and less than 2.3% in 1994 (IUCN, 1990; EFAP, 1994; EPA, 1997). According to recent estimates by Reusing (1998 and 2000), forest cover of Ethiopia was 1.41% from 1996 - 1997.

Tamrat Bekele (1993) reported the occurrence of isolated mature trees in farmlands and the patches of forests that are seen around church-yards and religious burial grounds indicating the presence of vast expanse of forests in the earlier time. At the moment, most of the remaining forests of the country are confined to south and south-western parts of the country. However, nowadays the remnant forests in these areas are threatened by human activities (Tamrat Bekele, 1994) and continuously degraded over years (Reusing, 2000).

Human activities related to fuelwood collection and agricultural expansion were frequently cited as the main causes of forest degradation, leading to loss of forest cover and biodiversity, erosion, desertification and reduced water resources (e.g., Zerihum Woldu, 1999; Ensermu Kelbessa and Teshome Soromessa, 2008; IBC, 2009). Deliberate fire and non-integrated investment activities were also reasons for the reduction of forests in Ethiopia (Demel Teketay, 2000). The high level of dependency of the local community on agriculture (more than 90%) and high rate of population growth (Sahlu Haile, 2004) have also accelerated the problems. Apparently, biodiversity resources along with their habitats are rapidly disappearing in many parts of the country (Tadesse Woldemariam, 2003; Demel Teketay, 2005; Feyera Senbeta, 2006; Feyera Senbeta and Denich, 2006).

Pollen and charcoal studies in northern Ethiopia have revealed that forest disturbance has a 3000-year history (Hurni, 1985; Darbyshire *et al*., 2003) showing that a long history of secondary vegetation and grasslands in northern Ethiopia and explaining its current

dominance in the same area. According to the Ministry of Agriculture (MoA, 2003), in 2003 the natural forest cover in Tigray was only 0.2% of the total land mass of the Region, indicating the severe forest degradation in the Region. The destruction of vegetation and environmental degradation has become issues of national and global concern in recent years. This is because of the fact that declining vegetation cover and depletion of natural resources are closely associated with drought and food shortages that have become major threat affecting the life of millions of people. The underlying causes for these problems originate from poverty, population growth, lack of alternative livelihoods, inadequate policy support, inappropriate investment and inadequacy of law enforcement.

Following the long history of land degradation, many land rehabilitation and conservation programs have been carried out in northeastern Ethiopia. A historical vegetation cover change study by Nyssen *et al*. (2009) indicated that the vegetation cover in northern Ethiopia has improved during the last century through land rehabilitation programs. Community woodlots and household tree plantations have also contributed to the improvement of the vegetation cover in northern Ethiopia (Jagger and Pender, 2003). In Tigray many degraded sites have been managed as exclosures, and attempts have been made to document the regeneration and ecology of the plants in these exclosures (Tesfay Yayneshet *et al*., 2009). Studies have indicated that vegetation recovery in the exclosures is quick, particularly in the younger stages (Tefera Mengistu *et al.*, 2005; Muluberhan Hailu *et al.,* 2006). Many of the exclosures on the degraded hillsides are covered with pioneer shrubs like *Acacia etbaica, Euclea racemosa, Dodonaea angustifolia*, and grass and many herb species (Tesfay Yayneshet *et al*., 2009). However, no significant

improvements in species diversity and biomass production were found after a decade of exclosures (Tesfay Yayneshet *et al*., 2009), signifying the need for introducing additional management measures to re-establish native key species.

Studies elsewhere indicate that restoration of the original flora may not be attainable for centuries, particularly in dry climates where moisture is a limiting factor (Woodwell, 1994). The natural forest remnants, which are expected to serve as sources of propagules for the restoration of the native species in the degraded sites, are not able to sustain the populations of the relevant forest species (Aerts *et al*., 2006). Thus, conserving the existing natural forest remnants needs conservation priority to keep species from local extinction.

The 1980s was a marked conservation turning point for Ethiopia when the impact of deforestation became more noticed and 58 National Forest Priority Areas (NFPAs) were set aside in different parts of the country for conservation purpose (EFAP, 1994; Reusing, 1998). Of these NFPAs, 46 of them were classified into different protected area management categories (EWCA, 2009) including Hugumburda-Gratkhassu NFPA from WU Floristic Region. Ever since it was known to exist, Hugumburda-Gratkhassu NFPA has passed through different management regimes. The forest was officially put under the auspices of State Forestry Agency in 1965 (SFCDD, 1997). Then in 1981 the area was identified as one of the NFPAs. Boundary demarcation, which is the basis for the current management of the Forest, was undertaken in 1993. The current study has used this demarcation as the basis and covers a total area of 21, 654 ha. Of this, about 533 ha is

plantation forest whereas the rest contains disturbed natural high forest, bushes, shrubs, agricultural plots and settlement area.

Previous studies carried out in the study area (Zenebe GebreEgziabher *et al*., 1998; Leul Kidane *et al*., 2010; Ermias Aynekulu, 2011) reported that Hugumburda-Gratkhassu NFPA ecosystems are already impacted from past disturbances which are reflected by fragmentation of forests into scattered inaccessible isolated patches and loss of unique biodiversity combinations. These studies also indicated a clear gap or lack of current comprehensive work on the status of vegetation. Zerihun Woldu *et al*. (2002) described, improving the management of the natural resources while providing ecological services and immediate economic needs are the major research and development challenges for the degraded areas of northern Ethiopia in particular and the drylands in east Africa in general. Nevertheless, sufficient scientific information is lacking on these issues. But such information is critically important in designing and executing appropriate conservation measures and sustainable utilization of the biological resources of the area. Thus, accommodating new and comprehensive conservation approaches assisted by GIS and remote sensing can contribute significantly to mitigate the problem of forest destruction in the study area.

Hence, in view of the need to develop more effective approaches to conservation and sustainable utilization of forest resources in Hugumburda-Gratkhassu NFPA, an investigation of the current status of vegetation encompassing species composition, patterns of community formation, population structure and regeneration, edge effects and

land-use and land-cover dynamics were conducted. In addition, analysis of patch area and fragmentation in relation to plant diversity was performed. No previous scientific research output of this kind is available for the study area to inform relevant government institutions about the management of the NFPA. Moreover, the study combines methods originating from different disciplines including plant taxonomy, landscape and plant ecology. As the main body of the thesis, the study result and discussion are presented following section of literature review, and materials and methods. Conclusion and recommendations are presented at the end of the dissertation. These data will enable local communities and forestry administration to design appropriate intervention strategies for managing the remnant Afromontane forest of the study area.

1.2. Research Problem, Questions and Objectives

1.2.1. Research problem and questions

The study area is at present a National Forest Priority Area where enhanced administration is being practiced. However, land degradation, in the very fragile and susceptible ecosystems of Hugumburda-Gratkhassu NFPA, is seriously threatening the natural resources to the extent of losing the distinctive plant diversity. As a result, there are no extensive natural forest cover in the area except remnants and patches of forest found in inaccessible parts. Thus, conserving the remnant patches of indigenous forest species before their entire extinction is an urgent task. Management of the ecosystems in the whole landscape has been a very difficult task. The main problem in the conservation

efforts relate to policy setting and implementation, which are often criticized for lacking an active participation of the local people (Campbell, 1991).

Earlier studies in the study area (Leul Kidane *et al*., 2010; Ermias Aynekulu, 2011) support this idea. These studies indicated degradation of the forest from past disturbances which are reflected by fragmentation of forests into isolated patches scattered in the inaccessible parts and loss of unique biodiversity combinations. These problems were the driving forces for the commencement of this project aimed at providing a comprehensive documentation and analysis of the ecology and plant diversity in Hugumburda-Gratkhassu NFPA along with impact of habitat fragmentation, edge effect and land use\land cover dynamics.

Therefore, the main research questions were

1. What are the floristic compositions of Hugumburda-Gratkhassu remnant forest patches?
2. What are the major plant community types and environmental factores determining the patterns of species distribution and community organization?
3. How are forest fragments organized in the study area and what are the main factors driving such variations?
4. Do the two large remnant forests (Gratkhassu and Hugumburda) differ in their plant species diversity and structure?
5. What is the regeneration status of these remnant forests? What are the major bottlenecks for regeneration of woody plants in the forest?

6. How do edge effect and patch area affect species richness and structure of the forest?

7. What are the rates and magnitude of land use\land cover changes in the study area? How do these changes relate to plant species diversity and landscape configuration?

8. What should be the appropriate actions for conserving forest resources of the study area?

Adequate scientific information is lacking on these problems. However such information is significantly important in designing and implementing appropriate conservation measures and sustainable utilization of the biological resources of the area. Therefore, the study required to investigate and document the diversity, structure, regeneration and status of vegetation to provide information and recommendation for the protection, management and sustainable utilization of the plant diversity of the study area. Based on these research questions the following study objectives were derived.

1.2.2. Objectives

1.2.2.1. General objective

This research was aimed at documenting and analyzing the ecology, plant diversity, structure, regeneration and status of vegetation in Hugumburda-Gratkhassu NFPA, northeastern Ethiopia, so as to provide conceivable recommendations appropriate for conservation, management and sustainable natural resource utilization.

1.2.2.2. Specific objectives

The specific objectives of the research were to:

1. Document plant species diversity in the study area;

2. Investigate the spatial patterns of plant species distribution in relation to the underlying causes determining these patterns;

3. Identify and describe the plant community types and analyze important environmental factors determining patterns of community formation; assess and compare plant species diversity between forest remnants in the study area;

4. Study the species similarities between Gratkhassu and Hugumburda forest fragments;

5. Estimate the total number of plant species of the study area;

6. Analyze current regeneration and structure status of selected keystone species in the study area and identify possible factors that hamper their regeneration ability;

7. Analyze forest fragmentation and patchiness in the study area;

8. Produce spatially and temporally quantified information on landcover dynamics in the study area; identify the driving forces for the landcover dynamics and patchiness; and

9. Provide the scientific background and conceivable recommendations that would contribute towards the development and sustainable utilization of natural resources of the NFPA.

CHAPTER TWO

2. LITERATURE REVIEW

2.1. Vegetation Types in Ethiopia

Ethiopia ranges in altitude from the highest peak (Ras Dejen, 4620 m a.s.l.) down to the depression of the Kobat sink (Afar depression about 110 m below sea level) (EFAP, 1994). Extensive highland plateau, with an altitude of over 2,500 m a.s.l., cover 40% of the country (Zerihun Woldu, 1999). The great African Rift Valley runs from north to south bisecting the plateau, and in combination with the surrounding lowlands, this feature isolates and separates Ethiopian plateau from other parts of the continent. The diverse landscape, the rift valley and surrounding lowlands have given Ethiopia a wide range of habitats, and a large number of endemic plants and animals (EFAP, 1994).

Different scholars, such as Logan (1946), White (1983), Friis and Sebsebe Demisew (2001), Friis *et al*. (2001; 2011) have tried to study the Ethiopian vegetation by employing different methods. The results of their study offered general description of the vegetation types and their floristic composition in different parts of Ethiopia. These authors set significant input towards the understanding of the vegetation of Ethiopia. As a result, the various vegetation types of Ethiopia have been formally grouped into eight vegetation types (Sebsebe Demissew, 1996; Zerihun Woldu, 1999; Friis and Sebsebe Demissew, 2001) although Friis *et al*. (2011) classified it into 12 potential vegetation types. These are desert and semi-desert scrubland, small-leaved deciduous woodland, moist evergreen montane forest, lowland semi-evergreen forest, broad-leaved deciduous woodland, dry evergreen Afromontane forest and grassland complex, afro-alpine and

sub-afroalpine vegetation and aquatic ecosystem**.** From these vegetation types, based on the broad categorization of Friis (1992) and Zerihun Woldu (1999), the study area includes dry evergreen Afromontane forest and grassland complex.

2.1.1. Dry evergreen Afromontane forest and grassland complex vegetation

The Ethiopian highlands encompass over 50% of the African land area covered by Afromontnae vegetation (Tamrat Bekele, 1993), of which dry Afromontane forests comprise the largest part (Demel Teketay, 2005). This vegetation represents a complex system of successions involving extensive grasslands rich in legumes, shrubs and small to large trees to closed forests at altitudinal range of 1350 - 3550 m a.s.l. with an average annual temperature and rainfall of 14 - 25°C and 700 - 1100 mm, respectively (Friis, 1992; Zerihun Woldu, 1999). According to IBC (2009) and Friis et al. (2011), this vegetation occurs largely as secondary growth after disturbance. The vegetation is largely characterized by *Juniperus procera, Olea europapea* ssp. *cuspidata, Prunus africana,* and *Ekebergia capensis.* Important grass genera are *Eragrostis, Pennisetum, Panicum, Sporobolus* and *Hyparrhenia* (Tewolde Berhan Gebre Egziabher, 1988) while herbaceous legumes are species of *Indigofera, Trifolium, Crotalaria* and *Tephrosia* (Friis *et al*., 2011). Climbers include *Embelia schimperi, Rubia cordifolia*, *Urera hypselodendron* and *Jasminum abysinicum* (IBC, 2009).

The prominent features of tropical dry forests are their seasonality with respect to rainfall compared with the rain forests where the environment is stable throughout the year. Dry evergreen montane forests experience long dry seasons (4-8 months) and the rainy period is somewhat unreliable. During the dry season, not only moisture stress but also

temperature increases and daytime humidity drops and watercourses either dry up or greatly diminish inflow (Demel Teketay, 2005). Degradation in this zone is very high and even severe in the northeastern Ethiopia where, forests have virtually disappeared, as a result most of the mountains sides are bare, valleys have been gullied, and springs and streams, which used to have water the whole year round are now mainly dry in the dry season (Zerihun Woldu, 1999).

Soil degradation can appear as a direct result of past agricultural practices in the highlands. The dissected terrain, the extensive areas with slopes above 16 percent, which are typical characteristics of the study area, lead to accelerated soil erosion once deforestation occurs. Besides, some of the farming practices within the northern highland encourage erosion. These include cultivation of cereal crops which require the preparation of a finely tilled seed bed and down slope plowing to facilitate drainage. Furthermore, insecurity of land and tree tenure has discouraged farmers from investing in soil conservation practices (Badeg Bishaw, 2009). As a result, these parts of the Ethiopian highlands have remnants of natural forests scattered in inaccessible, protected and religious areas (Alemayehu Wassie, 2007) suggesting that the highlands were once covered by high forest (White, 1983; Friis, 1992). Lastly, it can be said that, these problems originated from increased population pressure, agricultural encroachment, low productivity, frequent fire, intense grazing and deforestation are seriously degrading this unique and fragile ecosystem which eventually lead to loss of the unique combination of the biodiversity in Hugumburda-Gratkhassu NFPA.

2.1.2. Threats to plant diversity in Ethiopia

Owing to the diversity in climatic variations, Ethiopia is one of the richest centers of biological diversity and genetic resources in the world (EPA, 2003; NBSAP, 2005). The country has the fifth largest flora in tropical Africa with a rich endemicity (WCMC, 1992). Noss (1999) pointed out that growing human population and the demand for natural resources have put great pressure on the biodiversity wealth of the world. If present trends in population growth continue, deterioration of natural resources will be more rapid in the future and will result in a great loss of biodiversity (Ensermu Kelbessa *et al.,* 1992; Zerihun Woldu, 1999). In addition, Ensermu Kelbessa *et al.,* (1992) mentioned that, limitations to the opportunities for income generation caused by ecological and socioeconomic constraints exacerbated by a mounting population pressure have, over the years, forced people to cultivate marginal lands and allow over grazing and felling of trees, thus catalyzing spiral environmental degradation and deforestation in Ethiopia. This loss in biological diversity ultimately results in economic losses of the country and the world as a whole, including reduction in the quantity of carbon that can be sequestrated from the atmosphere (EFAP, 1994).

According the definition of FAO (2001), the vegetation of Ethiopia that may qualify as 'forests' are natural high forests, woodlands, plantation and bamboo forests, with an estimated area of 35.13 million ha. The recent data on forest resources of Ethiopia reported in FAO (2010) puts Ethiopia in countries with forest cover of 10 - 30%. According to this report Ethiopia's forest cover is 12.2 million ha (11%). It further

indicated that the forest cover showed a decline from 15.1 million ha in 1990 to 12.2 million ha in 2010, during which 2.65% of the forest cover was deforested.

Even though it is difficult to establish the precise cover of the past as well as the present forest vegetation of Ethiopia, it is believed that a substantial portion of the land area in the highlands of Ethiopia was covered with forests having wider coverage than at present (Friis, 1986). The presence of a number of isolated large-sized trees, even on farmlands, or patches of forests around churchyards and religious burial grounds in this country indicate the occurrence of vast expanses of high forests (Tamrat Bekele, 1993; Zerihun Woldu, 1999). About 35% of the landcover of Ethiopia consisted of high forests before human settlement took place and if savanna woodlands are included, about 66% of the country's total area was covered with forests (EFAP, 1994). By the early 1950s, high forests had been reduced to 16% of the total land area. Forests in the entire country then went to decline at a faster rate and had dwindled to 3.6% by 1980, and 2.6% by 1987 (EEPFE, 2008). The present estimate shows only about 3% of the total land of Ethiopia is covered with high forests (EEPFE, 2008; FAO, 2010). In Ethiopia, excessive exploitation of natural forests without a minimum repair effort, the extension of cultivation to marginal lands by clearing and burning fragile ecosystems, forest fires, lack of proper forest management, of compatible forest legislations, lack of constant and sustainable institutional organization have all resulted in almost total deforestation that led to the highest magnitude of soil erosion, exposing the country to drought and famine (EFAP, 1994). According to FAO (2010), the annual rate of deforestation in Ethiopia between 1990 and 2010 was estimated at 141,000 ha/year. The fast rate of deforestation in this

country has seriously exposed the fertile soil to the highest magnitude of erosion (EFAP, 1994).

The Ethiopian highlands constitute large parts of the Afromontane region of Africa, which stretch from Cameroon to eastern Africa, where biodiversity hotspots exist (Conservation International, 2007). However, as in many tropical types of vegetation, disturbance has changed the structure and floristic composition of vegetation in northern Ethiopia for millennia (Darbyshire *et al.,* 2003). According to Sebsebe Demissew (1980), most highlands of Ethiopia have felt the impacts of fundamental changes brought by the vegetation deterioration emanated from the effects of human pressure. Currently, natural high forests in Ethiopia mainly occur in the southwestern part of the country, while the forests that originally existed in northern Ethiopia have almost disappeared (EFAP, 1994), only scattered patches of forest are found in inaccessible, protected and religious places (Alemayehu Wassie, 2007).

The major causes of forest degradation in northern Ethiopia are conversion to agricultural land (Nyssen *et al*., 2004) and grazing (EFAP, 1994). However, the attention given to conservation and sustainable use of these biological resources is inadequate due to low level awareness about the role of the forests (Demel Teketay, 2005). Despite the high conservation value, the natural vegetation in northern Ethiopia has poorly been studied (Friis, 1992), and information necessary for effective biodiversity conservation is scant (Aerts *et al.,* 2006). Consequently, depletion of vegetation by anthropogenic and natural factors has led to the decline in number and area of distribution of many plant species.

About 120 threatened plant species that are in Ethiopia are documented in the work of Ensermu Kelbess *et al.* (1992), indicating effects of natural factors and human interference on the country's vegetation. Reversing long-existing trend of deforestation might be possible (though it takes a lot of time) as long as appropriate measures are taken by the concerned governmental sector and the people as a whole.

To address these threats, it is critical to raise awareness of government decision makers' in relation to the relevance of natural resource policies; facilitate dialogue among the government, civil society, and national and international organizations; and to closely examine and revise those policies requiring clarification or harmonization. Family planning can also help to reduce family size and slow population growth, and efforts toward generating alternative livelihoods activities can give people options other than farming or herding that are less dependent on limited natural resources. Other key actions include increased enforcement of rules and boundaries; environmental education; and efforts to strengthen the relationships among protected areas and communities, with emphasis on returning the benefits of the protected areas to those communities. Taking the lost opportunities into account, foresters and conservationists have to develop new initiatives to respond to the convergence of local communities and forests (Regassa Feyissa, 2001).

2.2. Perspectives on Plant Community Organization

Plant community is a collection or association of plant species within a designated geographical unit, which forms a relatively uniform patch, distinguishable from neighboring patches of different vegetation types. Differentiating plant communities has

been at the heart of vegetation science for century, with traditional focus on the composition, distribution and classification of plant communities (Kormondy, 2005). According to Grubb (1987), plant communities are defined as "assemblage of functionally similar species population found in one habitat type in one area, and integrated to a degree by complementarities, competition and dependence". If communities survive as a similar association or combination of plant species occurring frequently on similar habitats within a limited area as a result of competition (Walter, 1985), complementarities and dependence (Chapman and Reiss, 1992), then there should be some sort of patterns non-random, repetitive organization that has been caused by some general causal processes (Brown and Lomolino, 1998).

As it is noted by Palmer and van Staden (1992), investigation of community patterns, such as community unit and individualistic concepts, helps one to understand how plant communities and individuals may respond to environmental variables along a gradient. Environmental factors such as soil, climate and topographic conditions, and biotic factors such as competitive abilities of species, facilitative aspects such as shade tolerating vs. light phase species (Chapman and Reiss, 1992), influence transition of communities and the distribution of species. The spatial alteration of the environmental factors and the succeeding reply of the individual populations in a species to these changes will influence the allocation of species and therefore, transition of communities (Walter, 1985). Besides to the biotic and environmental factors, community association may be influenced by temporal factors. These comprise the dispersal mechanisms and subsequent gathering of species in a site and the developmental stage of vegetation or succession (Jacquemyn *et*

al., 2001). There has long been a debate on community structure in the past (Collins *et al.,* 1993).

Two general perspectives have been proposed to explain the structure of the plant community: the community-unit model of Clements (1936) and the continuum model of Whittaker (1953) and Curtis (1959), the latter based on Gleason's individualistic distribution of species. Experimental evidences support each of them (Collins *et al*., 1993). Community-unit model describes that communities are highly structured repeatable and identifiable association of species controlled by environmental gradients. On the other hand, continuum model states that plant communities change gradually along complex environmental gradients, such that no distinct associations of species can be identified (Whittaker, 1953).

Community-unit model viewed communities as holistic and mutually dependent and predicts that groups of species or communities would replace one another along certain gradients. Within each grouping, most species have similar distribution and the end of one group coincides with the beginning of another (Shipley and Keddy, 1987). In this case species with wider distribution of the community are separated from each other based on indicator or character or dominant species in combination with a distinctive floristic composition (Clements, 1936). Discontinuities in communities could occur if few potential dominant species control the environment and avoid others by competitive exclusion and subsequent modification of the environment by the vegetation (Roberts, 1987).

Continuum perspective contemplates that species have individualistic response along environmental gradient. According to this view, plant communities change gradually

along the complex environmental gradient and hence identification of distinct community association is not possible (Collins *et al.,* 1993). Rather each species has its own physiological tolerance limit to environmental variables and the environmental variables would also fluctuate and vary across spatial gradients. Although the debate still exists, the community organization perspectives (discrete and continuum) discussed above offer us insights into how plant species act in response to the environmental variables along complex gradients. Analysis of empirical data showed that the normal curve distribution assumption of continuum model is not consistent and universal pattern, rather more skewed form of distribution is expected (Shipley and Keddy, 1987). On the other hand, Roberts (1987) has argued that both community unit and continuum perspectives give greater emphasis to environmental factors and little attention was given to the modification effect of the vegetation and their subsequent influence of the community patterns.

In response of this, Collins *et al*. (1993) had proposed a third model, hierarchical continuum model by considering arguments from different ecological works related to patterns of community organization. As suggested by Roberts (1987), this model incorporates the dynamic nature of vegetation and environmental gradient variations. In regard to the dynamic nature of vegetation, distribution patterns and abundance of species will change along the environmental gradients (Collins *et al*., 1993). Environmental resources are not uniformly distributed, rather show sort of patchiness in their distribution. In response to this, plant species do not distribute uniformly. Considering such relationship, hierarchical continuum perspective assumes that some species are distributed widely across the sample region while others have limited distribution and

still others have much restricted distribution. Species with wider ranges of distribution and much restricted ranges (rare species) were not used by association analysis for community classification (discrete community concept). While, in the case of the continuum concept, the normal distribution curve is not a universal case since resource variable is not always continuous but rather fragmentation is actually observed (Collins *et al*., 1993).

In addition, the dynamic nature of vegetation in modifying the environment has its own influence on the organization of the community. Thus, the distribution of species and patterns of abundance assume a hierarchical organization where species with wider, intermediate and restricted distribution ranges show some sort of hierarchies rather than a continuum or discrete alone. A plant community with a hierarchical continuum concept can be understood as combination of plants that are dependent on their environment, influence one another, and modify their environment (Mueller-Dombois and Ellenberg, 1974). The hierarchical continuum perspective of Collins *et al*. (1993) can, therefore, be viewed as modern synthesis of the continuum and association models that recognizes the validity of both views and their complementarities in their application for different aspects of community analysis. In this study, classification of community types was carried out following the hierarchical continuum perspective of Collins *et al*. (1993).

Vegetation classification and ordination are multivariate techniques used to identify patterns in plant community structure and are complementary approaches used for data exploration and reduction (Leps and Smilauer, 2003). They assist in the arrangement of vegetation patterns into ecologically meaningful sets. Vegetation classification attempts

to define discrete, repeatable classes of relatively homogenous vegetation communities, in the hierarchical structure, about which reliable statements can be made. Ordination is a tool for ordering or arrangement of a vegetation sample in relation to each other in terms of their similarity of species and/or their associated environmental controls. It summarizes plant community data, providing an indication of the true nature of variation within the vegetation of the area under study (Kent and Cooker, 1992).

Ordination can be direct or indirect. With direct gradient analysis, sample units (e.g., plots) are ordinated based on measured environmental factors in those sample units (McCune and Grace, 2002). Gradient analyses can be univariate where sample units are ordinated along a single environmental factor or multivariate. Species distribution along an elevation gradient, for instance, is a univariate gradient. Indirect ordination ordinates sample units considering the variation within the vegetation independently of the environmental data. Indirect gradient analyses are more widely used than direct gradient analyses (Kent and Coker, 1992).

Indirect ordination is performed in ordination methods like principal component analysis, Bray-Curtis (polar) ordination, reciprocal averaging, correspondence analysis, and detrended correspondence analyses, while non-metric multi-dimensional scaling (NMDS) and canonical correspondence analysis (CCA) are rarely used (McCune and Grace, 2002). Each of the above ordination techniques has its pros and cons, and the choice of ordination method also depends on the type of the data, the sampling effort and the objectives of a study (Southwood and Handerson, 2000). Principal component analyses, for instance, requires linear relationships between variables and normal distribution of

variables (McCune and Grace, 2002), which rarely occurs in vegetation data. McCune and Grace (2002) recommend the use of NMDS in community ecology because it can be used with nonnormal data and avoids the assumption of linear relationships between variables.

2.3. Species Diversity, Richness, Evenness and Similarity

2.3.1. Species diversity

Species diversity can be seen from different approaches in terms of alpha, beta and gamma diversity. Alpha diversity (α) refers to the number of species within a sample area or community. Beta diversity (β) describes the difference in species composition between two adjacent sample areas or communities. It is sometimes called habitat diversity because it represents differences in species composition between different areas or environments (Kent and Coker, 1992). According to Zerihun Woldu (in press), beta diversity calculates the number of species that are not the same in two different communities. Beta diversity can help in defining regional-scale diversity and assessing changes across environmental and bio-geographic gradients through characterizing the rate of species accumulation from place to place. Beta diversity is low when the overlap between the species composition of two quadrats is high, and is highest when the samples have no species in common at all. Gamma diversity (γ-diversity) describes regional differences in species composition (e.g., the difference in species composition between comparable habitats on two adjacent mountain ranges). It is the measure of the overall diversity for the different ecosystems within a region. Gamma diversity is the diversity in sites or habitats at a more local scale and the differentiation among those habitats (Kent

and Coker, 1992). According to this reasoning alpha diversity and beta diversity constitute independent components of gamma diversity.

Areas with the very highest levels of species richness are those where there are many species per quadrat, but no two quadrats on a transect area alike in their species composition. In regions of the lowest species richness there are few species in any sample, and the vegetation is spatially monotonous. Even if the building blocks are not identical, extreme environmental conditions can lead to similar life forms.

The description of plant community involves the analysis of species diversity, evenness and similarity. Diversity and equitability of species in a given plant community are used to interpret the relative variation between and within the community and help to explain the underlying reason for such a difference. Species are the fundamental units of biological organization, and any small changes in the species diversity may alter to some extent ecosystem functions and services (You *et al*., 2009). Species diversity, species richness and biodiversity are widely used terms (sometimes interchangeably) in ecology and natural resource management.

Species diversity is a function of the number of species present (species richness or number of species) and the evenness with which the individuals are distributed among these species (species evenness, species equitability, or abundance of each species) (Spellerberg, 1991). According to Hamilton (2005), this definition may be the best one available at the moment. You *et al*. (2009) emphasized that the concept of species

diversity should be restricted to this extent if it should have any useful meaning. Species diversity indices take two aspects of a community into account, namely species richness and evenness or equitability (the distribution of abundance among the species) (Hamilton, 2005). Therefore, the description of plant community involves the analysis of species diversity, richness and evenness indices. These measures in a given plant community are used to interpret the relative variation between and within the community and help to explain the underlying reasons for such a difference.

2.3.2. Species richness

In biological diversity, the concept of species richness is one of the oldest and most fundamental concepts. Species richness can refer to the number of species present in a given area or in a given sample, without considering the number of individuals examined in each species (Hamilton, 2005). It is the simplest way to describe community and regional diversity (Magurran, 2004), and this variable i.e. number of species (species richness), forms the basis of many ecological models of community structure. Quantifying species richness is important, not only for basic comparisons among sites, but also for addressing the saturation of local communities colonized from regional source pool. Maximizing species richness is often a goal of conservation studies.

Biodiversity measures (i.e. species richness and species diversity) have been widely used as indicators of ecosystem status, and play a critical role in studies dealing with the assessment of human impact on ecological systems. Because biodiversity of any ecosystem is far too complex to be comprehensively quantified, suitable indicators of biodiversity are needed. Conceptually, species richness appears as the most intuitive and

simple parameter to measure biodiversity. However, for several reasons, to determine the true species richness of a community is not an easy task (Magurran, 2004). Thus, in this study different non-parametric species richness estimators were used to estimate the true number of species found in the NFPA based on the number of species observed in the study quadrats.

2.3.3. Species similarity among communities

Biotic similarity measures the extent to which two or more sites are similar in their species composition and relative abundance distribution. The concept of biotic similarity is important at large spatial scales for the designation of biogeographic provinces that harbor distinctive species assemblages with both endemic and shared elements. Measuring biotic similarity between communities is another way of measuring beta diversity where higher biotic similarity between communities would mean lower beta diversity.

2.4. Plant Population Structure and Regeneration Patterns

Studies on population structure and density of major canopy tree species help to understand the status of regeneration of species, and, thereof, management history and ecology of the forest. Plant population structure shows whether or not the population has a stable distribution that allows continuous regeneration to take place. Size class distribution as diameter at breast height (DBH) and height are used to evaluate structure and regeneration of species. The structure of the species may have inverted J-shape pattern, which is an indicator of stable population structure and is a reflection of healthy regeneration potential of the species exhibited in a natural forest where disturbance is minimum (Harper, 1977; Getachew Tesfaye *et al.,* 2010). Any population pattern differed

from inverted J-shaped, have been attributed to disturbed forest (Poorter *et al*., 1996), e.g. bell shape, J-shape and U-shape.

The variation in relative abundance of size classes and population structure of species is the result of past and present disturbance as well as management history of the forest. Based on intensity of disturbance species show variation in population structure pattern reflected through difference in the abundance of different size classes (Tamrat Bekele, 1994). Size class distribution or population structure, therefore, gives good indication of the impact of disturbance and forest successional trends.

The population structure, characterized by the presence of sufficient population of seedlings, saplings and adults, indicates successful regeneration of forest species. Seedling densities in forest understory's are dynamic and rates may vary among species and in gap and shade environments. Information on tree seedling ecology can provide options for forest development through improvement in recruitment, establishment and growth of the desired species. Regeneration studies have significant implication on the conservation and restoration of degraded natural forests (Getachew Tesfaye *et al.,* 2010). Thus, regeneration status of woody species can be predicted by the age structure of their populations. The study of regeneration of forest trees has important implications for the management of natural forests, and in one of the thrust areas of forestry. Research in this field contributes to planning, conservation and decision making in forest resources management programs (Pokhriyal *et al*., 2010).

2.4.1. Bottlenecks of natural regeneration in forest ecology

Regeneration is a key ecological process and a central component of forest ecosystem dynamics and restoration of degraded forest lands. Sustainable forest utilization is only

possible if adequate information on the regeneration dynamics and factors influencing important canopy tree species are available. If native long-lived trees are unable to survive and regenerate in a given forest, then there is little hope for maintaining and semblance of normal forest functioning in the long term (Harrington *et al.,* 1997). Therefore, the effectiveness to maintain and expand forests will depend mainly on the success of natural regeneration of the component woody species.

Densities of seedlings and saplings are considered an indicator of the regeneration potential of a forest. A species is considered in a good regeneration status if the number of seedlings is greater than saplings which in turn greater than tree (seedlings > saplings > tree); fair regeneration is a condition of a species where number of seedlings is greater than the number of saplings and the number of saplings is also less than the number of trees (seedlings > saplings < tree); poor regeneration is a status of regeneration where density of sapling is greater than seedlings. A species is considered not regenerating if it survives only in adult stage and a species is newly arrived if it present only in seedling stage (Pokhriyal *et al*., 2010).

Species composition, structure and regeneration process could be affected by the frequency and magnitude of disturbance. A good understanding of factors promoting or hampering seedling establishment of tree species could help in future restoration of degraded landscapes through appropriate management system.

Factors that potentially influence regeneration at the early stage are those that determine the probability of seedling establishment and those that affect seedling survival and growth. Seed availability can be influenced by local seed rain (recently dispersed seeds), soil seed bank (dormant seeds in the soil) and seedling bank (established, suppressed

seedlings in the understory) (Demel Teketay, 2005). Insufficient input of seed rain both in quality and quantity in forests may be attributed to fragmentation which brings about change in tree phenology as a result of increased mortality of reproductive individuals and/or genetic drift that might lead to a reduction of flowering and fruiting, fewer pollinators and dispersers, and increased pre-dispersal seed predation (Fenner and Kitajima, 1999). Reduced seed rain, lack of persistent soil seed bank and intense seed predation can decrease seed availability; this in turn creates a bottleneck for seedling establishment.

In some instances, the available seeds may not be able to establish seedlings due to unfavorable microsite. Drastic variability and fluctuation in the microenvironment may lead to either seed mortality or seed dormancy resulting in reduced germination and successful establishment (Demel Teketay, 2005). Another factor hampering early stage regeneration is related to seedling survival and growth. Once seeds have germinated and seedlings are established the next challenge is to survive, grow and join sapling and adult stage. Seedling mortality may increase due to unfulfilled optional requirements like light and moisture. In fragmented forests, a consequence of drastic variability and fluctuation in the physical microenvironment is crucial. Physical damage and destruction due to trampling and grazing, competition by weeds and herbs raise seedling mortality (Kitajima and Fenner, 2000).

There are other factors determining regeneration dynamics of native woody species. From the long lists of these factors grazing is a common practice in the study area. Grazing is believed to have both positive and negative roles in the regeneration of species. It has been suggested that a light level of grazing can increase tree regeneration

by removing competitive vegetation, remove fire hazard and fertilizing by droppings. Large herbivores can also create patches of bare ground through grazing and trampling. These disturbances may create "safe sites" for seeds to germinate (Kitajima and Fenner, 2000). On the other hand, continuous and intensive grazing may cause irreversible damage. Unrestricted animal movement may also destroy seeds and press them deep in the soil from where they are not able to emerge. Trampling and grazing of seedlings may lead to low seedling survival (Henriquez and Simonetti, 2001). Though grazing has both negative and positive effects, balance of evidence indicated that free and intensive livestock grazing has deleterious effects on regeneration of species through trampling and soil compaction.

Regeneration studies contribute to planning, conservation and decision making in forest resources management programs (Pokhriyal *et al*., 2010). Several studies carried out on regeneration of different forests and forest patches elsewhere in Ethiopia (Alemayehu Wassie, 2007; Getachew Tesfaye *et al.*, 2010; Getinet Masresha, 2014), concluded that most forests are dominated by shrubby and lower size secondary type and regeneration rate of most indigenous species are hampered by anthropogenic activities. Therefore, conservation and sustainable utilization of the few remnant natural forests in Ethiopian highlands, which are potential sources of propagules, is an urgent task (Alemayehu Wassie, 2007). Information on the regeneration ecology of the different forest types in Ethiopia is scanty (Demel Teketay, 2005). Part of this study, therefore, focused on the natural regeneration of woody specie in Hugumburda-Gratkhassu NFPA.

2.5. Habitat Fragmentation and Edge Effect on Biodiversity

2.5.1. Habitat fragmentation

Habitat loss and fragmentation are major causes of global biodiversity loss, as they frequently result in reductions in population sizes leading to extinction (Turner *et al.,* 2001). Serious forest disturbance leads to deforestation, forest fragmentation and degradation, and subsequent exotic species invasion, all of which adversely affect plant diversity (Raghubanshi and Tripath, 2009). Habitat fragmentation or subdivision is defined as a process in which a wide area of habitat for example, natural forest is changed into a number of smaller patches of smaller total area, isolated from each other by a matrix of different landuses distinct from the previous landuse type (Lindenmyer and Fischer, 2006). Habitat fragmentation could be caused either by natural disturbance (e.g. fire, wind throw) or due to anthropogenic disturbance (human caused) which includes clearing of forest for agriculture, settlements, road and dam construction and logging. However, fragmentation due to anthropogenic disturbance is more serious than that caused by natural disturbance especially from species extinction point of view (Pimm and Raven, 2000). The rapidly growing world population increased human activity which exacerbates deforestation and fragmentation, threatening many of our plant species and humanity is rapidly destroying habitats that are most species-rich (Pimm and Raven, 2000), by doing so human activities have dramatically accelerated the global rate of species extinction.

Fahrig (2003) distinguished four different effects of habitat fragmentation on habitat pattern. These include; (a) reduction in habitat amount, (b) increase in number of habitat patches, (c) decrease in sizes of habitat patches, and (d) increase in isolation of patches.

As explained by Lindenmyer and Fischer (2006), fragmentation is more than habitat removal since it causes not only habitat loss but also changes the properties of the remaining habitat. It changes the physical environment and biogeographic entities. Hence, habitat connectivity is considered to be very important to dispersal success, persistence, and genetic diversity of species in fragmented landscapes. In most cases fragmentation is strongly associated to human induced disturbances and it influences plant community structure and maintenance of biological diversity (Yates *et al*., 2000). In this regard, loss and fragmentation of natural habitat are considered major threats to global biodiversity (Pimm and Raven, 2000).

2.5.2. Edge effect

Fragmentation creates abrupt changes in habitat or vegetation type with the resulting edge acting as a transition zone between the fragments and surrounding matrix habitats (Honnay *et al.,* 2010). Forest fragmentation results in changes in environmental conditions at forest edges (abiotic effects), changes in species composition (direct biotic effects), and alteration of species interactions (indirect biotic effects); these are collectively referred to as "edge effects". Abiotic effects at forest edges include increased radiation levels, air and soil temperatures, vapor pressure deficits and wind speeds (Laurance and Curran, 2008).

Concentrations of nutrients and pollutants may also be elevated at edges, as a result of human activities in the adjacent matrix. Edge effects on microclimate tend to vary according to adjacent vegetation type and aspect, with the strongest effects occurring next to open habitats (Harper *et al*., 2005). Changes in plant communities frequently occur at edges in response to these abiotic effects, and include changes in species composition and

seedling recruitment patterns, lower tree basal areas and elevated canopy tree mortality. Although the majority of forest-dwelling species are likely to be negatively affected by forest fragmentation, some species appear to be unaffected or respond positively to edge and area effects. For example, large, old-growth trees may be replaced by shorter-lived successional trees and the density of lianas may increase near edges (Harper *et al*., 2005; Honnay *et al.,* 2010).

Altered species interactions at edges include predation, competition, herbivory, pollination and seed dispersal. Increased rates of species turnover or hyperdynamism in population and community dynamics can also occur at edges. Flower production and pollination rates may increase at forest edges in some plant species but decrease in others, and this will have a major effect on fruit and seed set and ultimately population viability. Most empirical studies have found that edge effects penetrate less than 150 meters into fragmented forests (Harper *et al*., 2005; Broadbent *et al*., 2008). The combined repercussion of the above factors leads to population reduction in susceptible forest interior species that require a habitat, at a minimum distance from habitat edge. Other impacts include increase in tree mortality and penetration of exotic species displacing rare and endemic native species.

The physical structure of an edge can have a major influence on the strength of edge effects in fragments, as edges mediate many of the interactions between the patch and the matrix (Harper *et al*., 2005). Following edge creation, plants may alter their growth habit and create a 'curtain' of dense vegetation formed from lateral branches of trees and shrubs, which can impede the flow of organisms and materials into fragments. The age of an edge will also influence the strength of edge effects, as forest plants may take a

number of years to respond to edge creation. Newly created forest edges (often referred to as 'hard edges') are usually more open and permeable than older edges, which may become buffered by dense vegetation over time (i.e. 'soft edges') (Broadbent *et al*., 2008).

2.5.3. Effect of habitat fragmentation and edge on biodiversity

Forest fragmentation has impact on biodiversity, increasing isolation of habitats, endangering species of plants, mammals and birds and on a variety of population and community dynamic processes (Valladares *et al.,* 2006). However, the effects of habitat fragmentation on species diversity vary among different habitats and taxa. These effects could be both positive and negative (Fahrig, 2003). Positive effects include the creation of edge habitat increasing the abundances of edge or gap species and negative impacts include increasing the local rate of extinction by reducing population sizes through reducing habitat size and/or making patches of habitat, creating forest edges and altering microclimate at forest edges, changing forest dynamics and increasing predation at forest edges. As a result, fragmentation has been described as one of threats to global biological diversity.

The current extinction rate of species ranges from approximately 1000 to 10,000 times higher than natural extinction rates (Kellert and Wilson, 1993), and if this trend continues, as many as two million species of plants and animals will be exterminated worldwide by the middle of this century. Since biodiversity is essential for the sustainable functioning of agriculture, forest, and natural ecosystems on which human depend, this forecast is alarming. Biotic factors are rarely causes of species extinction, but habitat destruction, over exploitation and the impact of exotic species, which are all manmade

problems, are the major causes of modern species extinction. The effects of fragmentation and edge is more severe in tropical forests where diversity is high and forests are being removed and fragmented at an alarming rate (Pineda and Halffter, 2004).

These impacts of fragmentation and edge on biodiversity may be resulted from one or a combination of four separate effects: forest fragmentation *per se*, the loss of habitat during fragmentation, habitat degradation following the isolation of fragments and the effect of isolation *per se* (Harrison and Bruna, 1999).

Fragmentation isolates and reduces size of habitat patches, increasing vulnerability of local population of plants and animal species to environmental and demographic threats. The broader habitat loss increases the distance between habitat patches and the hostility of inter-patch habitat for certain species. In metapopulation, it leads to reduction in the migration rates among habitat patches, causing increase in extinction of local populations and decrease in rates of recolonization of vacant patches (Kellert and Wilson, 1993).

Generally, the effects of fragmentation and edge on biodiversity operate at multiple spatial scales, ranging from local (at different distances from the edge within fragments), to patch (whole fragments), to landscape and regional scales (Laurance and Curran, 2008). These effects can be examined separately, but they are not truly independent because the patterns and processes of fragmentation are usually inter-correlated and interact with each other. In parts of this study the effects of forest fragmentation and

edges on plant diversity at multiple spatial scales ranging from local to landscape scales were tried to be analyzed.

2.6. Temporal Changes in Land use\Land cover

2.6.1. Definitions and rationale of land use\land cover change study

Landcover is defined by the attributes of the earth's land surface captured in the distribution of vegetation, water, desert and ice and the immediate subsurface, including biota, soil, topography, surface and groundwater, and it also includes those structures created solely by human activities such as mine exposures and settlement (Chrysoulakis *et al*., 2004). On the other hand, landuse is the intended employment of land management strategy placed on the landcover by human agents, or land managers to exploit the landcover and reflects human activities such as industrial zones, residential zones, agricultural fields, grazing, logging and mining among many others (Olson and Maitima, 2006). Landuse change is defined to be any physical, biological or chemical change attributable to management, which may include conversion of grazing to cropping, change in fertilizer use, drainage improvements, installation and use of irrigation, plantations, building farm dams, pollution and land degradation, vegetation removal, changed fire regime, spread of weeds and exotic species and conversion to non-agricultural uses.

Landuse and landcover (LULC) changes may be grouped into two broad categories as conversion and modification. Conversion refers to changes from one cover or use type to

another, while modification involves maintenance of the broad cover or use type in the face of changes in its attributes (Baulies and Szejwach, 1998).

According to Lambin (2005), sustainable resource use refers to the use of environmental resources to produce goods and services in such a way that, over the long term, the natural resource base is not damaged so that future human needs can be met. One of the most significant global challenges in this century relates to management of the transformation of the earth's surface occurring through changes in land use\land cover. For example, landuse in East Africa has changed swiftly over the last half-century: expansion of mixed crop-livestock systems into former grazing land and other natural areas and intensification of agriculture are the two largest changes that have been detected (Olson and Maitima, 2006).

Accordingly, landcover classification has recently been a hot research topic for a variety of applications. A great deal of research has been conducted throughout the world in an attempt to understand major shifts in land use\land cover and to relate them to changing environmental conditions. Understanding the mechanisms leading to land use\land cover changes in the past is crucial to understand the current changes and predict future ones. These changes occurred at different time periods, paces, and degrees of magnitude and with diverse biophysical implications (Baulies and Szejwach, 1998).

Therefore, LULC research needs to deal with the identification, qualitative description and parameterization of factors which drive changes in land use\land cover, as well as the integration of their consequences and feedbacks (Baulies and Szejwach, 1998). However, one of the major challenges in LULC analysis is to link behavior of people to biophysical

information in the appropriate spatial and temporal scales (Codjoe, 2007). But, it is argued that land use\land cover change trends can be easily assessed and linked to population data, if the unit of analysis is the national, regional, district or municipal level.

Land use\land cover changes result from various natural and human factors within social, economic and political contexts. Hence, the local human activities expressing the drivers can be determined by measuring the rates and types of changes and analyzing other relevant sources of data like demographic profiles, household characteristics and policies related to land resources administration.

To achieve this, it is crucially important to consider multiple sources of information and to acquire temporal, spatial and other non-spatial forms of data. This is due to the fact that landuse attributes are complex and the boundaries between different types of data are quite diffuse (Baulies and Szejwach, 1998). LULC studies have been designed to improve understanding of the human and biophysical forces that shape land use\land cover change. Thus, linking human behavior and social structures to biophysical attributes of the land is a fundamental aspect of LULC research (Baulies and Szejwach, 1998). Land use\land cover study plays an important role in global environmental change and sustainability, including response to climate change, effects on ecosystem structure and function, species and genetic diversity, water and energy balance and agro-ecological potential (Codjoe, 2007).

Generally, determining the effects of land use\land cover change on the earth system depends on an understanding of past landuse practices, current land use\land cover patterns, and projections of future land use\land cover, as affected by human institutions,

population size and distribution, economic development, technology and other factors. LULC assessment is an important step in planning sustainable land management that can help to minimize plant diversity losses and land degradation, especially in developing countries like Ethiopia.

2.6.2. Main causes of landuse\landover change

Landuse is never static; it constantly changes in response to the dynamic interaction between underlying drivers and proximate causes (Lambin, 2005). Globally, landcover today is altered principally by direct human use, such as: agriculture and livestock raising, forest harvesting and management, and urban and suburban construction and development. Besides, natural events such as weather, flooding, fire, climate fluctuations and ecosystem dynamics may also initiate modifications upon landcover. However, recently human activities and social factors were recognized to have a paramount importance for understanding of land use\land cover change. More specifically, it is land clearing for agriculture that has been the most significant process by far and is a process that continues today (Blair and Dockray, 2004).

In general, the causes of land use\land cover change can be categorized in to: proximate driving forces and underlying driving forces. Proximate (direct) causes are immediate actions of local people in order to fulfill their needs from the use of the land (Lambin, 2005). These causes include agricultural expansion, wood extraction, infrastructure expansion and others that change the physical state of landcover. It operates at the local level (individual farms, householders, or communities). Whereas, underlying (indirect or root) driving forces are fundamental socio-economic and political processes that push

proximate causes into immediate action on land use\land cover. Underlying driving forces, i.e. including demographic pressure, economic status, technological and institutional factors, influence land use\land cover in combination rather than as single causations (Blair and Dockray, 2004). Unlike with the proximate causes, it operates at regional and national levels such as districts, provinces, or countries.

2.6.2.1. Population growth as the main driving force for land use\land cover change

The population of Ethiopia has increased from an assumed 16 million around 1950 to about 65 million at the turn of the 20th century (Gete Zeleke and Hurni, 2001) and almost 85 million today (CSA, 2010). According to Sahlu Haile (2004), unsustainable population growth, particularly the Ethiopian highlands which is the most densely populated area, contributed significantly to environmental degradation. Therefore, in the context of the Ethiopian highlands, most studies pinpoint population pressure as one of the major drivers of land use\land cover change through destruction of the vegetation cover, mainly for agricultural expansion (Zerihun Woldu and Mesfin Tadesse, 1990; Amsalu Aklilu *et al*., 2007).

The growing population is one of the most critical drivers of the observed landcover dynamics because the livelihood of almost the entire rural population is dependent on a mixed farming system of crop production and livestock. At the same time, the growing demand for cultivated land and settlement and trees for fuel and construction purposes aggravates the change. Population growth coupled with lack of migration for non-farm employment options makes young adults remain in rural areas, either unemployed or sharing the land with someone else. This leads to further fragmentation of land and

intensification of landuse and subsequent reduction of fallowing practice or abandonment of fallow periods, ultimately expansion of cultivated land into forests, grazing lands and other marginal lands (Girmay Gebresamuel *et al*., 2010).

According to Millennium Ecosystem Assessment (MEA, 2005), human activities are the major drivers for the proximate landuse changes that in turn have myriad impacts on human livelihood too. Various human environment conditions react to and reshape the impacts of drivers differently leading to specific pathways of landuse and forest cover change. It is precisely, this combination that needs to be conceptualized and used as the basis of land change explanation and models.

2.6.3. Major environmental consequences of land use\land cover changes

Land use\land cover change is increasingly recognized as an important driver of environmental change on all spatial and temporal scales. It contributes significantly to earth atmosphere interaction, forest fragmentation and biodiversity loss. Changes in landcover by landuse do not necessarily imply a degradation of the land. However, many shifting landuse patterns, driven by a variety of social causes, result in landcover changes that affect biodiversity, water and radiation budgets, trace gas emissions and other processes that, cumulatively, affect global climate and biosphere (Blair and Dockray, 2004). Generally, LULC change has major impacts on biodiversity, earth climate and hydrology.

2.6.3.1. Impact of land use\land cover change on biodiversity

Biodiversity describes the interrelated existence of the diverse life on earth - animals and plants. It boosts ecosystem productivity whereby every species, no matter how big or small, have an important productivity role to play. However, human activity is vastly

altering the Earth's biodiversity. The loss of biodiversity due to land use\land cover changes takes place at multiple levels (landscape, ecosystem and species), and in multiple dimensions (structure and function) (Turner *et al.*, 2001). A major impact of land use\land cover is on biotic resources. For instance, the annual loss of plant species in tropical Africa is high and about 29 percent of the world land surface was originally under forest cover. Presently, however, it is only a fifth of this original which remains undisturbed (FAO, 2001). The loss of plant biodiversity may lead to the decline of ecosystem integrity and loss of plant genetic resources. Such changes have considerable consequences for the health and resilience of ecosystems and for human welfare.

Ecosystem services are the benefits people obtain from ecosystems. Biodiversity plays an important role in the way ecosystems function and in the many services they provide. Services include nutrient and water cycling, soil formation and retention, resistance against invasive species, pollination of plants, regulation of climate, as well as pest and pollution control by ecosystems. For ecosystem services it matters which species are abundant as well as how many species are present.

However, this valuable resource, biodiversity, is declining rapidly due to factors such as landuse change, climate change, invasive species, overexploitation, and pollution. Such natural or human-induced factors - referred to as drivers - tend to interact and amplify each other. While changes in biodiversity are more clearly linked to direct drivers such as habitat loss, they are also linked to indirect drivers that are at the root of many changes in ecosystems. The main indirect drivers are changes in human population, economic activity, and technology, as well as socio-political and cultural factors. The greatest human impact on biodiversity is the alteration and destruction of habitats, which occurs

mainly through changes in landuse: draining of wetlands, clearing of land for agriculture, felling of forests for timber, and pollution of the environment and fragmentation. In terrestrial ecosystems, the main driver of biodiversity loss is landcover change such as the conversion of forest to agriculture.

Overall, the main factors directly driving biodiversity loss are: habitat change, such as fragmentation of forests; invasive alien species that establish and spread outside their normal distribution; overexploitation of natural resources; and pollution, particularly by excessive fertilizer use leading to excessive levels of nutrients in soil and water (Zerihun Woldu and Mohamed Saleem, 2000). Many drivers affecting biodiversity are stronger today than they were in the past and are also occurring together. Because exposure to one threat often makes a species more susceptible to another, multiple threats may have unexpectedly dramatic impacts on biodiversity (Groombridge and Jenkins, 2000).

2.6.3.2. Forest resource degradation

Forest ecosystems play multiple roles at global as well as local levels: as providers of environmental services to nature in general, humans in particular and as source of economically valued products. Forests especially natural forests are used for various ecological and economic purposes. They could maintain local climate, regulates hydraulic cycle, used as wild life habitats and reduce runoff and soil erosion (FAO, 2010). The economic importance of forests includes supply of raw material for pulp and paper, provision of timber for construction and other activities and used as fuel wood for energy source. Therefore, depletion of this important natural resource will affect other

valuable and life supporting resources like water and soil. In addition to these, the climate change and biodiversity will also be greatly affected by deforestation. It is very strategic and wise approach to avoid other problems by solving this key problem. Throughout history, the fate of the world's forests has strongly reflected the pattern and intensity of landuse by societies. Demand for agricultural land, timber, and other forest products, as well as technological change in agriculture, significantly impacts the mode and rate of transformation of forested areas. Biophysical triggers may also play a role, such as fire dynamics, which are linked to agricultural activities or natural phenomena (FAO, 2001). Deforestation occurs relatively quickly, and in contrast to some other transitions, is easily observable by the human eye. Through the use of remote sensing technologies, large areas can be monitored, and estimates of deforestation can be obtained. According to FAO (2010), the effects of deforestation, forest degradation and forest fires represent a permanent loss of the potential capacity of forest resources to generate economic benefits. Deforestation is a major issue in Ethiopia, since it is one of the main causes of the prevailing land degradation and loss of biodiversity. Tree cutting is a common occurrence which has been taking place for centuries. A long time back in history some parts of Northern Ethiopia, which are today suffering from conditions caused by land degradation, were covered with forests. In present day Ethiopia, however, forests are being destroyed at an alarming rate and the area covered by forests at present is only less than 2.65 percent compared to the estimated 40 percent before one hundred years initial coverage (FAO, 2010).

2.6.4. Common steps in digital image analysis

Digital Image Processing refers to the manipulation and interpretation of digital images,

by a computer system, to prepare an image for display and interpretation and/or to extract useful information from the image. The possible forms of digital image manipulation are literally infinite (Lillesand *et al*., 2006). Digital Image Processing is largely concerned with four basic operations: image rectification and restoration, image transformation, image enhancement, and image classification (Lillesand *et al*., 2006).

a) Pre-processing

Raw satellite image is full of errors and will not be directly utilized for features identification and any applications. It needs some correction before the main data analysis and extraction of information. Pre-processing aims to correct distorted data in order to create more faithful representation of the original scene, this typically involves the initial processing of raw image data to correct for geometric distortions, to calibrate the data radiometrically and to eliminate noise present in the data (Lillesand and Kiefer, 2000).

b) Image enhancement

Image enhancement is used to increase the detailness of the image by assigning the image maximum and minimum brightness values to maximum and minimum display values, and it is done on pixel values, and this makes visual interpretation easier by increasing the visual discrimination between features in a scene and assists the human analyst: False color composite (FCC), spatial re-sampling, etc.

c) Image transformation

According to Rechards (1999), image transformation refers to the derivation of new imagery as a result of some mathematical treatment of the raw image bands. Image

transformation involves processes that are similar in concept to those processes in image enhancement, but unlike image enhancement these process are normally applied on multi channel (band) images.

d) Image classification

Image classification is the process of creating thematic maps from satellite imagery. A thematic map is an informational representation of an image that shows the spatial distribution of a particular theme (Jensen, 2005). The computerized interpretation of images from remote sensors is known as a quantitative analysis due to its ability to identify pixels based on the numerical properties. For quantitative analysis usually different procedures of classification are used (Diday, 1994). Classification is a method that assigns categories to different pixel groups according with the spectral character. There are two main spectrally oriented classification procedures for landcover mapping: unsupervised and supervised classifications.

According to Diday (1994) unsupervised classification is computer-automated and it enables user to specify some parameters that the computer uses to uncover statistical patterns that are inherent in the data. On the other hand, in supervised classification the image analyst supervises the pixel categorization processes by specifying, to the computer algorithm, numerical descriptors of the various landcover types present in a scene (Lillesand and Kiefer, 2000). To do this, representative sample sites of known over type, called training areas are used to create the parametric signatures of each class. Each pixel in the data set is then compared numerically to each category in the interpretation key and labeled with the name of the category.

2.6.5. Change detection methods

Change detection is the process of identifying differences in the state of an object or phenomenon by observing it at different times. Essentially, it involves the ability to quantify temporal effects using multi temporal data sets (Lu *et al*., 2004). Many change detection methods have been developed and used for various applications. For example, there are post-classification comparison, image differencing, image rationing, image regression, principal component analysis. However, they can be broadly divided into: post-classification and spectral change detection approaches (Chen, 2000).

2.6.5.1. Post classification approach

Post classification is among the most widely applied techniques for change detection purpose. Numerous studies have been carried out using post classification approach. In post classification change detection approach two images from different dates are classified and labeled. The area of change is then extracted through the direct comparison of the classification results. Main advantages of post classification include: detailed 'from-to' information. It bypasses the difficulties associated with the analysis of images acquired at different times of year or sensor. The main disadvantage of the post classification approach is the dependency of the landcover change results on the individual classification accuracies. This approach can produce a large number of erroneous change indications since an error on either data gives a false indication of change (Singh, 1989). Therefore, it is imperative that the individual classification be as accurate as possible.

2.6.5.2. Spectral change detection approach

According to Chen (2000), a large number of techniques are in the spectral change

identification category. Spectral change detection techniques rely on the principle that landcover changes result in persistent changes in spectral signature of the affected land surface. These techniques involve the transformation of the two original images into a new single band or multiband image, in which the area of spectral change is highlighted. Most of the spectral change detection techniques are based on some type of image differencing or image rationing. Singh (1989), in his studies identified image differencing as the most accurate change detection technique. This technique is performed by subtracting images from two dates pixel by pixel. Then threshold boundaries between 'change' and no-change pixels are determined for the different image to produce the change map (Singh, 1989).

2.7. Landscape pattern metrics

Landscape pattern metrics quantify landscape structure using categorical maps depicting the composition and configuration of LULC classes on the map. Metrics that focus on composition usually enumerate the number of classes that occur in a landscape, the proportion of each class in a specific area relative to the classes on the entire map, and/or the diversity of classes in an area, but they are not spatially explicit. Metrics that describe landscape configuration quantify characteristics of LULC such as spatial distribution, orientation and shape of individual patches within a landscape (McGarigal *et al.,* 2012).

Usually, landscape pattern metrics can be calculated over three levels for a given landscape: 1) for the overall landscape, 2) for each class or for a specific class and 3) for each patch. The characteristics of individual patches, such as size, shape, and distance to neighboring patches are quantified at the patch-level (a unique value for each patch).

Class-level metrics use these values for all the patches in the same LULC type to return a value for the entire class in the landscape. Landscape-level metrics provide unique values without reference to individual patches or classes, as they aggregate the properties for all the patches in the landscape. In fact, many of the metrics at this level are derived by summing or averaging over all patches or classes (McGarigal and Cushman, 2005; McGarigal *et al.,* 2012).

The software most commonly used for calculating landscape pattern metrics is FRAGSTATS, initially developed by McGarigal and Marks (1995). The most current version of FRAGSTATS (version 4.1) computes metrics over all three levels as noted above (landscape, class and patch-level), and provides options for computing more than 100 landscape pattern metrics. Additionally the cell-level metrics and surface pattern metrics that can be used for the landscape characterization under the gradient paradigm can be computed (McGarigal *et al.,* 2012). McGarigal and Cushman (2005) identified a number of issues that may lead to improper use or interpretation of landscape pattern metrics. Some of the issues in landscape analysis with landscape pattern metrics include misunderstandings about the impacts of scale, grain, and extent, the influences of differing methods used to define patches, and the effects of techniques used to classify and map landcover.

Patch definition

Landscape pattern metrics are also influenced by the method used to define landscape patches. Most often, patches are defined using LULC data derived from remotely-sensed images; therefore, in practical terms, a patch is a contiguous group of pixels classified in

the same LULC class. Two methods have been established to define contiguity. The most common method (called the four-neighbor rule, cardinal directions, or Rook's move), considers a pixel and its four nearest neighboring pixels in the orthogonal directions. If two pixels of the same LULC class share an edge they will be part of the same patch; however, pixels of the same LULC type arrayed along a diagonal will be treated as separate patches. This method is the most frequently used in the landscape ecology literature (Turner *et al.*, 2001). A more graceful method is the eight-neighbor rule (or Queen's move) (McGarigal *et al.,* 2012). This technique considers eight adjacent cells when looking at the neighboring pixels, thus pixels of the same LULC class that are oriented in either the cardinal or diagonal directions will be considered as members of the same patch.

The selection of the method to define patches is crucial in a study, because each method produces different results for metrics such as patch size, perimeter and number of patches, which are fundamental descriptors of a landscape and are used to calculate many other metrics. Additionally, it is important to point out that when using the four-neighbor rule if the landscape is rotated 45°, those pixels that were touching only diagonally could become neighbors, then pixels that were analyzed as being part of different patches, after this rotation could form a unique patch (McGarigal *et al.,* 2009). This could generate different results for the same landscape if analyzed with images that have different orientation. Thus, in this research the eight-neighbor rule was used because it delineates patches in a way that is possibly closer to reality, creating patches that represent the real discontinuities in LULC classes.

As noted above, landscape pattern analysis is almost always based on landcover maps

derived from remotely sensed data. Landscape pattern metrics can be greatly influenced by the methods used for LULC classification and mapping (e.g., number and types of classes). In addition, accuracy of LULC mapping can have important effects on landscape pattern metrics. To ensure consistency and reliability in use of landscape pattern metrics it is necessary to have a high degree of classification accuracy in LULC maps. Wickham *et al*. (1997), using landcover data derived from Landsat TM satellite imagery, tested the sensitivity to LULC misclassification and the effects of land-cover composition on landscape pattern metrics such as average patch compaction, fractal dimension and contagion. Their results suggest that differences in land-cover composition need to be about 5% greater than the misclassification rate to be confident that differences in landscape pattern metrics are not due to misclassification in the LULC maps. Shao and Wu (2008), using similar Landsat TM-derived LULC data, found that the variation in the values of landscape pattern metrics generally decreases when LULC classification accuracy increases, but the sensitivity of each landscape pattern metric to the LULC classification accuracy is variable.

Although there is still much to be observed about effects of LULC classification on landscape pattern analysis, it is clearly imperative to employ a LULC classification scheme that provides a reliable landscape analysis according to the objectives of the study. It is also very important to report the LULC classification accuracy of the dataset used in each study to provide an assessment of the reliability of the landscape analysis.

2.7.1. Selection of landscape pattern metrics

Landscape metrics can assist in quantifying the fragmentation process and in assessing the biological integrity of the remaining forest (McGarigal and Marks, 1995). Landscape

metrics are algorithms that quantify specific spatial characteristics of patches, classes of patches, or entire landscape mosaics. An overabundance of metrics has been developed to quantify categorical map patterns. These metrics fall into two general categories: those that quantify the *composition* of the map without reference to spatial attributes, and those that quantify the *spatial configuration* of the map, requiring spatial information for their calculation (Turner *et al.*, 2001).

Although there are hundreds of metrics available, addressing several question using landscape metrics in assessment efforts is difficult depending on: (1) What are the objectives of the study (i.e. are the selected metrics related to the ecological processes being examined)? (2) What is the behavior of the metrics over a range of landscape configurations? (3) What are the effects of scale on the metrics? And (4) are the metrics correlated or redundant (Turner *et al.,* 2001). The fragmentation index is a useful tool for measuring the level of disturbance and vulnerability of forests. During the last three decades several studies, have demonstrated that many landscape pattern metrics are highly correlated with one another and, thus, provide essentially redundant information. As a consequence, much research has been directed towards identification of a small set of key independent measures that can be used to characterize landscape structure (Cushman *et al*., 2010).

Li and Reynolds (1994) proposed that five attributes should suffice to describe landscape pattern as depicted in LULC maps: (1) the number of classes that occur in an area of study, (2) the percentage of the area occupied by each class, (3) the spatial arrangement of patches, (4) patch shape, and (5) contrast between neighboring patches.

According to Cushman *et al*. (2010), it is not necessary to employ all landscape pattern

metrics to characterize a landscape, but the selection of the set of metrics to use is an important decision in a study. Although previous studies demonstrate that landscape structure can be characterized using a few key metrics, they are not in complete agreement on the minimum set of metrics that are most useful. This is likely due to several factors, including the fact that the studies were undertaken in different landscapes, used different landcover map sources with different LULC classifications, and used different grain and scales. However, components such as LULC diversity, patch size, shape and complexity, landscape texture, inter-patch distance, and patch interspersion appear frequently among the components that explain most of the variance in landscape patterns.

According to McGarigal *et al.* (2012), all landscape metrics need the user to define some parameters during the landscape analysis, including its extent and grain and the patches that comprise it, additional input parameters such as edge effect distance, edge contrast weights, and search distance before any of these metrics can be computed. Thus it is incumbent on the investigator to determine those parameters as they are deemed necessary for an application under consideration. The identification of patches reflects a minimum mapping unit that is chosen for practical or technical reasons and not for ecological reasons. Subjectivity in defining patches, surface pattern techniques can provide an objective means to help determine the scale of patchiness (McGarigal *et al.,* 2009).

Additionally, the format (raster versus vector) and scale (grain and extent) of the data can have a profound influence on the value of many metrics (McGarigal and Cushman, 2005). Furthermore, Li and Reynolds (1994) stated that, failure to understand the

theoretical behavior of the metric can lead to erroneous interpretations. Thus McGarigal *et al.* (2012) advised to select a metric representing a landscape pattern in a manner and at a scale ecologically meaningful to the phenomenon under consideration to draw sound conclusions there from.

2.7.2. Habitat fragmentation and the spatial characteristics of habitat patches

Fragmentation is a system level phenomenon. To estimate the effects of fragmentation, one should have a firsthand knowledge of how the entire system operates. When a habitat becomes fragmented and surrounded by another habitat this generally causes an increase in pressure at habitat transitions, often referred to as an edge effect. Fragmentation of an intact forest results in increased amount of edge. This can have far reaching consequences for the native biota because generalized species that are adapted to the edge habitats can more easily penetrate forest remnants (Ewers and Didham, 2006).

A fragmented forest habitat is reduced in size but tends to retain ecological characteristics and organisms of the forest. Up to a point, in fragmentation, no species are lost. However, as the process continues, the remaining area is reduced to a critical size below which it will not support many of the original species and a number of them would disappear (Cushman *et al*., 2010). Different components of an ecosystem can respond in different ways to habitat fragmentation (McGarigal and Cushman, 2005). The effects seen among populations can be hidden at the level of aggregated community variables. Therefore understanding the consequences of habitat fragmentation requires documenting the system attributes that do not change as well as those that do. It has now become increasingly necessary to understand the consequences of the destruction and fragmentation of natural habitats. Habitat fragmentation can influence the entire suite of

ecological processes, from individual behavior through population dynamics to ecosystem fluxes (Cushman *et al*., 2010).

Fragmentation requires a landscape-scale focus of the composition, configuration and relationship of patches within a landscape. Ewers and Didham (2006) suggest that the effects of fragmentation can be grouped into five categories that provide a better perspective of the spatial characteristics of habitat patches: structure of the matrix, patch area, patches shape, isolation/connectivity and edge effects. Ewers and Didham (2006) explained that, these five categories are not independent of one another; not only do they interact with and possibly enhance the effects from another category but there are additional factors that may confound the effects of these processes over large scales.

Smaller and isolated habitats usually have higher extinction rates and can support smaller local populations. Saunders *et al.* (1991) found that smaller fragments undergo more biogeographical changes than larger fragments. This is due to smaller patches having reduced habitat area, which can have negative effects on species richness, trophic structure, dispersal and colonization success and genetic diversity (Turner *et al.*, 2001). Harrison and Bruna (1999) conducted a review of fragmentation studies and concluded that habitat fragments support few habitat specialists and more generalists and usually have altered biodiversity. Smaller habitat fragments cannot support equivalent species diversity compared to larger habitats and the loss of keystone species may cause altered ecological interactions and decreases in species diversity at lower trophic levels (Harrison and Bruna, 1999).

Generally, many researchers believe that there are “landscape thresholds” in fragmented landscapes which means that there are critical amounts of habitat below which the

success of a certain species is unlikely (Ewars and Didham, 2006). This critical value is dependent on the individual species and its sensitivity to fragmentation in addition to the spatial arrangement of habitat patches. Saunders *et al.* (1991) developed an area-based model to describe the temporal dynamics of species abundances in forest islands and found that changes in species richness occur when the area of a forest is below a certain threshold (but this threshold is different for edge and interior species). Therefore, to describe the effects of habitat fragmentation it is essential to have better perspective of the spatial characteristics of habitat patches.

2.8. Importance, threats and management practices in Hugumburda-Gratkhassu forest

Hugumburda-Gratkhassu NFPA is wealthy in plant diversity composition and is home to many rare and endemic species (Leul Kidane *et al*., 2010; Ermias Aynekulu, 2011). But, it is a fragile ecosystem because of its unique geological formation and historical events. Annoyance with the vegetation in Hugumburda-Gratkhassu NFPA has almost permanent negative effect, affecting both the uniqueness of the habitat and inhabitants. Earlier and present disturbances changed much of the natural features of the ecosystem. Harsh climatic conditions, rugged nature of the landscape and altitudinal effects intensify the effects of anthropogenic disturbances. This very important ecosystem entails protection measures to stop threats so as to return the valuable natural resources.

2.8.1. Importance of the forest priority area

Hugumburda-Gratkhassu NFPA is one of the national forest priority areas of the country with its own unique collection of biodiversity. Besides to being home to many rare and endemic species, the priority area provides socio economic and cultural benefits, through

the provision of direct livelihood support from illegal collection of fire wood and grazing, water and economic benefits to surrounding inhabitants.

The forest priority area is used as a water catchment to the surrounding lowlands. Protection of the resources and ecosystem processes in the area has also a climate stabilizing effect on the eco-region, including the priority area and the surrounding degraded areas. Vegetation cover has a major role in regulating soil temperature as well as water retention and, hence, downstream hydrological dynamics. Vegetation cover is also a monitor of local and global climate change through carbon sequestration, moisture retention and temperature regulation. Hugumburda-Gratkhassu NFPA, as part of a highland ecosystem is the most susceptible to the earliest effects of global and local climate warming. Thus, the priority area provides an ideal and important reference site for monitoring climate change and validating climate prediction model before effects are observed more widely. This early warning system is also significant for devising alleviation measures to natural resources and ecosystem processes that are likely to be affected by global warming.

2.8.2. Threats to the forest priority area

Agricultural expansion, livestock overstocking, over-harvesting of natural resources and settlement have been identified as high level threats in Hugumburda-Gratkhassu NFPA and require immediate management action. The northern highlands are highly fragile due to the extreme climatic conditions. According to Hedberg (1964), survival in these ecosystems is with certain adaptation. Due to exhaustive human pressure most of the resources are now at risk. Degradation of natural resources, mainly vegetation and soils, is common. Rapid population growth resulted in scarcity of farmland, shortage of fallow

periods, and deforestation even in the last remnants of natural forests. Soil degradation has become a major threat to the ecosystem. Population pressure has pushed the farmers onto steeper slopes, which can only give yields for a few years before the soil is washed away further threatening the natural fragile environment (Ensermu Kelbessa *et al,* 1992). The loss in vegetation and soils obviously implies great losses in valuable genetic material (Tewolde Berhan Gebre Egziabher, 1991). As a result of human pressure dry afromontane remnant vegetation are scattered and restricted in inaccessible areas. If the profound exploitation of the area continues, resources will eventually be exhausted. Therefore, awareness is needed to halt further threats and rate of destruction in the area.

2.8.3. Management practices

Forest Priority Areas have great administrative advantages that limit human interferences and facilitate the conservation role to be maintained. In detection of the threats and the opportunities for conservation and development, protection efforts have been taking place. These inclued: sustainable and environmentally friendly livelihood diversification, soil conservation activities such as terracing are practiced, reduction in grazing and cultivation pressure in and around the forest area. But, these conservation efforts are far less compared the rate of degradation and habitat loss and fragmentation of the forest. The challenge is to apply the strategies and action plans of the management plan into programs of development and conservation that benefit human and non human resources (SFCDD, 1997). Therefore, to resolve these confront, it is critical to facilitate dialogue among the government decision makers' and communities in relation to the relevance of natural resource strategy.

CHAPTER THREE

3. MATERIALS AND METHODS

3.1. Description of the study area

3.1.1. The study area

This study was carried out in Hugumburda-Gratkhassu NFPA (Fig. 1). It is located between 12^0 22' and 12^0 42'N latitude, 39^0 28' and 39^0 40' E longitude, in the southern Zone of Tigray Regional State at about 600 km north of Addis Ababa or some 160 km south of Mekele, the capital of Tigray Regional State.

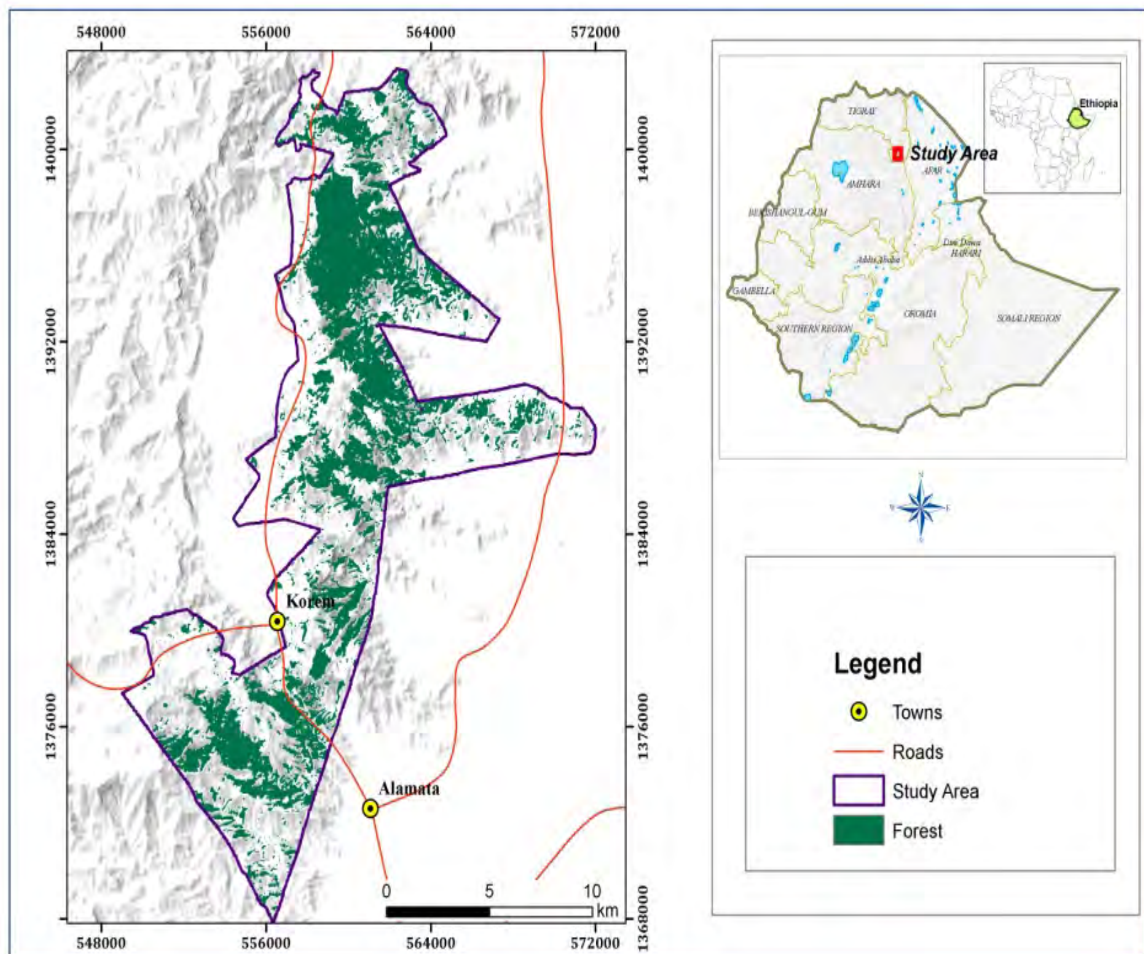

Fig. 1 Map of Ethiopia showing Regional States and Hugumburda-Gratkhassu National Forest Priority Area

Topographically, the study sites are characterized by plane to steep slopes dissected by stream incisions. Whereas little part of the study area is flat with little variation in altitude but most parts of the study sites were covered with shrubland, most of the areas have highly rugged topography which is unsuitable for agriculture. Digital Elevation Model (DEM) of the study area (Fig. 2) has shown that, the study area lies in an elevational range from 1500 m to 3500 m above sea level.

3.1.2. Geology and soils

The study area has a diverse geological formation, which plays a major role in soil variability. Geologically, the majority of the landscape of the study area originated from tertiary basalt. Alkali-alluvial basalt and tuff are wide spread in the area. In eroded areas, exposed rocks are found as boulders and rock outcrops. Spheriodally weathered rocks are observable in most parts of the NFPA. Foot slopes and valley bottoms are covered with colluvial and/or alluvial sediments transported from upper slopes. Two groups of soils are recognized in the forest area, *leptosol* and *regosols* in the uplands and *fluvisols* on the valleys and low lands. Generally, the uplands of the Forest area have extremely shallow to shallow soils while relatively deep soils occur in gently sloping areas and valley bottoms. The soils of the NFPA have low to medium erodability. Water erosion is a serious problem on steep slopes and valley sides in which numerous drainage ways and gulleys are already formed. This calls for the construction of physical conservation measures such as terraces, micro basins and check-dams which are indispensable to reduce soil erosion and landslides on the area (Kebede Agize and Admasu Bizuneh, 1998).

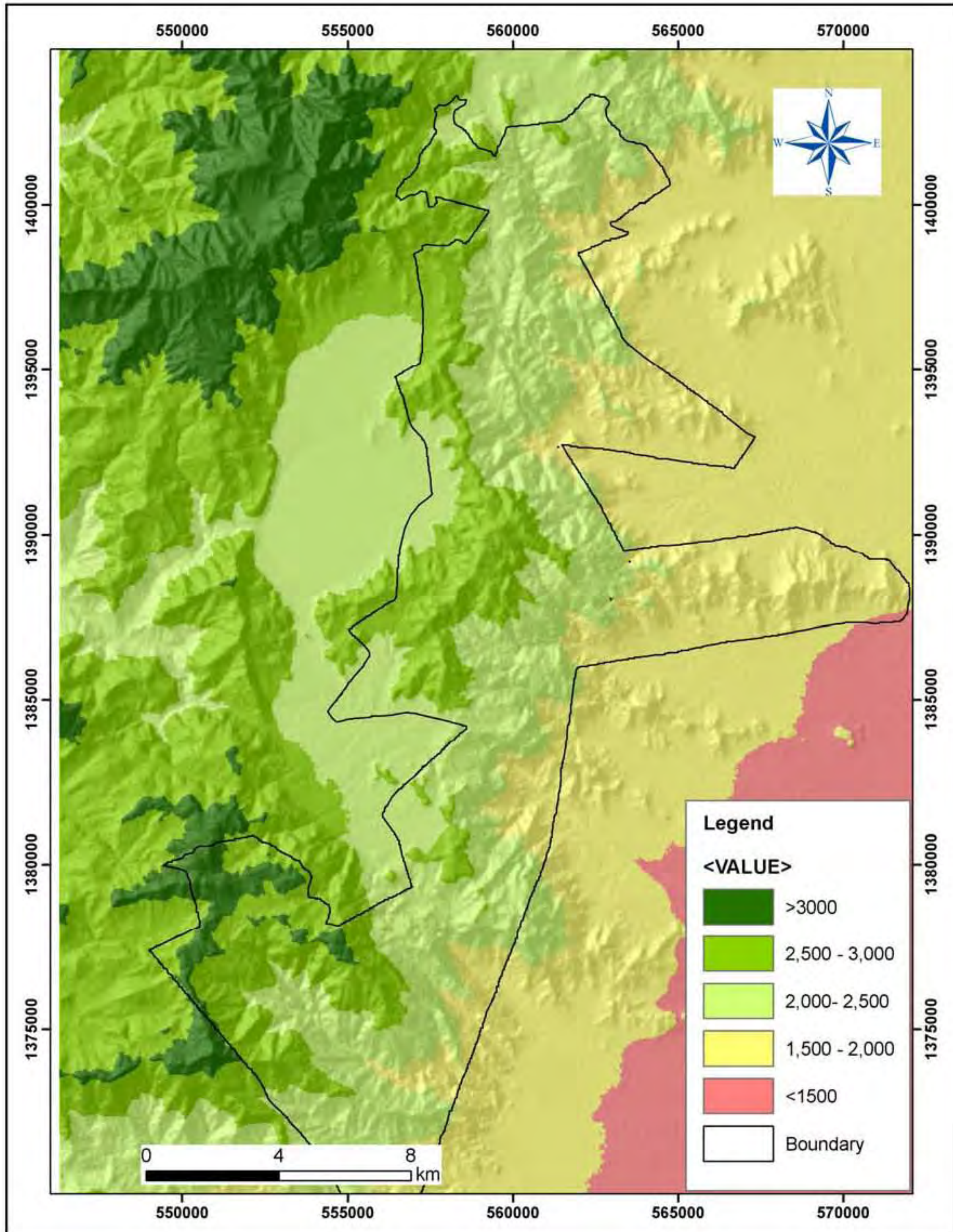

Fig. 2 Digital elevation map of Hugumburda-Gratkhassu National Forest Priority Area

3.1.3. Climate

White (1983) noted that it is not possible to express climate satisfactorily by figures or formulae, however complex, because of the seasonal rhythm of the most important factors, and variation from one year to another. By contrast, diagrams, although far from perfect, can be used to summarize an enormous amount of relevant information, and permit rapid visual comparison of different stations and different vegetation types or different chorological and climatic regions.

There are two meteorological stations (Alamata and Korem) which are near the study area. Thirty five years (1978 – 2013) of meteorological data from these stations were acquired from NMSA, to describe the climate of the study area. Analysis of the meteorological data showed that the mean annual temperature for Alamata was 21.9 ^{0}C and the mean minimum and maximum were 12.1 and 33.5 ^{0}C respectively (Fig. 3a). On the other hand, the mean annual temperature of Korem station was 15.3 ^{0}C with a mean minimum of 5.4 ^{0}C and a mean maximum of 24.7 ^{0}C (Fig. 3b). The hottest months are April and June, while coldness is from September to November. The mean annual rainfall for Alamata and Korem are 705 and 986 mm respectively, which varied greatly from year to year. Generally the study area has bimodal rainfall pattern, with low rainfall from February to May and the main rainy season (June – September) (Fig. 3).

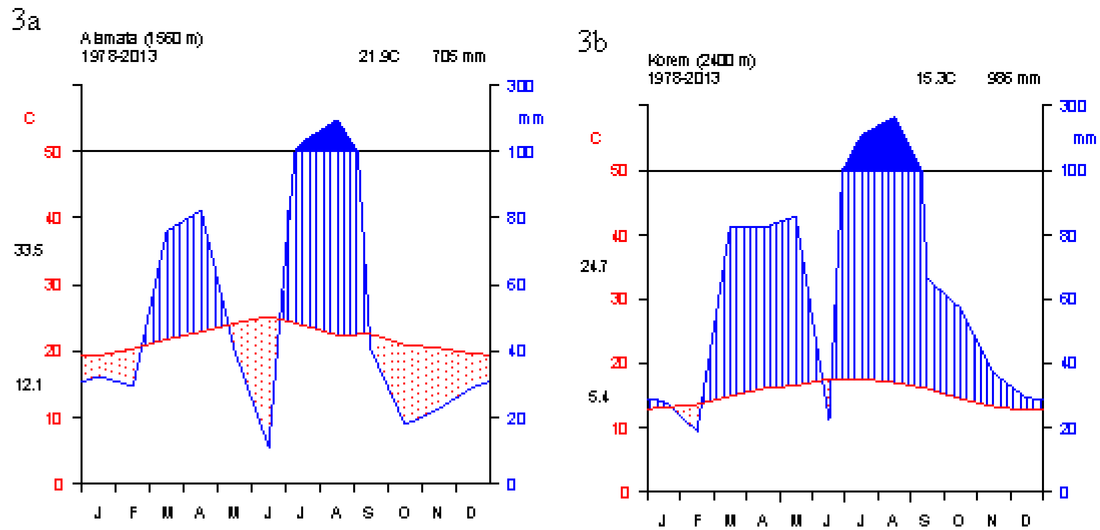

Fig. 3 Climatic diagram, 3a denotes Alamata and 3b denotes Korem stations. Data source: National Meteorological Service Agency, Ethiopia

3.1.4. Vegetation

Based on the broad categorization of the forests and forest trees of Ethiopia (Zerihun Woldu, 1999; Friis *et al*., 2011), Hugumburda-Gratkhassu Forest is categorized as dry evergreen Afromontane and grassland complex forest, which is characterized by dry climate (annual mean precipitation less than 1000 mm) and with *Juniperus procera* in the canopy and *Olea europaea* ssp. *cuspidata* as dominant tree species. The study area provides diverse habitats for plants and it is the only more or less intact large forest with many locally rare tree species from the pre-disturbance period. *Podocarpus falcatus* and *Hagenia abyssinica,* which are rare in other parts of northeastern Ethiopia, are found in this Forest. Formerly, the area was a dense forest with *Juniperus procera, Podocarpus falcatus, Olea europaea* ssp. *cuspidata, Acacia abyssinica, Prunus africana, Pittosporum viridiflorum, Olinia rochetiana* and *Rhus natalensis*. According to Zenebe Gebre-Egziabher *et al.* (1998) and information obtained from local informants, the natural forest

was exploited by the Italian concessionary named Montu Doro who installed sawmills at Hugumburda in 1950 with the permission of the then governor of Welo Province.

The major timber products were mainly extracted from *Juniperus procera* and *Podocarpus falcatus*. Further, this activity has resulted in indiscriminate clearing of the Forest until 1965 when it was demarcated as state forest. This high rate of the clearing of the forest, logging coupled with shifting cultivation on logged, open areas have put the forest into its present state where only bushes, shrubs and scrubs of secondary forest are remaining. The 1984/85-drought occurrence in the area was also noted as one of the most critical periods in making the Forest the last resort for the livelihood of the rural households.

The Forest was officially put under the auspices of State Forestry Agency in 1965 (SFCDD, 1997). Then, in 1981 the area was identified as one of the National Forest Priority Areas (NFPAs). Boundary demarcation, which is the basis for the current management of the Forest, was undertaken in 1993. Based on this demarcation, the project covers a total area of 21, 654.24 ha. Out of this 532.75 ha is plantation forest whereas the rest contains disturbed natural high forest, bushes, shrubs, agricultural plots and settlement area.

3.1.5. Population and economic activities

There are 26,889 households within and around the Forest boundary (CSA, 2007); out of which 5,496 households are fully within the Forest area and the rest (21,393) reside in the periphery of the Forest. In general, the male population is a little more than the female population, which is 53.8%. The average population size is six persons per households with an average density of 138-persons/square kilometer (Zenebe GebreEgziabher *et al*.,

1998; Gebrekidan Teklu, 2000). Children between the ages of 7 and 15 are involved in important production activities such as looking after cattle or sheep and in safeguarding crops from birds and animals. Hence, assuming that the ages between 8 and 60 as economically active, 62.7% of the population can be considered as potential labor force in the study area.

With regards to the land tenure system, all rural lands are collectively owned by the state. Distribution of land was done based on the fertility status of the soil (fertile, less fertile, moderately fertile), and the functional category of the plot (field plot and backyard plot). Concerning the land holding size in the area, it is estimated that a household has 0.85 ha on average (Gebrekidan Teklu, 2000). Crops are produced in rainfed agriculture mainly for subsistence purpose. The major crops growing in the area are teff, maize, barley, wheat, beans, peas, lentils, sorghum and pepper. Due to the prevalence of a bimodal rainfall pattern, the system of double cropping (production of two crops on the same plots of land in a year) is practiced both on the highlands and lowlands. Manuring and crop rotation are practiced very rarely. Diversion of floods and water harvesting mechanisms are also practiced, especially in the lowlands where rainfall is relatively scarce.

Livestock husbandry is an integral part of the mixed farming system in the study area. Among the various livestock reared, cattle constitute the highest proportion (61.5%) (Zenebe GebreEgziabher *et al*., 1998). Other important domestic animals next to cattle in the order of quantity and importance are sheep (20.3%) and goats (10.5%). Donkeys, horse, mules and camels are the most important means of transport for the rural

households. In general, the livelihood of the livestock family depends on natural vegetation and crop residues. The use of tree parts as fodder is very limited in the study area. “Hizaeti”and “Mewaya” or “Mewcha” are the two most practiced traditional systems of grazing in the area. These are common pool natural resources management systems or a common grazing place designated particularly for oxen and cattle other than oxen respectively.

3.2. Materials

During data collection, Garmin GPS 60, Compass, Sunnto Clinometer, Digital camera, DBH calliper, Rope, metal frame, measuring tape were used in this study. Plant press was used to dry and pack plant specimens and Auger for soil sample collections.

3.3. Methods

3.3.1. Reconnaissance survey

A reconnaissance survey of the study area was made from May to June 2011 in order to obtain an impression about the internal variation in site condition and physiognomy of the vegetation. During the survey, study sites were selected, sampling methods were determined, accessibility of the study patches was assessed, and altitudinal gradient and topography of the forest patches were inspected. Detailed field protocols and sampling methods were further developed following this reconniascence survey.

3.3.2. Floristic study

After reconnaissance survey, data collection was made from October to December, 2011; from April to June, 2012; from November to January, 2013 and from June to July, 2013. A stratified preferential sampling design technique with flexible systematic model was

used for data collection. Flexible systematic model is a special technique based on the principle of optimum allocation of samples (Smartt, 1978). This method may be viewed as a special form of stratified sampling, since samples are allocated on the basis of some predetermined criteria (Kent and Coker, 1992). Local variation of floristic diversity was used as a criterion for sampling. The study area was first divided up into two (Gratkhassu and Hugumburda) forest remnants based on physiognomy, before samples were chosen on the basis of patch size and variation in the landscape. To take representative samples, 34 patches have been selected from Gratkhassu and Hugumburda forest remnants. In each patch, sampling was carried out based on patch size and local diversity of vegetation cover. In fragments where patch size, diversity and rate of change of vegetation cover was higher more samples were taken whereas in areas with small patch size and little change in vegetation heterogeneity small number of sample plots were laid out.

Within each patch 2-5 quadrats were taken from the central portions of the patches and at least 30 m from the edge and covered variations in aspect and slope. Generally, the quadrats for sampling were selected in such a manner that each is a representative of the vegetation of which it is part and that each plot sampled therein yields a more or less typical description of that vegetation in terms of both floristic composition and structure.

Data on vegetation and environmental parameters were gathered from 77 quadrats (37 from Gratkhassu and 40 from Hugumburda) (Fig. 4), of 20 m x 20 m (400 m^2) size (Mueller-Dombois and Ellenberg, 1974; Kent and Coker, 1992; McCune and Grace, 2002). All vascular plant species were recorded and collected. To produce species

diversity/richness of the study area, species occurring outside of the quadrats, but inside the forest were also recorded only as ‘present’, but they were not used in the subsequent data analysis.

Species richness and the presence or absence of herbaceous plants and seedlings and saplings of all woody plants (shrub, liana and trees) were recorded in a 2 m x 2 m (4 m^2) subplot inside each main quadrat from five points, one at each corner and one at the centre as used by Tadesse Woldemariam (2003) and Getinet Masresha (2014) (Appendix 7).

Cover/abundance (ground cover) for each plant species was estimated following the procedure of Braun-Blanquet (Braun-Blanquet, 1965; Kent and Coker, 1992), to determine the plant community types. The percent cover values, which are visually estimated in the field, were later converted into a modified 1-9 Braun-Blanquet scale (van der Maarel, 1979):

1 = rare, generally one individual with less than 5% cover of the total plot area,

2 = occasional or sporadic with less than 5% cover of the total plot area,

3 = abundant, with less than 5% cover of total plot area,

4 = very abundant, with less than 5% cover of the total plot area,

5 = 5-12% cover of the total plot area,

6 = 12.5 -25% cover of total plot area,

7 = 25-50% cover of the total plot area,

8 = 50-75% cover of the total plot area, and

9 = 75-100% cover of the total plot area

During data collection period, plant specimens were collected, pressed, dried properly and brought to the National Herbarium (ETH), Addis Ababa University for identification and verification. The identification was done using the Flora of Ethiopia and Eritrea (FEE) and by comparing with the authentic specimens in the ETH. Plant nomenclature followed FEE. The accuracy of identification was checked and confirmed.

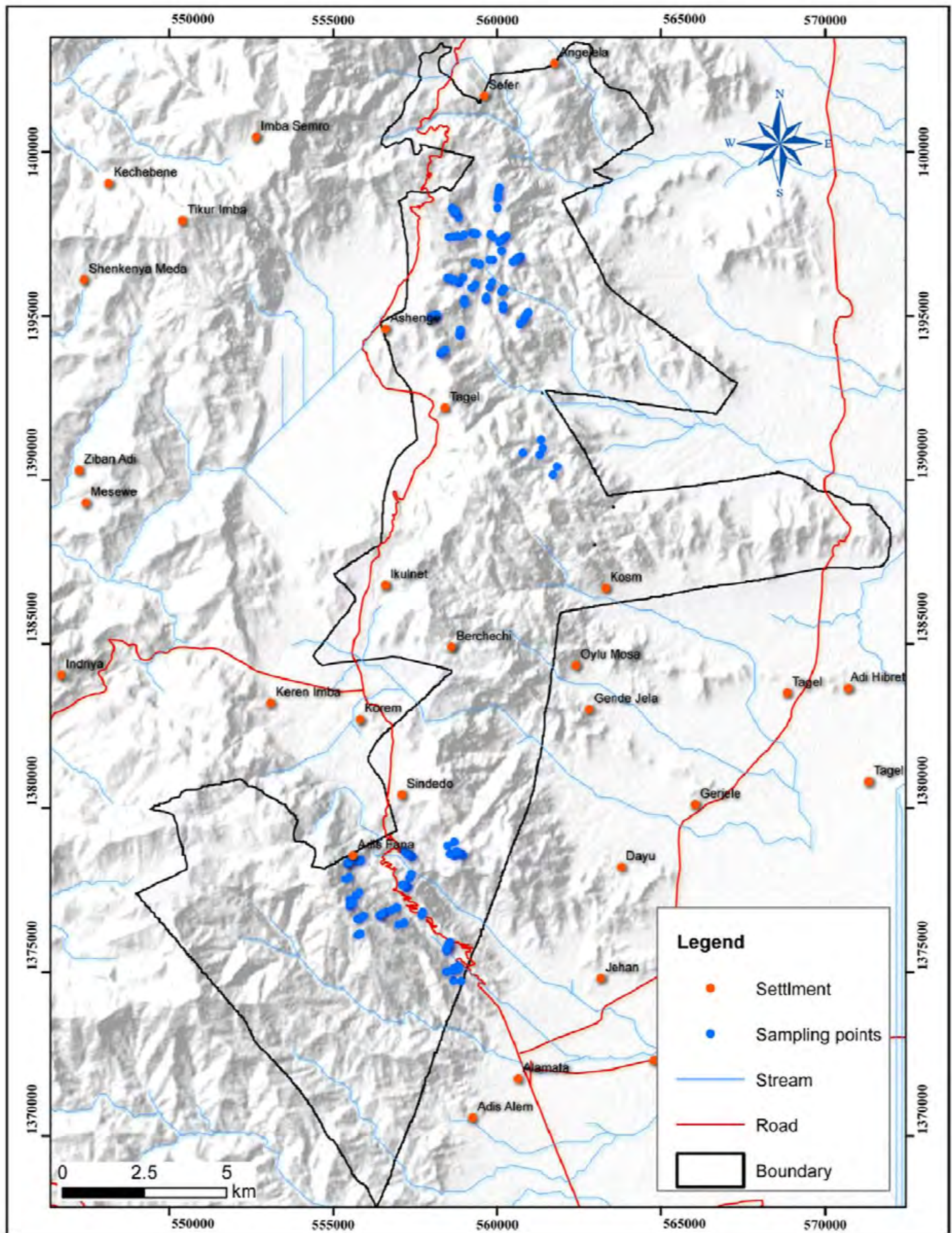

Fig. 4 Distribution of sampling plots (blue circles) in the study area

3.3.3. Population structure data collection

In each quadrat, DBH and height were measured for all woody plants (Appendix 6). DBH was measured with calliper (at 1.3 m above ground) for trees and shrubs having DBH ≥ 2 cm. Plants with multiple stems below 1.3 m height were treated as a single individual and the DBH of all the stems were taken and then the average of the diameter was used for basal area calculation. If a tree was buttressed and abnormal at 1.3 m, the diameter was measured just above the buttress where the stem assumes near cylindrical shape. Height was measured using Clinometer and a pole marked at 0.5 m intervals and 4 m long. For the purpose of the study, seedlings, saplings and mature trees/shrubs were categorized as seedling (height < 1 m), sapling (height between 1 m and 2 m) and tree/shrub (height > 2 m) following Feyera Senbeta and Demel Teketay (2001).

3.3.4. Environmental and disturbance data collection

Topographic data

Environmental data on topographic attributes were gathered from each quadrat (Appendix 4). Altitude and geographical coordinates were measured using Garmin GPS 60 in the middle of the main quadrat. Slope (in %) was measured using clinometers, and aspect was determined using Suunnto compass. As a possible indicator of total solar energy, aspect was codified according to Zerihun Woldu *et al.* (1989), where N=0, E=2, S=4, W=2.5, NE=1, SE=3, SW=3.3 and NW=1.3.

Soil data

Soil samples were collected from 77 quadrats for soil analysis. From each quadrat, five soil samples (from a depth of 20 cm) were collected from each corner and center and

mixed to produce a composite soil sample, each weighing about 1 kg. The soil samples were air dried by spreading on plastic trays, crushed and sieved with a mesh size of 2 mm. Then, measurements and analysis for pH, Electric conductivity and soil texture were done in the soil laboratory of AAU following standard procedures as described in section 3.4.6.

Disturbance

Grazing intensity was estimated by assessing ground vegetation height, proportion of bare ground and abundance of faecal matter droppings by herbivores, and by observing evidence of trampling, browsing and animal trails in each quadrat. The grazing rate was classified visually as ‘no or slightly grazed (0)’, ‘intermediately grazed (1)’, ‘intensively grazed (2)’, and ‘over grazed (3)’ as used by Getinet Masresha (2014). The state of human interference at each stand was estimated following Gebremedhen Hadera (2000) and Kumlachew Yeshitla and Tamrat Bekele (2002). A 0-3 subjective scale was taken into consideration to record the presence or absence of stumps, logs and signs of fuelwood collection. Therefore, the magnitude of the impact was quantified as follows: 0=nil; 1= low; 2= moderate; and 3=heavy.

3.3.5. Vegetation and environmental data collection for edge effect study

To describe plant species composition and population structure of the edge (0–100 m), intermediate (100–200 m), and interior (>200 m) habitats of the forest, 17 patches (9 from Gratkhassu and 8 from Hugumburda) (Appendix 8) were systematically selected along the forest fragments. In order to minimize variability, the 17 patches were selected to have similar matrix vegetation (i.e. forest edges adjacent to grazing area only). The edge

of each forest patch was defined by the limit of tree canopy and adjacent to grazing area only following Harper *et al.* (2005) and Chen *et al.* (1990). Within each patch one edge gradient (transect) was located along the longest axis of the polygon of the patch, 220 m from each edge (0 m), into the inner forest (“inner edge”; Thomas *et al*., 1979) and establishing sampling points for vegetation and environmental variables at 0, 50, 100, 150 and 200 m from the edge, at an interval of 50 m from each other. In total, 83 quadrats of 20 m × 20 m size were systematically established at every sampling point. From each quadrat, vegetation and environmental data were collected the same way to the scheme used under 3.3.2 - 3.3.4.

Vegetation data collected for this study include floristic list, density, frequency and DBH of woody species. Based on their frequency of occurrence, the recorded plant species were categorized into forest specialists and drought resistant species. Canopy cover of each quadrat was visually estimated and recorded on 1 to 3 scale bases. When the canopy cover was estimated to be < 33% (open canopy) 1 was given, 2 was given when it was estimated between 33 and 66 per cent (medium shade) and 3 when the canopy cover was estimated to be > 66% (high shade canopy) following Chen and Franklin (1992). Environmental data include altitude, slope, aspect and location of each quadrat. Disturbance measurements were collected in the same way to the scheme used under 3.3.4. In addition, the number of snags and logs were also recorded from each quadrat.

3.3.6. Land use\land cover change data acquisition and processing

One important method of understanding ecological dynamics, such as natural and human disturbances, ecological succession and recovery from previous disturbances, is the analysis of changing landscape patterns (Turner *et al.,* 2001). To quantify the magnitude and rate of the change as well as the dynamics of major land use\land cover types in the study area, three orthorectified satellite images from Land sat MSS for 1973 and TM for 1986, were obtained from Ethiopian Mapping Authority (EMA) and TM for 2011 was obtained from USGS (Table 1). These sources of information were used to analyze landcover\landuse changes over the years for the study area.

The data from these Landsat images were processed by ERDAS EMAGINE 9.2 software and spatial analysis, interpolation and change detection were done by ENVI 4.7 software (Table 1). A blend of steps and procedures were developed to analyze, map, interpret and quantify the available data sets (Fig. 5). These steps are critical to the success of multitemporal remote sensing studies (Elmore *et al.*, 2000).

Table 1 Data sources and materials for land use\land cover

I. Satellite Image Data					
	Sensor	Path	Row	Spatial Resolution	Date of Acquisitions
Landsat 1	MSS	181	51	60 X 60 meter	1973
Landsat 5	TM	168	51	30 X 30 meter	1986
Landsat 5	TM	168	51	30 X 30 meter	2011
II. Softwares Used					
ERDAS Imagine 9.2 and ENVI 4.7 were used for the development of land use\land cover classes and subsequently for the change detection analysis					
ArcGIS 9.3 used for GIS analysis and mapping purpose					

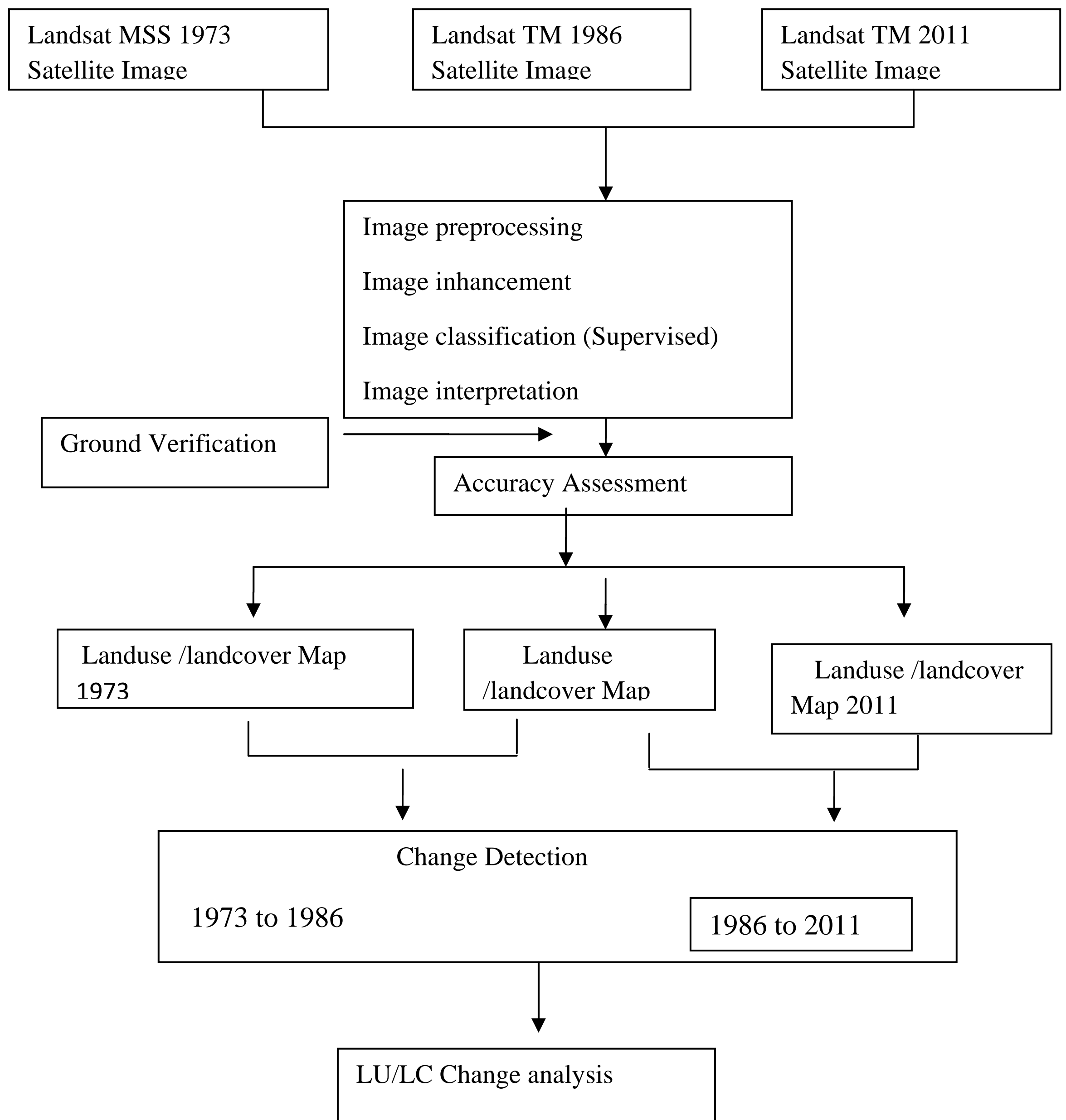

Fig. 5 Flow chart showing the general methodology of land use\land cover analysis

Field observation was carried out to obtain Ground Control Points (GCPs) for georeferencing the images, to understand the features of the different landcover classes, support visual interpretation of the images and select reference areas. Ninety five reference points were taken using GPS receiver for ground truth verifications. At every

reference point, co-ordinates and the current human activity evidences on each land use\land cover were documented.

Based on the collected training sample points, visual and digital interpretations of the satellite images, six LULC categories (Table 2) were distinguished, so that it was possible to investigate changes that occurred since 1973. In relation to this, the year 2011 land use\land cover classification result was evaluated by employing accuracy assessment technique using ERDAS Imagine 9.2 software to investigate how the result reflects the reality on the ground.

Table 2 Descriptions of landcover categories for change detection between 1973 and 1986, and 1986 and 2011

Land use\land cover type	**Land use\land cover description**
Forest	Areas covered by trees forming closed or nearly closed canopies, more than 10% canopy cover (FAO, 2010).
Shrub land	Land covered by small trees, bushes, and shrubs, less dense than forests.
Grassland	Represents open land having a ground story or vegetation cover in which grasses are dominant life forms.
Cultivated land	Areas of land ploughed/prepared for growing various crops. This category includes areas currently under crop, fallow and land under preparation.
Settlement	Area of land within and around the forest that is populated with permanent residents.
Bare land	Area of land with in and around the forest that have no vegetation cover, mainly covered by bare rock.

3.3.7. Fragmentation and patch area computation

3.3.7.1. Metrics selection and computation

Landscapes often depict a mosaic of different patches. Thus, landscapes are usually characterized by the structure and composition of constituent patches besides their spatial pattern or configuration. Information on the structure, composition and configuration of patches and spatial pattern of varied landscapes has been widely assessed using FRAGSTATS, a spatial pattern analysis program for quantifying landscape features (McGarigal and Marks, 1995; McGarigal *et al.*, 2012).

The LULC classes of 1973, 1986 and 2011 were used for the purpose of analyzing fragmentation at patch, class and landscape level. Selecting metrics for a given study involves a number of considerations. Many landscape pattern metrics are highly correlated with one another and, thus, provide redundant information (Li and Reynolds, 1994). Therefore, identification and selection of a small set of key independent measures that can be used to characterize landscape structure is an important decision in a study of habitat fragmentation (Cushman *et al*., 2008). Taking these into consideration, in this research, five patch level metrics, nine class level metrics and nine landscape level metrics (Tables 3 and 25) were selected to quantify and examine the spatio-temporal changes in landscape composition and configuration between 1973 and 2011 in the study area. These metrics are grouped into three components according to the aspect of landscape pattern measured as follows: 1) area and shape metrics 2) aggregation metrics and 3) diversity metrics. Within each of these groups, metrics are further grouped into patch, class and landscape level metrics. The description of each metric and equations for their calculation presented below follows McGarigal *et al*. (2012):

1. Area and shape metrics

This group of metrics represents the size and shape of patches on the landscape. In this study they are measured by the area of patches, total class area and percentage of landscape, largest patch index, core area and shape index, and fractal dimension.

A) Area metrics

i) Patch level metrics

a) Area: equals the area (m^2) of the patch, divided by 10,000 (to convert to hectares).

$$AREA = a_{ij}\left(\frac{1}{10,000}\right)$$

Where a_{ij} = area (m^2) of patch ij.

b) Core area (CORE): equals the area (m^2) within the patch that is further than the specified depth-of-edge distance from the patch perimeter, divided by 10,000 (to convert to hectares).

$$CORE = a_{ij}^{c}\left(\frac{1}{10,000}\right)$$

Where a_{ij}^{c} = core area (m^2) of patch ij based on specified edge depths (m).

c) Core Area Index (CAI): equals the patch core area (m^2) divided by total patch area (m^2), multiplied by 100 (to convert to a percentage); in other words, CAI equals the percentage of a patch that is core area.

$$CAI = \frac{a_{ij}^{c}}{a_{ij}}(100)$$

Where a_{ij}^{c} = core area (m) of patch ij based on specified edge depths (m)

a_{ij} = area (m^2) of patch ij.

ii) Class level metrics

a) Total class area (CA): is a measure of how much of the landscape is comprised by a particular LULC type. It is a measure of landscape composition.

$$CA = \sum_{j=1}^{n} AREA_{ij} \left(\frac{1}{10{,}000}\right)$$

Where a_{ij} = area (m^2) of patch ij.

b) Mean patch area (AREA_MN): at the class level is a function of the number of patches in the LULC class and the total LULC class area.

$$AREA_MN = \frac{\sum_{i=1}^{n} AREA_{ij}}{n_i}$$

c) Percentage of landscape (PLAND): equals the sum of the areas (m^2) of all patches of the corresponding patch type, divided by total landscape area (m^2) and multiplied by 100 (to convert to a percentage).

$$PLAND = P_i = \frac{\sum_{j=1}^{n} a_{ij}}{A}$$

Where Pi = proportion of the landscape occupied by patch type (class) i.

a_{ij} = area (m^2) of patch ij.

A = total landscape area (m^2).

PLAND quantifies the proportional abundance of each patch type in the landscape. Like total class area, it is a measure of landscape composition important in many ecological applications.

d) Largest Patch Index (LPI (%)): equals the area (m^2) of the largest patch of the corresponding patch type divided by total landscape area (m^2), multiplied by 100 (to convert to a percentage). As such, it is a simple measure of dominance.

$$LPI = \frac{\max_{j=1}^{a}(AREA_{ij})}{A}\ (100)$$

Where a_{ij} = area (m^2) of patch ij. A = total landscape area (m^2).

iii) Landscape level metrics

a) Mean patch area (AREA_MN): equals the sum, across all patches in the landscape, of the corresponding patch metric values, divided by the total number of patches.

$$AREA_MN = \frac{\sum_{i=1}^{m}\sum_{j=1}^{n} AREA_{ij}}{N}$$

Where N = total number of patches.

b) Largest Patch Index (LPI): equals the area (m^2) of the largest patch in the landscape divided by total landscape area (m^2), multiplied by 100 (to convert to a percentage).

$$LPI = \frac{\max(a_{ij})}{A}(100)$$

Where a_{ij} = area (m^2) of patch ij

A = total landscape area (m^2)

Table 3 Description of landscape metrics used in this study

Index (unit)	Description
CA (ha)	Class area
PLAND	Percentage of landscape
NP	Number of patches
LPI (%)	The percentage of the landscape comprised by the largest patch
AREA_MN (ha)	Average size of patches
SHAPE_MN (ha)	Mean patch shape complexity weighted by patch area, based on shortest edge-to-edge distance
PROX_MN (m)	Average proximity index for all patches in a class
ENN_MN(m)	Mean Euclidean nearest neighbour distance
IJI (%)	Interspersion and juxtaposition index measures the juxtaposition of a focal patch class with all other classes
SHDI	Shannon’s diversity index is amount of patch per individual;
SHEI	Shannon's evenness index is the observed level of diversity divided by the maximum possible diversity for a given patch richness

B) Shape metrics

i) Patch level metrics

a) Shape index (SHAPE): equals patch perimeter (m) divided by the square root of patch area (m^2), adjusted by a constant to adjust for a square standard.

$$SHAPE = \frac{.25 p_{ij}}{\sqrt{a_{ij}}}$$

Where p_{ij} = perimeter (m) of patch ij

a_{ij} = area (m^2) of patch ij

Shape index corrects for the size problem of the perimeter-area ratio index by adjusting for a square standard and, as a result, is the simplest and perhaps most straightforward measure of shape complexity.

b) Fractal dimension index (FRAC): equals 2 times the logarithm of patch perimeter (m) divided by the logarithm of patch area (m^2); the perimeter is adjusted to correct for the raster bias in perimeter. Fractal dimension index is appealing because it reflects shape complexity across a range of spatial scales (patch sizes).

$$FRAC = \frac{2 \ln(.25\, p_{ij})}{\ln a_{ij}}$$

Where p_{ij} = perimeter (m) of patch ij.

a_{ij} = area (m^2) of patch ij.

ii) Class level metrics

a) Mean patch shape index (SHAPE_MN): equals the sum, across all patches of the corresponding patch type, of the corresponding patch metric values, divided by the number of patches of the same type.

$$SHAPE_MN = \frac{\sum_{j=1}^{n} SHAPE_{ij}}{n_i}$$

ii) Landscape level metrics

a) Mean patch shape index (SHAPE_MN): equals the sum, across all patches in the landscape, of the corresponding patch metric values, divided by the total number of patches.

$$SHAPE_MN = \frac{\sum_{i=1}^{m}\sum_{j=1}^{n} SHAPE_{ij}}{N}$$

Where N = total number of patches.

2. Aggregation metrics

This group of metrics represents the tendency of patch types to be spatially aggregated; that is, to occur in large, aggregated or "contiguous" distributions. It is also often referred to as landscape texture. Aggregation metrics include several closely related concepts such as: 1) dispersion, 2) interspersion, 3) subdivision, and 4) isolation. In this study, they are evaluated by the number of patches, mean Euclidean nearest-neighbor distance, interspersion and juxtaposition index and proximity index.

i) Class level metrics

a) Number of patches (NP): is a simple measure of the extent of subdivision or fragmentation of the patch type. Higher NP indicates greater fragmentation.

$NP = n_i$

Where n_i = number of patches in the landscape of patch type (class) i.

b) Mean Euclidean Nearest-Neighbour Distance (ENN_MN): is a measure of patch dispersion or isolation. It measures the distance to the nearest neighboring patch of the same type, based on shortest straight-line distance computed from cell centers.

$$ENN_MN = \frac{\sum_{j=1}^{n} X_{ij}}{n_i}$$

Where X_{ij}= the distance (m) from patch ij to the nearest neighbouring patch of the same LULC class.

The values for ENN are always larger than zero, without limit. ENN approaches zero as the distance to the nearest neighbour decreases. The minimum ENN is constrained by the cell size, and is equal to twice the cell size when the 8- neighbour patch rule is used (Posada, 2012). The upper limit is constrained by the extent of the landscape. ENN is undefined if the patch has no neighbours (i.e., no other patches of the same class).

c) Proximity Index (PROX_MN): equals the sum, across all patches of the corresponding patch type, of the corresponding patch metric values, divided by the number of patches of the same type.

$$PROX_MN = \frac{\sum_{j=1}^{n} X_{ij}}{n_i}$$

Where X_{ij} = distance (m) between patch ij and patch ij, based on patch edge-to-edge distance, computed from cell center to cell center.

Proximity index was developed by Gustafson and Parker (1994) and considers the size and proximity of all patches whose edges are within a specified search radius of the focal patch. The index is dimensionless (i.e., has no units) and therefore the absolute value of the index has little interpretive value; instead it is used as a comparative index.

d) Interspersion and Juxtaposition Index (IJI): expresses observed interspersion over the maximum possible interspersion for the given number of patch types. The IJI is based on patch adjacencies, not cell adjacencies. As such, it does not provide a measure of class aggregation like the CONTAG, but rather isolates the interspersion or intermixing of patch types.

$$IJI = \frac{-\sum_{k=1}^{m}\left[\left(\frac{e_{ik}}{\sum_{k=1}^{m} e_{ik}}\right)\ln\left(\frac{e_{ik}}{\sum_{k=1}^{m} e_{ik}}\right)\right]}{\ln(m-1)}(100)$$

Where e_{ik} = total length (m) of edge in landscape between patch types (classes) i and k.

m = number of patch types (classes) present in the landscape, including the landscape border, if present.

ii) Landscape level metrics

a) Number of Patches (NP): is the number of patches of the corresponding patch type (class).

$NP = N$

Where N = total number of patches in the landscape.

b) Mean Euclidean Nearest-Neighbour Distance (ENN_MN)

$$\text{ENN_MN} = \frac{\sum_{i=1}^{m}\sum_{j=1}^{n} X_{ij}}{N}$$

Where X*ij* = the distance (m) from patch *ij* to the nearest neighbouring patch of the same LULC class.

c) Interspersion and Juxtaposition Index

$$\text{IJI} = \frac{-\sum_{i=1}^{m}\sum_{k=i+1}^{m}\left[\left(\frac{e_{ik}}{E}\right)*\ln\left(\frac{e_{ik}}{E}\right)\right]}{\ln\left(0.5[m(m-1)]\right)}(100)$$

Where e_{ik} = total length (m) of edge in landscape between patch types (classes) i and k.

E = total length (m) of edge in landscape, excluding background.

m = number of patch types (classes) present in the landscape, including the landscape border, if present.

IJI approaches zero when the distribution of adjacencies among unique LULC classes becomes increasingly uneven. IJI is equal to 100 when all LULC classes are equally adjacent to all other LULC classes (i.e., maximally interspersed and juxtaposed to other LULC classes). IJI is undefined if the number of LULC classes is less than 3.

d) Mean Proximity Index (PROX_MN): equals the sum of patch area (m^2) divided by the nearest edge-to-edge distance squared (m^2) between the patch and the focal patch of all patches of the corresponding patch type whose edges are within a specified distance (m) of the focal patch.

$$\text{PROX_MN} = \frac{\sum_{i=1}^{m}\sum_{j=1}^{n} X_{ij}}{N}$$

Where X_{ij}= distance (m) from patch ij to the nearest neighbouring patch of the same patch type, based on patch edge-to-edge distance, computed from cell center to cell center.

Proximity index considers the size and proximity of all patches whose edges are within a specified search radius of the focal patch. The index is dimensionless (i.e., has no units) and therefore the absolute value of the index has little interpretive value; instead it is used as a comparative index.

3. Landscape diversity metrics

These metrics quantify landscape composition at the landscape level. They are not affected by the spatial configuration of patches. In this study, they are measured by Shannon's Diversity Index and Shannon's Evenness Index.

a) Shannon's Diversity Index (SHDI): equals minus the sum, across all patch types, of the proportional abundance of each patch type multiplied by that proportion. SHDI increases as the number of different patch types (i.e., patch richness, PR) increases and/or the proportional distribution of area among patch types becomes more equitable.

$$\mathrm{SHDI} = -\sum_{i=1}^{m}(P_i * \ln P_i)$$

Where P_i = proportion of the landscape occupied by patch type (class) i.

b) Shannon's Evenness Index (SHEI): equals minus the sum, across all patch types, of the proportional abundance of each patch type multiplied by that proportion, divided by the logarithm of the number of patch types. In other words, the observed Shannon's Diversity

Index divided by the maximum Shannon's Diversity Index for that number of patch types. Shannon's evenness index is expressed such that an even distribution of area among patch types results in maximum evenness. As such, evenness is the complement of dominance.

$$\text{SHEI} = \frac{-\sum_{i=1}^{m}(P_i * \ln P_i)}{\ln m}$$

Where Pi = proportion of the landscape occupied by patch type (class) i.

m = number of patch types (classes) present in the landscape.

All metrics were analyzed using Raster version of FRAGSTATS spatial pattern analysis software (ver.4.1) developed by McGarigal *et al.* (2012). An eight-neighborhood criterion for the definition of patches was adopted (Fig. 6). FRAGSTATS allows the user to define a specific edge depth - a distance from the surrounding matrix into the forest interior beyond which is the core area. Therefore, 30 meter edge influence was chosen to assess mean core area (Kumar *et al.,* 2002).

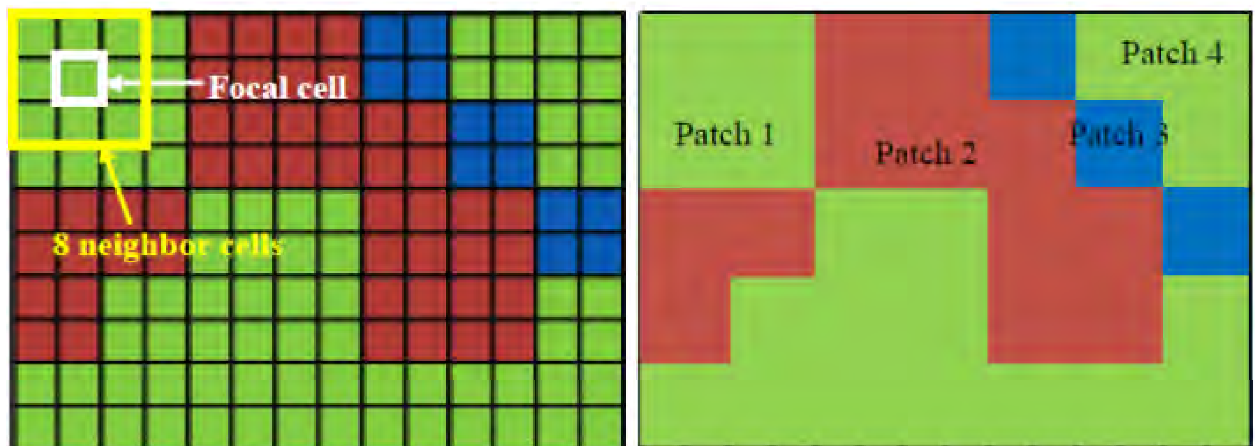

Fig. 6 Schematic representation of the 8 neighbor rule to define patches. Source: Posada (2012)

3.4. Data analysis

3.4.1. Cluster analysis

Classification by means of cluster analysis is the most common multivariate technique used in ecological works. In this study, a hierarchical (agglomerative) cluster analysis was performed using the free statistical software R version 3.1.2 (R Development Core Team, 2014) using package cluster to identify plant communities. Similarity ratio with Ward's group linkage method was applied for cluster analysis i.e. to determine plots that can be classified into the same groups based on the species abundance data. Indicator species analysis was performed in R using package labdsv (Roberts, 2012). Linear projection dendrogram was used to depict the outcome of cluster analysis. The difference in floristic composition among the plant communities was tested with the nonparametric Multi-Response Permutation Procedure (MRPP) using PC-ORD version 5.31 (McCune and Grace, 2006). Shannon-Wiener and Simpson`s diversity indices and Shannon’s evenness were computed to describe species diversity of the plant community types in the vegetation.

The community types identified from the cluster analysis were further refined in a synoptic table and species occurrences are summarized as a synoptic-cover abundance values. Synoptic values are the product of the species frequency and average cover abundance values (van der Maarel, 1979). Dominant species of each community type were identified based on their synoptic values. Finally, the community types were named based on two dominant species.

3.4.2. Ordination

Ordination is most often used in ecology to seek and describe pattern, hence in community ecology its most common use is to describe the strongest patterns in species composition (McCune and Grace, 2002). Ordination analysis of floristic data (species - environment matrix) has been carried out to investigate the relationship between floristic gradients and ten environmental variables. Nonmetric multidimensional scaling (NMDS) ordination analysis was used to investigate the environmental factors that influence the structure of plant communities. NMDS is the most robust form of ordination for detection of ecological pattern (Minchin, 1987) and avoids the assumption of linear relationships among variables (Clarke, 1993).

Canonical correspondence analysis (CCA) was used for detection of patterns of variation in species distribution and for identification of variables causing those patterns in the study area. CCA differs from other ordination methods because it incorporates the correlation and regression between floristic and environmental data within the ordination analysis (ter Braak, 1995). The variables considered were altitude, slope, aspect, human disturbance, grazing, pH, electrical conductivity, clay, silt and sand. Step wise variable selection (following Zerihun Woldu, in press; Oksanen, 2012) was applied to identify the environmental variables with significant influence on the plant distribution pattern of the study area. The resulting ordination diagram, expresses not only patterns of vegetation in floristic composition but also demonstrates the principal relationship between the species and each of the environmental variables. Hence, for CCA analysis, 77 sample plots, 207

species and ten environmental variables were included. NMDS and CCA were computed by using the free statistical software R version 3.1.2 (R Development Core Team, 2014).

3.4.3. Plant diversity analysis

3.4.3.1. Diversity and similarity measures

Diversity measures such as Shannon diversity index (H´), Simpson diversity (D), evenness (E´) and Fisher alpha were calculated. The formulae for computing diversity indices and evenness were indicated below:

a) Shannon Weiner's index of diversity was calculated by the formula:

$H' = - \Sigma ((n_i/N) * \ln (n_i/N))$

Where H' = the value of the Shannon's diversity index

n_i = number of individuals belonging to the 'i' species

N = total number of individuals in the sample, and ln = the natural log of the number.

b) Simpson's index of diversity (D) was calculated by the formula:

$$D = \frac{\sum n(n-1)}{N(N-1)}$$

n = the number of individuals each species and

N = the total number of individuals of the species for the site

c) Evenness was calculated as the ratio of observed diversity to maximum diversity using the equation:

$E = H'/H\text{max}$

Where Hmax = ln$S$

Hmax is the maximum level of diversity possible within a given population,

S = species richness (total number of species).

To describe the relationship between the number of species and total number of individuals, Fishers' α, which is often known as log-series distribution (Magurran, 2004) was used. The diversity index α can be obtained from the equation:

$$\alpha = \frac{N(1-x)}{x}$$

Where α is a constant that depends on diversity, and x is a variable that depends on sample size.

To evaluate the similarity of plant communities of Gratkhassu and Hugumburda Forest fragments, the widely used and classical similarity indices; Sørensen and Jaccard similarity indices as well as their respective abundance based indices (Chao-Sørensen estimator and Chao-Jaccard estimator) were calculated (Magurran, 2004). The following equations were used to calculate the Sørensen and Jaccard similarity indices respectively:

$S = \frac{2C}{A+B}$ and $J = \frac{C}{A+B-C}$

Where S is Sørensen‟s similarity index

J is Jaccard's similarity index

C is the number of species common to both sites,

A is the number of species present in one of the sites to be compared

B is the number of species present in the other site.

Shannon diversity index, Simpson's index of diversity and evenness were calculated using R version 3.1.2 (R Development Core Team, 2014). Fisher alpha and all the similarity indices were computed using EstimateS version 9.1.0 (Colwell, 2013). To evaluate whether there was statistically significant difference between Shannon diversity indices of the two forest remnants t-test was conducted using SPSS software (Version 16).

3.4.3.2. Rarefaction and species accumulation curves

Since number of species is highly dependent on sample size, comparing communities having different sample size is problematic (Magurran, 2004). Hence to overcome this problem, all samples from different communities should be standardized to a common sample size of the same number of individuals. Sanders (1968) proposed rarefaction method for achieving this goal. Rarefaction is a statistical method for estimation of the number of species expected in a random sample of individuals taken from a collection.

In this study, sample based rarefaction curves (Gotelli and Colwell, 2001) were computed using EstimateS version 9.1.0, to compare the species richness of Hugumburda and Gratkhassu Forest fragments. To evaluate the effectiveness of the species estimators and to examine the degree of species collection (sampling) species accumulation curve was also plotted.

3.4.3.3. Species richness estimation

The number of species or species richness in a species assemblage is a significant measure of biodiversity at the habitat level (Mao and Colwell, 2005). However, it is virtually impossible to detect all species and their relative abundance with a limited

number and intensity of samples (Chao *et al*., 2005). To meet this challenge, several methods have been developed for estimating species richness from sample data, either through extrapolation of species accumulation curves or through application of non-parametric methods (Magurran, 2004). The latter approach involves the estimation of unseen species (species that is likely to be present in a larger homogenous sample of the assemblage, but that are missing from actual sample data) (Chao *et al*., 2005).

Non-parametric species richness estimators including ACE, Chao1, Chao 2, Jackknife 1, Jackknife 2 and Bootstrap (Colwell, 2013) were used to estimate the number of species in Hugumburda-Gratkhassu Forest. These non-parametric species richness estimators are applicable to quadrat-based data sets that can be treated as either the random samples of space or as fixed samples of individuals. Moreover these estimators are free of parametric (normal) species abundance distribution model (Magurran, 2004). By employing appropriate formulae of the different nonparametric estimators, estimate of species richness, was computed using EstimateS version 9.1.0 (Colwell, 2013). In order to calculate the mean estimator and expected number of species for each sample accumulation level, the sample order was randomized 100 times.

3.4.4. Species population structure data analysis

Matured woody species recorded in the study quadrats were used in the analysis of population structure. Frequency, density, DBH, height and basal area were used for the description and comparison of population structure in the study area. Tree density was computed by converting the count from the sample plot to a hectare basis. The diameter at breast height (DBH) was classified into 10 DBH classes and the percentage

distribution of trees and shrubs in each class were computed. In addition, height was classified into eight height classes and the percentage distribution of the plants in each class was calculated to assess pattern of population dynamics of the forest fragments. The remaining population parameters were computed and summarized on Microsoft Office Excel (2007) spread sheet using the following formulae.

Frequency (F) - the proportion of quadrats in which a species found. The frequency value reflects the pattern of distribution and expressed as number of quadrats in which species recorded per total number of quadrats as a percentage (Goldsmith *et al*., 1986).

$$F = \frac{\text{Number of quadrats in which a species occur}}{\text{Total quadrats laid}} \text{ X } 100$$

Where F was expressed as percentage for each species.

Density (D) - is the count of individuals per unit area. It is calculated from the count of the individuals from the study area per hectare basis.

$$D = \frac{\text{Number of above ground stems of a species counted}}{\text{Sample area in hectare}} \text{ X100}$$

Where D was expressed as percentage for each species.

Basal area (BA) - is the area outline of a plant near ground surface for tree and selected shrub species. It was calculated based on value of diameter at breast height (DBH), and it was expressed in square meter per hectare (Mueller-Dombois and Ellenberg, 1974). In

the present study basal area was calculated from DBH of each adult woody species using the following formula:

$BA = \pi (d/2)^2$

Where d is diameter at breast height and $\pi = 3.14$

3.4.5. Regeneration status

According to Harper (1977), studies on densities of age classes (seedlings, saplings and adult plants) help to determine regeneration status of forests. Regeneration status of shrubs and trees for each of the two forest remnants was analyzed and compared by using the densities of different age classes (seedlings, saplings and mature plants) whereas the regeneration status of the two forest remnants was analyzed and compared using density ratios between age classes (ratios between seedlings and saplings and mature plants). Recruitment potential of the two forest remnants were compared based on the number of seedlings to saplings using the non-parametric (Mann-Whitney U) test. Computation was made using SPSS version 16.0 statistical software. Seedlings and saplings count in the five subplots were summed and considering as a value for a plot with an area of 20 m^2 finally converted to per hectare basis. Generally a total of 99 woody species of which 36 were tree species, 51 were shrubs and 12 liana species distributed in 78 genera and 48 families were used for this analysis.

3.4.6. Soil analysis

The soil samples were analyzed in the Ecology and Ecophysiology Lab. at the Department of Plant Biology and Biodiversity Management (AAU) (Appendix 12), following the methods described by Juo (1978) and Sahlemedhin Sertsu and Taye Bekele (2000). A brief description of each of the parameters analyzed is given below.

Soil pH was determined using 1:2.5 soil - water ratio following Sahlemedhin Sertsu and Taye Bekele (2000). Ten gm of soil was mixed with 25 ml of distilled water in 50 ml beaker. The suspension was stirred occasionally and the pH was measured after 30 minutes using digital pH meter (inolab pH/Ion level 2, WTW). The pH meter was standardized using buffer solutions of pH 4.0 and 9.2.

Electrical conductivity (EC) is a measure of soluble salts in soil. To determine EC, soil-water suspension was made by mixing 10 gm of soil in 25 ml of distilled water (Sahlemedhin Sertsu and Taye Bekele, 2000). The suspension was stirred for 30 minutes and EC reading was taken using conductivity/TDS/salinity/resistivity conductometer.

Soil texture (sand, silt and clay) was determined using hydrometer method of mechanical analysis (Juo, 1978; Sahlemedhin Sertsu and Taye Bekele, 2000). Sodium hexametaphosphate (40 gm) and calcium carbonate (10 gm) were used as dispersing agent. Fifty gm of soil was mixed with 100 ml of dispersing agent in a 500 ml plastic bottle. The suspension was shaken for three hours on an oscillatory shaker. Then the dispersed soil suspension was transferred into a hydrometer jar and diluted with distilled water up to one liter mark. The suspension was mixed by covering the top of the cylinder with a palm till the materials deposited at the bottom disperse in the suspension. Hydrometer and temperature readings were taken after 40 seconds and three hours. From the hydrometer and temperature reading obtained, particle size was then calculated using the following formula:

% Sand = 100 – ($H_1 + T_1 – 2$) x 2

% Clay = ($H_2 + T_2 – 2$) x 2

% Silt = 100 – (% Sand + % Clay)

Where H_1 = first hydrometer reading after 40 seconds, H_2 = second hydrometer reading after three hours, T_1 = first temperature reading (in ^oF) after 40 seconds and T_2 = second temperature reading after three hours (in ^oF).

3.4.7. Land use\land cover change data analysis

3.4.7.1. LULC classification and mapping

Performing image analysis is an inevitable task to extract meaningful information from remotely sensed data. In this study, LULC classification and mapping was performed in four stages: 1) Preprocessing of the images 2) Determination of landcover types 3) Supervised classification of the image into LULC classes and 4) Accuracy assessment. These applications were performed using ERDAS Imagine 9.2, ENVI 4.7 and ArcMap softwares.

Pre-processing image

The standard image processing techniques of extraction, layer stacking, geometric correction/georeferencing, image enhancement and change detection were performed on the three Landsat images (Table 1). They served as the primary data for this study.

Image classification

Image classification is the process of sorting pixels into a finite number of individual classes, or categories of data based on their data file values. If a pixel satisfies a certain

set of criteria, then the pixel is assigned to the class that corresponds to those criteria. For this study a supervised classification scheme with maximum likelihood classifier decision rule was used by following three stages, assigning training sites, classification and outputs (Fig. 5). The LULC classification was done on the basis of reflectance characteristics of the different land use\land cover types by using false color composites. This was supplemented by field visits that made it possible to establish the main land use\land cover types (Table 2). For each LULC categories six regions of interest was selected during classification.

After classification, majority analysis was used in order to avoid minor fragmented classification arrangements on the output map. The majority analysis was repeated four times until fine classification was obtained. The approach consists of comparing the properly coded results of two separate classifications. Finally, ground truthing was made in order to check the precision of the classified LULC map. Based on the ground verification necessary correction and adjustments were made.

3.4.7.2. Accuracy assessment

After classification is performed, evaluating the classification result is the next task for testing the accuracy of the classification since it gives evidence of how the results reflect the reality on the ground. In this study, accuracy assessment of the classification result was made on the basis of the year 2011 landcover\landuse image classification using ERDAS Imagine 9.2 software. This was used to assess the errors of omission and commission resulting in misclassifying pixels. Producer accuracy, user accuracy, overall

accuracy and kappa coefficient (Table 18) were used to indicate how much the classified image represented the actual features on the ground.

3.4.7.3. Post image classification and change detection

Post classification comparison was carried out for the three independent images using pixel-based statistical analysis, which is the most proven technique that deals with change detection (Shalaby and Tateishi, 2007). Such change detection matrices allow a more rigorous quantitative analysis of land use\land cover change trends over time. This approach measures the change dynamic in each cell for all land use\land cover classes and class changes as well as total gains and losses of specific land use\land cover types.

A precondition for the application of change detection matrices is a similar spatial resolution of geo-information when comparing one time step with the next (Alphan *et al*., 2009; Yohannes Kidane, *et al*., 2012). The Landsat TM 30 m image of 1986 and 2011 were resampled to 60 m resolution in order to match the pixel size to the Landsat MSS of 1973 image. The ENVI 4.7 modeler was used to detect landcover\landuse changes between the three datasets. So that, two comparisons based on three classified maps, 1973, 1986 and 2011 were made. The first comparison was between 1973 and 1986; and the second comparison was between 1986 and 2011 maps. This helped to establish the degree of forest cover change during the past thirty nine years. The flow chart (Fig. 5) shows the procedures followed during the land use\land cover change analysis and the input resources used.

Finally, the rate of change was calculated for each land use\land cover type using the following formula:

Rate of change = $\frac{(A-B)}{C}$

Where, A = recent areas of land use\land cover in ha, B = previous area of land use\land cover in ha, C = time interval between A and B in years.

3.4.8. Analysis of metrics in FRAGSTATS

For FRAGSTATS to compute landscape metrics, recode and modeling processes were first carried out on the images in ERDAS Imagine 9.2 software. The resulting images were stored as signed-8 bit files (Kabba and Li, 2011). The raster version of FRAGSTATS 4.1 was used to compute patch, class and landscape level metrics for the three study years (1973, 1986 and 2011). After running the FRAGSTATS program, the results were exported to an Excel spreadsheet and tables were constructed to summarize the landscape pattern metrics. Then analyses of the results were made by examination of the values (maximums, minimums and extreme values) observed for each metric.

Patch area and fractal dimension computation

LULC map of 2011 was used to compute three patch metrics, namely, patch size, core area and patch fractal dimension to evaluate their relationship with plant species richness. Eighteen patches (Fig. 42), which were well distributed over the landscape and provided valuable descriptions of the local diversity were chosen for this analysis. Computation of these metrics was done after the map of the fragments was produced in a GIS

environment by using the patch analyst extension of the ArcGIS (ESRI) software. The shape files of the landcover classification were loaded into FRAGSTATS and analyzed to get the values for each of the parameter for all patch types.

A total of 83 sample plots (qaudrats), size 20 m x 20 m (400 m^2) were systematically selected from both forest remnants (18 patches) based on their size. Then, floristic inventories (presence/absence of all plants) were conducted from each qaudrat in order to determine the relationship of patch area to abundance of plants.

CHAPTER FOUR

4. RESULTS

4.1. Plant diversity studies in Hugumburda-Gratkhassu NFPA

4.1.1. Floristic composition

Including those plants that were found out of the study plots (quadrats), totally 326 plant species from 238 genera and 88 families were recorded (Appendix 1). Out of these 326 species, only 224 were recorded in the present study and the remaining 102 species were also recorded in a previous study by Leul Kidane *et al*. (2010). From the total 326 species recorded from the study area, 10 (3.07%) species were newly recorded for WU Floristic Region of Ethiopia (Appendix 3). When the number of species, genera and families from the two forest remnants were analyzed separately, Gratkhassu forest fragments had higher share in all aspects. As shown in Fig. 7 the number of families, genera and species which were recorded in Gratkhassu were higher than that of Hugumburda forest fragments. In Gratkhassu, a total of 226 species were recorded while in Hugumburda the number of species was 211, which is not statistically significant ($P > 0.05$).

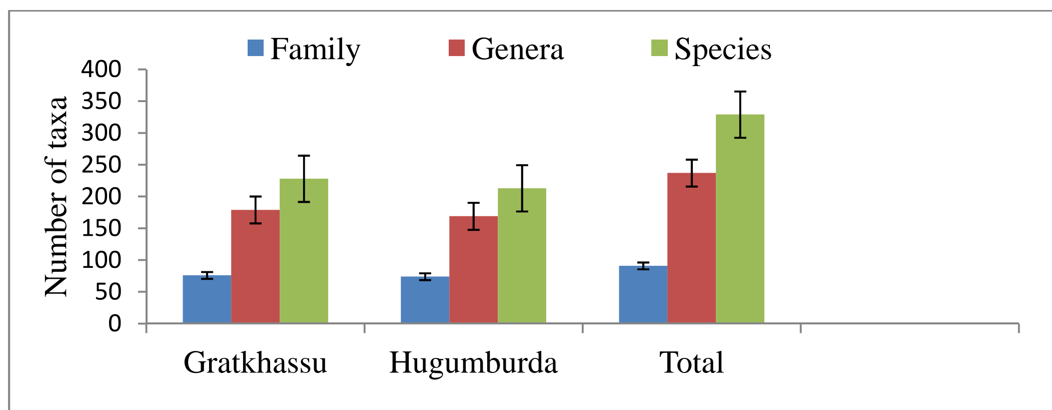

Fig. 7 Total number of families, genera and species recorded in Hugumburda-Gratkhassu Forest (bars show density ± standard error)

Similarly, 76 and 74 families were identified in Gratkhassu and Hugumburda forest remnants, respectively. The number of genera also decreased from 177 to 167 comparing Gratkhassu to Hugumburda forest remnants (Fig. 7).

Out of the 326 species recorded, 34% of them (111 species) occurred in both forest remnants. One hundred fifteen species (35.3% of the total) were recorded only in Gratkhassu while 30.7% (100 species) were recorded only in Hugumburda Forest fragments.

When the whole data set is considered together, the species richest family of Hugumburda-Gratkhassu Forest was Asteraceae, followed by Poaceae and Fabaceae (Table 4). The next species rich families were Lamiaceae, Solanaceae and Euphorbiaceae (Appendix 2).

Table 4 Most species rich families recorded in Hugumburda-Gratkhassu Forest

Families	Number of species	Number of genera
Asteraceae	38	27
Poaceae	31	20
Fabaceae	31	19
Lamiaceae	22	12
Solanaceae	13	7
Euphorbiaceae	10	6
Others	181	148
Total	326	238

The order of richest families was a bit different when the data was analyzed separately (Table 5). In Gratkhassu, as indicated in Table 5, Poaceae was the richest family being represented by 28 species and 17 genera. The second richest families were Asteraceae and Fabaceae which had 22 species each and 18 and 14 genera respectively. Followed by Lamiaceae and Solanaceae, consisting of 15 and 8 species respectively (Table 5).

Table 5 Families with seven or more species in Gratkhassu and Hugumburda Forest remnants

Gratkhassu			Hugumburda		
Families	**Number of species**	**Number of genera**	**Families**	**Number of species**	**Number of genera**
Poaceae	28	17	Asteraceae	22	16
Asteraceae	22	18	Fabaceae	19	14
Fabaceae	22	14	Poaceae	14	14
Lamiaceae	15	10	Lamiaceae	11	7
Solanaceae	8	5	Solanaceae	11	6
Euphorbiaceae	7	5	Euphorbiaceae	7	4
Others	124	108	Others	127	106
Total	226	177	Total	211	167

In Hugumburda, Asteraceae was the richest family being represented by 22 species and 16 genera. The second richest family was Fabaceae with 19 species and 14 genera followed by Poaceae consisting of 14 species and genera. Lamiaceae, Solanaceae and Euphorbiaceae were other species rich families in Hugumburda which were represented by seven or more species each (Table 5).

In both forest remnants the first six species rich families comprised considerable number of species. In Gratkhassu, 44.73% (101 species) of the recorded species belong to the top

six families. However, 28 of the species (12.28%) recorded in this forest fragments were represented only by one family. The same is true for Hugumburda, where, 39.4% (84 species) of the total species were from top six species rich families and 10.33% (22 species) were represented only by a single family (Table 5).

Based on the available information on the published floras of Ethiopia and Eritrea, of the total 326 species 15 (4.6%) species were found to be endemic to Ethiopia and Eritrea. Out of this only four species (*Aloe debrana, Crassocephalum macropappum, Laggera tomentosa* and *Urtica simensis)* were found to be endemic to Ethiopia and nine species have been registered in the IUCN Red List Categories of Ethiopia and Eritrea (Vivero *et al*., 2005) (Table 6).

Table 6 Endemic plant species of Ethiopia and Eritrea occurring in Hugumburda-Gratkhassu Forest**.** Key: LC = Least Concern; NT = near threatened; VU = Vulnerable; CR= critically endangered; E=Endemic

Species scientific name	**Family**	**Habit**	**Endemism**	**IUCN Category**
Aloe debrana	Aloaceae	Succulent	E	Not evaluated
Anarrhinum forskaohlii subsp. *abyssinicum*	Scrophulariaceae	Herb	E	Not evaluated
Astragalus atropilosulus subsp. *abyssinicus*	Fabaceae	Shrub	E	Not evaluated
Becium grandiflorum	Lamiaceae	Shrub	E	NT
Clutia abyssinica	Euphorbiaceae	Shrub	-	VU
Crassocephalum macropappum	Asteraceae	Herb	E	LC

Euphorbia monacantha	Euphorbiaceae	Herb	E	Not evaluated
Gomphocarpus purpurascens	Asclepiadaceae	Herb	E	Not evaluated
Laggera tomentosa	Asteraceae	Herb	E	NT
Lippia adoensis	Verbenaceae	Shrub	E	LC
Phagnalon abyssinicum	Asteraceae	Herb	E	Not evaluated
Solanum marginatum	Solanaceae	Herb	E	LC
Thymus schimperi	Lamiaceae	Herb	E	LC
Urtica simensis	Urticaceae	Herb	E	LC
Verbascum pedunculosa	Scrophulariaceae	Herb	E	Not evaluated
Vernonia rueppellii	Asteraceae	Tree	E	LC

4.1.2. Diversity measures

As indicated in Table 7 all Fisher α, Shannon diversity and Simpson diversity indices were higher in Gratkhassu than in Hugumburda Forest fragments. These values were compared statistically to see if the difference was significant. The t-test showed that the difference was significant (t-value = -19.09, $P < 0.001$). This showed that Gratkhassu Forest fragments had relatively higher species diversity than that of Hugumburda. Even if higher species diversity had been recorded at Gratkhassu than at Hugumburda, both sites were having almost equivalent evenness index which was 0.678 for Gratkhassu and 0.742 for that of Hugumburda (Table 7).

Table 7 Various diversity measures for Hugumburda and Gratkhassu Forest remnants

Diversity measurement	Gratkhassu	Hugumburda	Total
Fisher's α	58.29	51.7	74.88
Shanon diversity (H')	4.31	4.17	4.47
Evenness index (E')	0.678	0.742	0.802
Simpson Inv Mean	42.83	37.72	46.08

4.1.3. Species accumulation curve

As indicated in Fig. 8, species accumulation curves (rarefaction) were plotted for the plant species recorded in Hugumburda and Gratkhassu Forest remnants.

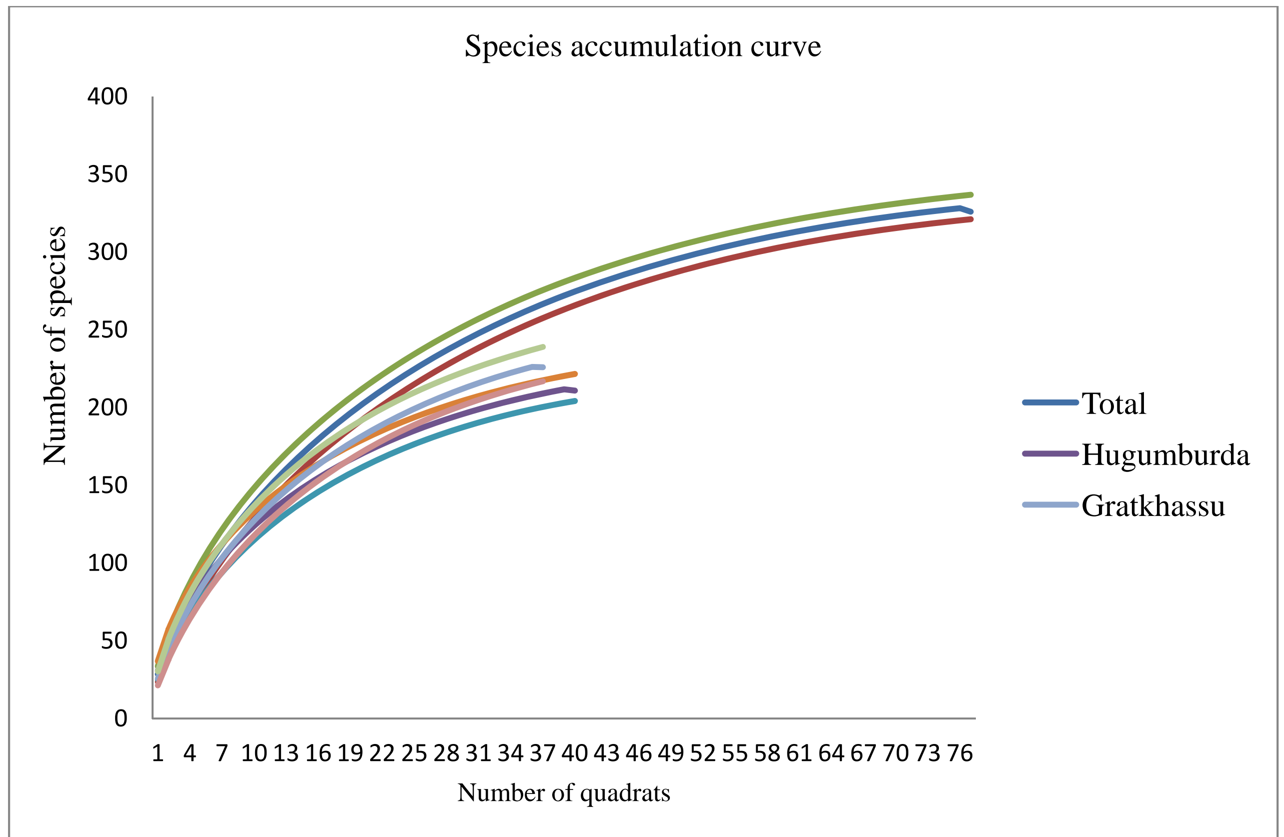

Fig. 8 Species accumulation (rarefaction) curves and 95% confidence intervals for Gratkhassu and Hugumburda Forest remnants

From Fig. 8 it was observed that there are still more species to be recorded in both forest remnants though at decreasing rate. The rate was still higher for Gratkhassu Forest fragments. The rarefied number of species based on 37 quadrats (plots of Gratkhassu) resulted in 217.63 species, in Hugumburda Forest fragments however, 226 species were recorded at Gratkhassu forest fragments at this sampling effort. Rarefaction curve is also useful to test whether there is significance difference in species richness between different sites or not. As it is depicted from Fig. 8 the observed species accumulation curve of Gratkhassu was outside of the 95% confidence interval of Hugumburda revealing that Gratkhassu had significantly higher species richness than that of Hugumburda.

4.1.4. Species richness estimation

A total of 326 plant species (including those found outside of the study plots) were recorded in the study area. However, all the species found in the ecosystem, especially those which are rare, may not have been recorded. Hence, it is valid to estimate the total number of species. Based on different species richness estimators, the total number of species in Hugumburda-Gratkhassu Forest was estimated to be between the range of 347 and 383 (Table 8). The lowest estimate (i.e. 347) was given by Chao1, while the highest estimate was obtained using Jackknife 1 (i.e. 383) (Table 8). Based on these different estimations, in this study 85.57 to 94.52% of the estimated species were recorded from the study area.

Table 8 Species richness estimation (based on different estimators) of Hugumburda-Gratkhassu Forest

Estimators	Gratkhassu	Hugumburda	Combined
Number of samples	37	40	77
Sobs	208	204	301
ACE	274.16	246.09	355.77
Chao 1	269.64	237.54	347.01
Chao 2	265.01	234.76	345.47
Jackknife 1	293.19	261.75	383.29
Jackknife 2	302.31	257.41	352.27
Bootstrap	261.65	239.8	363.9
Species collection degree	75.42 - 87.14%	81.38 - 90.73%	85.6-94.52%

As shown in Table 8 using different methods, species richness (number of species) of both Gratkhassu and Hugumburda Forest remnants was separately estimated. All the species richness estimators used in this study resulted in higher species richness than the observed number of species (Sobs). In both forest patches the highest species richness estimation (302.31 for Gratkhassu and 261.75 for Hugumburda) was obtained by second and first order Jackknife estimators, respectively (Table 8). Chao 2 estimator gave the lowest species richness estimate (234.76) for Hugumburda. While the smallest estimate for Gratkhassu (261.65) was given by Bootstrap estimator. In this study 75.42 - 87.14% of the species estimated by various estimators for Gratkhassu as well as 81.38 - 90.73% for that of Hugumburda were recorded (Table 8).

4.1.5. Floristic similarities

As indicated in Table 9 almost equal incidence based similarities were observed for Gratkhassu and Hugumburda Forest fragments. Sørensen similarity index was 0.71 for Gratkhassu and 0.72 for Hugumburda and that of Jaccard similarity index was 0.55 for Gratkhassu and 0.56 for Hugumburda. Relatively higher abundance based similarities were observed for Gratkhassu. Chao-Sørensen estimator and Chao-Jaccard estimator showed high value for Gratkhassu (0.82 and 0.69) than Hugumburda (0.77 and 0.62, respectively) (Table 9).

Table 9 Incidence and abundance based similarity indices of Hugumburda and Gratkhassu Forest remnants

Gratkhassu		**Hugumburda**	
Incidence based	**Abundance based**	**Incidence based**	**Abundance based**
Sørensen 0.71	Chao-Sørensen estimator 0.82	Sørensen 0.72	Chao-Sørensen estimator 0.77
Jaccard 0.55	Chao-Jaccard estimator 0.69	Jaccard 0.56	Chao-Jaccard estimator 0.62

4.1.6. Plant community types

Five plant community types were identified from the hierarchical cluster analysis (Fig. 9). The analysis was based on the abundance data of the species. The test statistic **t** value from MRPP technique (no-difference hypothesis) for the five groups was -41.92 ($P < 0.001$) showing that the five groups are different. The agreement statistics A was 0.714. The test statistics, t, describes the separation between the groups (McCune and Grace, 2002). The more negative it is, the stronger the separation. The P-value associated with t is determined by numerical integration of the Pearson type III distribution and it is useful

for evaluating how likely an observed difference is due to chance (McCune and Grace, 2002).

The identified plant communities of this study varied in size ranging from 9 to 25 plots (Fig. 9). A synoptic cover-abundance value for species reaching a value ≥ 1.0 in at least one community type is presented in Table 10. The community was named after the dominant tree or shrub species in that specific community with higher synoptic values (Table 10). Accordingly, the following is a description of the identified five plant communities of Hugumburda-Gratkhassu Forest.

1. *Juniperus procera – Nuxia congesta* Community Type

This Community consists of 13 plots, in which all belong to Gratkhassu Forest fragments. The altitudinal range of this community was from 2214 to 2666 meter above sea level (having 2422 m mean altitude). Topographically this community was situated on gentle slope (30% mean slope). Totally, 122 species were associated with this community, *Juniperus procera* and *Nuxia congesta* were dominant species and *Ocimum suave* was characteristic species of the type. The common tree species of this community were *Maytenus undata, Osyris quadripartita, Olea europaea* ssp. *caspidata*, *Dovyalis abyssinica, Rhus glutinosa, Bersama abyssinica, Cassipourea malosana* and *Cupressus lusitanica. Calpurnia aurea, Carissa spinarum, Clutia abyssinica* and *Dovyalis verrucosa* were the common shrub species in this Community. The herb layer was dominated by *Orobanche ramosa, Hypoestes forskaolii, Cyanotis barbata, Haplocarpha schimperi, Phagnalon abyssinicum, Spermacoce sphaerostigma, Vigna membranacea* and *Vermifrux abyssinica.* There were also Lianas such as *Asparagus racemosus* and *Solanum benderianum* in this Community.

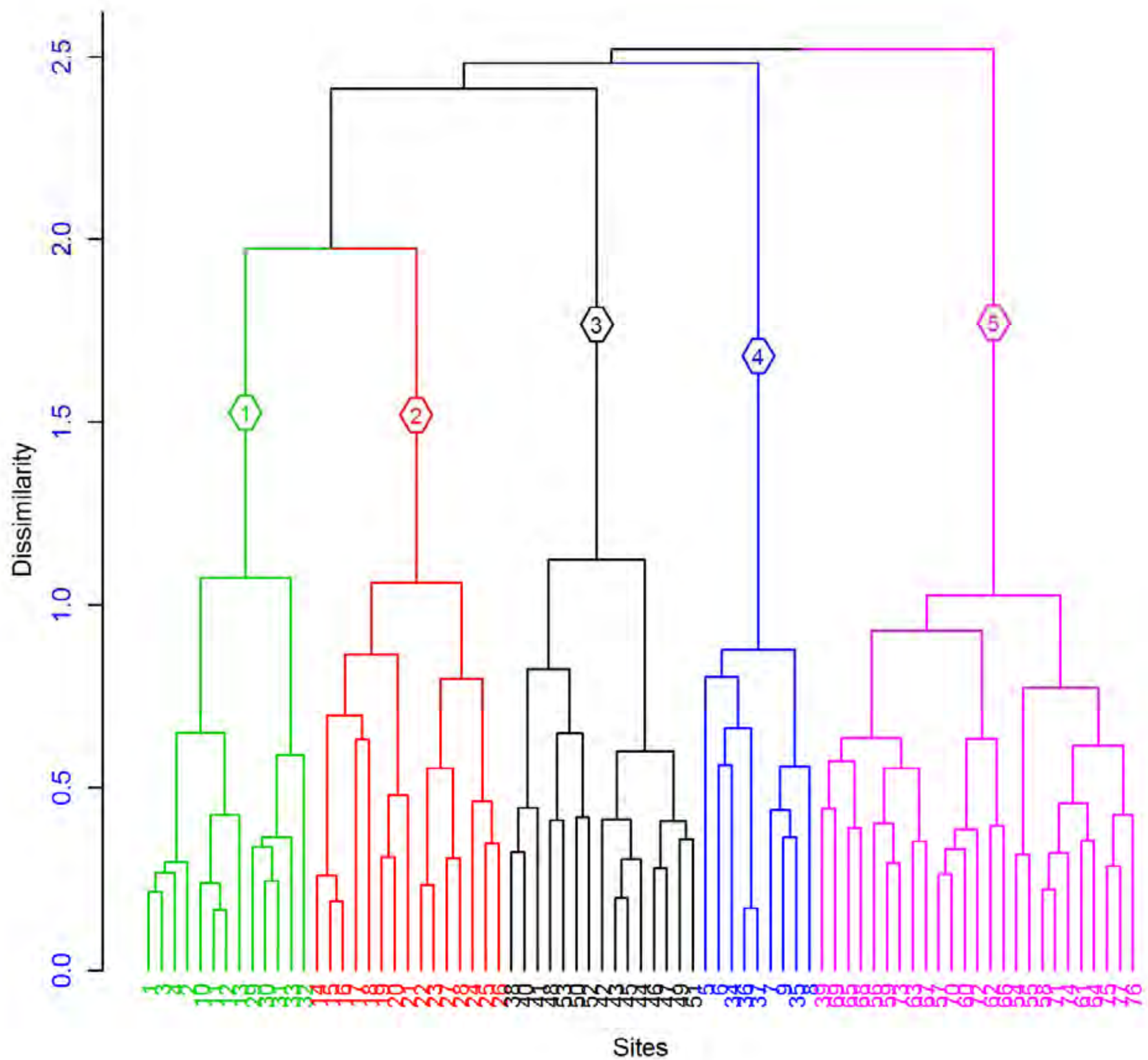

Fig. 9 Dendrogram showing plant community types of Hugumburda-Gratkhassu Forest

Table 10 Species list with synoptic cover-abundance values for species having a value of ≥ 1 in at least one community type. The probability value refers to Monte Carlo tests, while values under each group are synoptic values (with significant values, P < 0.05) in a decreasing order of their synoptic values which indicate the faithfulness of occurrence of species within a particular community.

Species name/community type	**1**	**2**	**3**	**4**	**5**	
Number of plots	**13**	**15**	**15**	**9**	**25**	Probability
Juniperus procera	8.08	5.6	3.27	1.67	5.6	0.0002
Nuxia congesta	6.46	3.07	4.87	0	2.08	0.0002

Hypoestes forskaolii	6.38	1.93	5.87	1.89	4.32	0.0002
Maytenus undata	5	1.33	1.6	0.67	1.56	0.0002
Osyris quadripartita	4.38	1.2	0.27	0.11	0.96	0.0002
Olinia rochetiana	3.62	0	1.4	0	1.28	0.0002
Calpurnia aurea	3.62	1.2	0	0.56	0.56	0.0002
Dombeya torrida	1.38	0	0.87	0.11	0.04	0.001
Hypericum quartinianum	1.15	0	0	0	0	0.0056
Orobanche ramosa	1	0	0.13	0	0	0.0076
Cyanotis barbata	1	0.07	0	0	0	0.0148
Cassipourea malosana	2.85	6.4	3.73	0	1.48	0.0002
Andropogon abyssinicus	1.23	5.33	2.67	1.33	3.6	0.0154
Myrica salicifolia	0	4.67	0	0	0	0.0002
Bersama abyssinica	0.31	3.6	1.33	0	0.32	0.0054
Dovyalis abyssinica	2.92	3.33	0	0	0.28	0.0028
Cupressus lusitanica	1.46	3.4	0.33	0	0.04	0.0014
Rosa abyssinica	0	1.13	0.13	0	0.32	0.001
Myrsine africana	0.23	1.13	0.8	0.22	0.4	0.0076
Psydrax schimperiana	0.46	0.2	4.87	0	1.76	0.0002
Podocarpus falcatus	0	0	4.53	0	0	0.0002
Tagetes minuta	0.54	0	4.47	0.67	1.32	0.0004
Ficus sur	0	0	3.93	0	0	0.0002
Dichanthium foveolatus	1.85	0	3.67	1.44	3.24	0.0002
Allophylus abyssinicus	0	0	3.47	0	0	0.0002
Ekebergia capensis	0.62	0	2.93	0	0.04	0.0002
Teclea simplicifolia	0	0	2.87	0.33	0.92	0.0002
Dovyalis verrucosa	1.85	0	2.6	1	0.08	0.0018
Pinus patula	0	0	1.4	0	0	0.0002
Celtis africana	0.77	0	1.93	0	0.32	0.0002
Achyranthes aspera	0	0	1.13	0.11	0.08	0.0002
Cordia africana	0	0	1.2	0	0.08	0.0004
Debregeasia bicolar	0.08	0	1	0	0.04	0.0008
Acacia etbaica	0	0.13	0	6.56	0	0.0002
Acacia tortilis	0	0	0	5.44	0.6	0.0002
Cymbopogon pospischilii	1	3.47	0.53	5.33	0	0.0002
Themeda triandra	1.77	2.53	0	5.22	0	0.0002
Balanites aegyptiaca	0	0	0	5	0.28	0.0002
Acacia abyssinica	1.85	2.6	0.07	4.89	1.84	0.0006
Grewia mollis	0	0	0	2.44	0	0.0002
Rhus retinorrhoea	0	0	0.2	2.33	0.04	0.0006
Acacia asak	0	0	0	2	0.16	0.0012

Acacia seyal	0	0	0	1.89	0.04	0.0002
Helinus mystacinus	0.15	0	0	1.78	0.04	0.0014
Ziziphus spina-christi	0	0	0	1.67	0.04	0.0012
Kalanchoe lanceolata	0	0	0.07	1.56	0	0.0014
Rhus natalensis	0	0.2	1.13	1.44	0.68	0.0014
Dichrostachys cinerea	0	0	0	1.22	0.56	0.0014
Acacia mearnsii	0	0	0	1.11	0	0.0012
Monechma debile	0	0	0	1.11	0	0.0012
Olea europaea ssp. *caspidata*	4.69	2.13	2.13	2.33	5.96	0.0002
Cadia purpurea	0	0	0	1.11	4.32	0.0002
Cynodon dactylon	1.46	0	0.87	1.22	4.44	0.0002
Carissa spinarum	1.77	1.4	1.4	1.44	3.4	0.0002
Dodonaea angustifolia	1.54	0	0.27	1.11	3.28	0.0002
Cynanchum abyssinicum	0	0.2	0.53	0.22	2.64	0.0014
Rhus glutinosa	1.08	1.33	0.8	0.11	2.92	0.0014
Pittosporum viridiflorum	0.31	0	1.2	1.33	1.44	0.0014
Erica arborea	0	0	0.07	0	1	0.0014

2. *Cassipourea malosana* - *Myrica salicifolia* Community Type

This Community occurred in altitudinal range of 2292 to 2805 m. All the plots belonging to this community were drawn from the disturbed part of Gratkhassu Forest fragments. It comprised of 15 plots and 107 species that had association with this Community. *Myrica salicifolia, Cassipourea malosana* and *Dovyalis abyssinica* were among the dominant species of the type. The common tree species of this community were *Bersama abyssinica, Cupressus lusitanica, Acacia abyssinica* and *Osyris quadripartita*. *Rosa abyssinica, Clutia abyssinica, Clerodendrum myricoides, Conyza hypoleuca, Crotalaria lachnophora, Hypericum quartinianum, Otostegia fruticosa,* and *Rumex nervosus* were the most common shrub species. The herb layer of this community was dominated by *Verbascum sinaiticum, Andropogon abyssinicus, Crepis rueppellii, Caylusea abyssinica, Crassocephalum rubens, Euphorbia schimperiana, Kalanchoe marmorata, Medicago*

polymorpha, Pseudognaphalium oligandrum, Silene macrosolen, Cymbopogon pospischilii, Panicum maximum and *Themeda triandra*. This Community had dense herbaceous layer when compared to the other communities. *Clematis hirsuta, Cynanchum altiscandeus* and *Periploca linearifolia* were the common Liana found in this Community.

3. *Psydrax schimperiana - Podocarpus falcatus* Community Type

This Community comprised 15 plots which were all from the least disturbed part of Hugumburda Forest fragments. There were 135 species that were associated with this community and had many indicator species with significant indicator values (Table 10). It occurs in the altitudinal range of 2100 to 2280 m (having 2246 m mean altitude) on gentle slope of topography. *Podocarpus falcatus, Allophylus abyssinicus, Ficus sur* and *Ekebergia capensis* were among the dominant species and *Phytolacca dodecandra* was character species of the type. The other common tree species of the community include *Pinus patula, Celtis africana, Cordia africana, Maytenus arbutifolia, Teclea simplicifolia* and *Vernonia rueppellii*. *Psydrax schimperiana, Dovyalis verrucosa, Pavetta oliveriana, Pterolobium stellatum, Berberis holstii, Abutilon longicuspe, Debregeasia bicolar, Gomphocarpus fruticosus* and *Meriandra bengalensis* were the most common shrub species. The herb layer of this community was dominated by *Tagetes minuta, Achyranthes aspera, Laggera crispata, Cyperus alternitolius, Datura stramonium, Lycopersicon esculentum, Nepeta azurea, Pelargonium alchemilloides, Peperomia abyssinica, Rhynchosia elegans* and *Rumex nepalensis. Rhoicissus tridentata, Cynanchum abyssinicum* and *Jasminum abyssinicum* were the common lianas found in this Community.

4. *Acacia etbaica - Balanites aegyptiaca* Community Type

This Community consisted of 9 plots, which were all from Gratkhassu Forest fragments. The altitudinal range of this community was from 1814 to 2200 m (having 1948 m mean altitude) on gentle slope of topography. Totally, 110 species were associated with this community. There were also several indicator species with significant indicator values in this group and the major ones were *Acacia etbaica, Acacia tortilis, Balanites aegyptiaca, Grewia mollis, Acacia seyal, Cymbopogon pospischilii,* etc. (Table 10). The common tree species of this community were *Acacia seyal, Acacia abyssinica, Acokanthera schimperi, Acacia asak, Schinus molle, Acacia mearnsii* and *Eucalyptus camaldulensis*. The shrub layer was dominated by *Rhus retinorrhoea, Ziziphus spina-christi, Calotropis procera, Euclea schimperi, Grewia ferruginea, Grewia tembensis* and *Aloe debrana*. *Cymbopogon pospischilii, Themeda triandra, Monechma debile, Kalanchoe lanceolata, Arthraxon prionodes, Hyparrhenia hirta, Pennisetum thunbergii, Celosia anthelminthica, Barleria ventricosa* and *Silene macrosolen* were the common herb species. *Helinus mystacinus* and *Clematis hirsuta* were lianas found in this community.

5. *Olea europaea* ssp. *caspidata - Cadia purpurea* Community Type

This Community occurred in altitudinal range of 2137 to 2505 m on gentle slopes. It was comprised of 25 plots and 148 species that had association with this Community. *Olea europaea* ssp. *caspidata, Cadia purpurea* and *Rhus glutinosa* were among dominant species of the type. The common tree species of this community were *Discopodium penninervium, Pittosporum viridiflorum, Rhus natalensis, Teclea simplicifolia, Maesa*

lanceolata and *Juniperus procera.* There were also several shrub species in this group and the major ones were *Carissa spinarum, Dodonaea angustifolia, Pterolobium stellatum, Dichrostachys cinerea, Laggera tomentosa, Opuntia ficus-indica, Psydrax schimperiana, Sageretia thea, Senecio hadiensis, Ficus palmata, Astragalus atropilosulus,* etc. This community had very dense shrub layer when compared to the other communities. *Cynodon dactylon, Heracleum abyssinicum, Hypoestes forskaolii, Bidens biternata, Bidens pilosa, Argemone mexicana, Carduus nyassanus, Cynoglossum lanceolatum, Euphorbia petitiana, Ocimum urticifolium, Solanum incanum, Tagetes minuta* and *Dichanthium foveolatus,* were the common herb species. *Cynanchum abyssinicum, Rhoicissus tridentata, Toddalia asiatica* and *Tragia cinerea* were the common lianas found in this Community.

4.1.6.1. Species diversity, richness and evenness of the plant communities

Shannon`s diversity and Simpson`s diversity indices showed, Community 4 had the highest species diversity followed by communities 2 and 5 (Table 11). On the other hand, Community 5 had the highest number of species (130) and Community 3 the lowest (77). Community 2 followed by communities 4 and 3 had the highest evenness value. While, Community 5 had the least evenness value (Table 11).

Table 11 Species diversity, richness and evenness of the plant communities

Community	Species richness (S)	Shannon`s diversity index (H’)	Simpson`s diversity index (D)	Species evenness (E)
1	79	3.561	23.457	0.815
2	82	3.767	29.440	0.854
3	77	3.548	23.987	0.816
4	107	3.922	34.821	0.839
5	130	3.747	26.672	0.769

4.1.6.2. Ordination

The distribution of the study plots in Hugumburda-Gratkhassu Forest over environmental gradients was well explained using NMDS and CCA. For the NMDS ordination, the greatest reduction in stress with the fewest number of dimensions was achieved with a three-dimensional solution (McCune and Grace, 2006). The bi-plot of the first two NMDS axes indicates the partitioning of the five plant communities along the measured environmental gradients. The NMDS axis of compositional variation was significantly correlated with altitude, slope and human impact (Fig. 10). Altitude, human impact and aspect were positively correlated with axis one and are significant (at $P < 0.05$), while slope, EC and clay were negatively correlated to axis one and positively to axis 2 (Fig. 10).

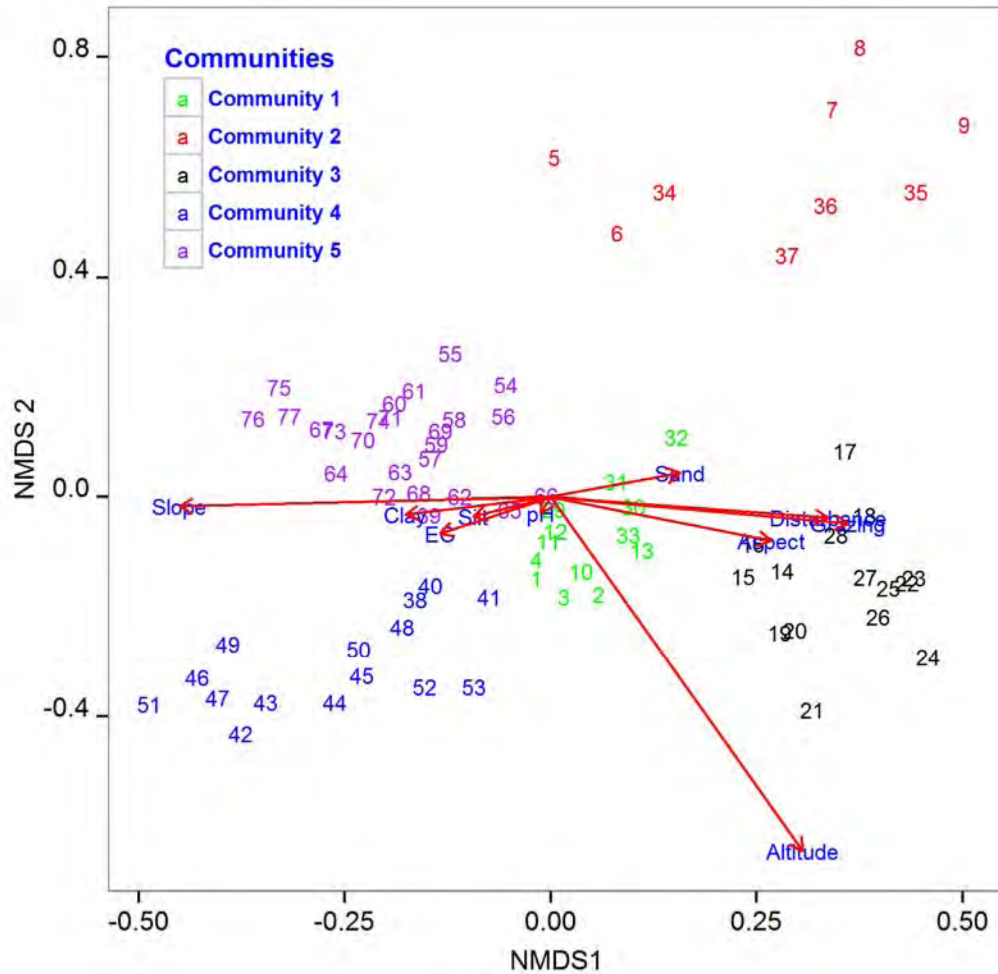

Fig. 10 Nonmetric multidimensional scaling (NMDS) ordination of the 77 plots sampled in Hugumburda-Gratkhassu Forest

The NMDS (Fig. 10) and CCA (Fig. 11) ordinations were comparable, the placement of environmental axes and stands being almost similar. The CCA ordination (Fig. 11) indicated that communities 1 and 4 occurred on sites with higher human and grazing disturbance with low EC and clay content, while Community 5 occurred on sites with higher slope, EC, clay and silt content.

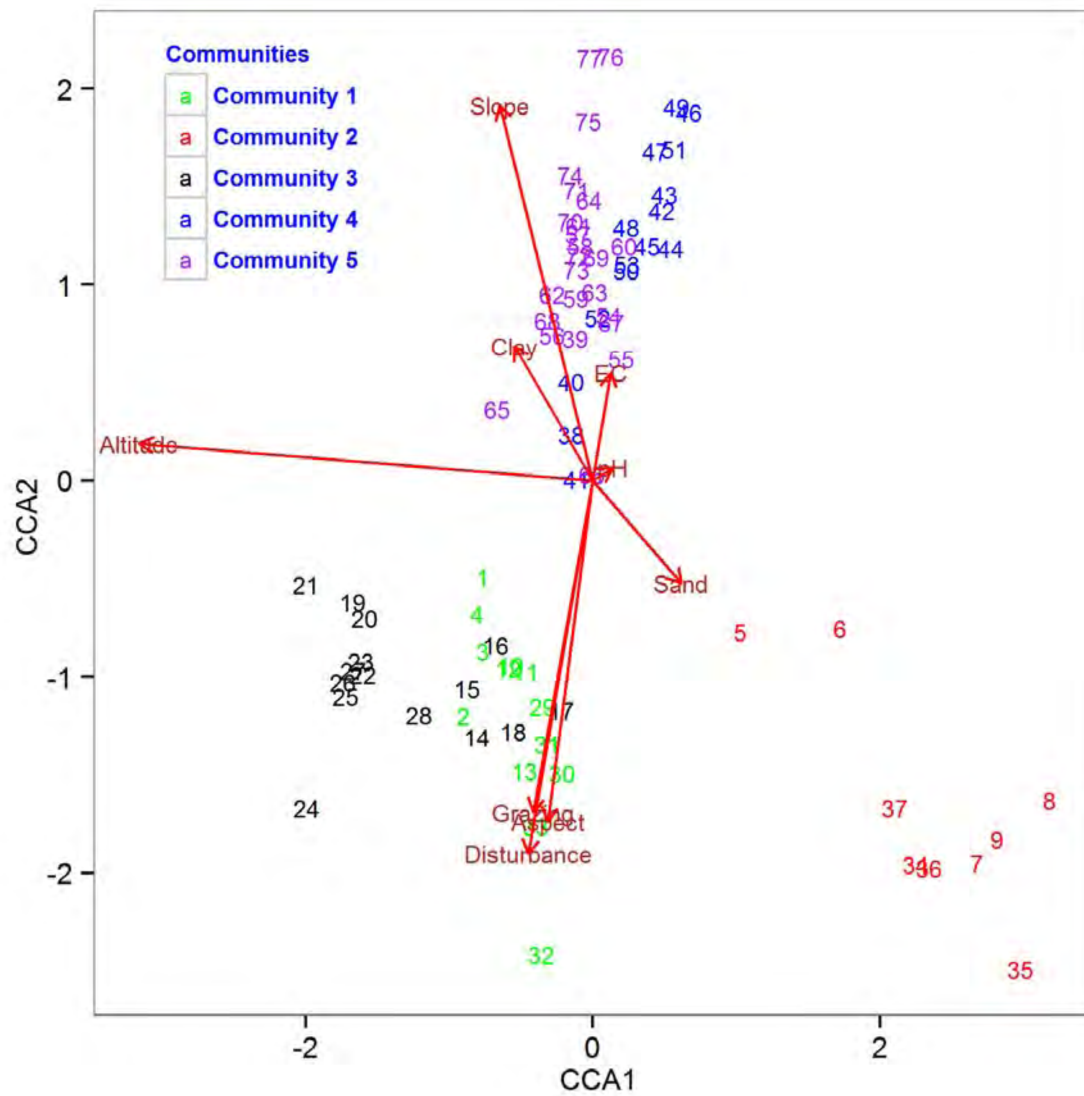

Fig. 11 CCA ordination diagram of plots and environmental variables

4.1.7. Species population structure

4.1.7.1. Growth forms

All species from the study area were categorized into four growth forms (Fig. 12). The growth forms classification adapted for the present analysis was very simple and general as compared to other classifications (Raunkiaer, 1934; Hedberg, 1964). The information

obtained from field and literature (Flora of Ethiopia and Eritrea.) were used for categorizing the recorded species into one of the four different growth forms, i.e. trees, shrubs, herbs and lianas.

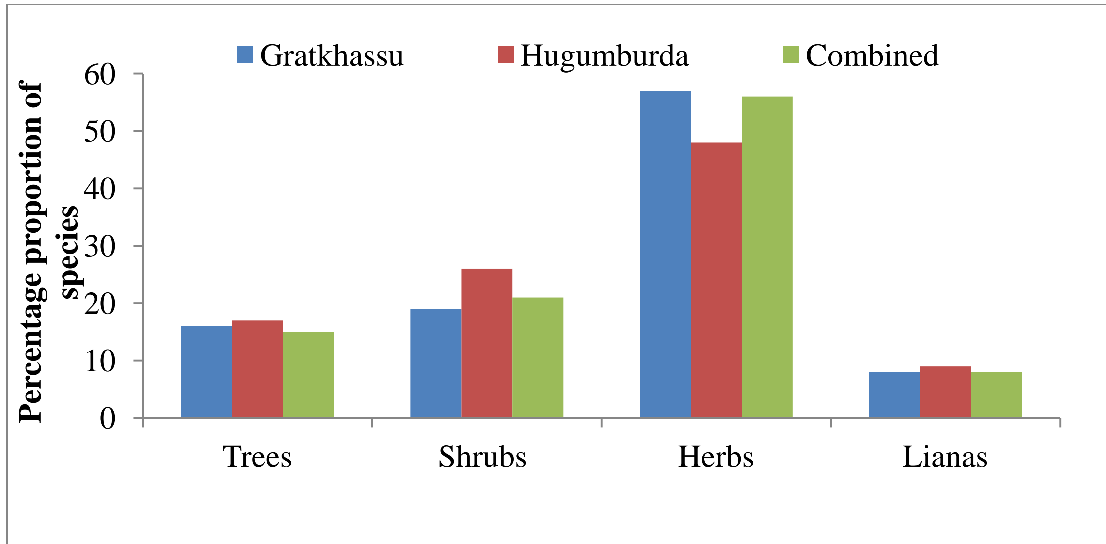

Fig. 12 Species diversity according to growth form in Hugumburda-Gratkhassu forest

As it was depicted in Fig. 12, herbaceous species accounted for 57% of the total number of species observed in Hugumburda-Gratkhassu Forest. Trees and shrubs together comprised 36% (trees 15% and shrubs 21%) of the total number of species. In Gratkhassu Forest fragments still herbs accounted for the upper hand (58%). Shrubs, trees and lianas contributed 19, 16 and 7 percent, respectively. In Hugumburda forest fragments, herbs, shrubs and trees comprised 49, 26 and 17 percent respectively, while lianas accounted for 8% (Fig. 12).

4.1.7.2. Height and DBH distribution pattern of woody species

The cumulative height class distribution pattern was an inverted J-shape, showing a decline in density of woody plants with increasing height classes (Fig. 13). For instance,

37.5% of the individuals were found in the first height class (≤ 2 m). The second height class (2.01 – 6 m) accounted for 35% of the woody plant individuals. The rest of the height classes (i.e. class 3 to class 8) contributed only 27.55% of the total individuals (Fig. 13). Similar to the general height class distribution of Hugumburda-Gratkhassu, in both Gratkhassu and Hugumburda forest remnants density of woody species generally declined with increase in height class (Fig. 13).

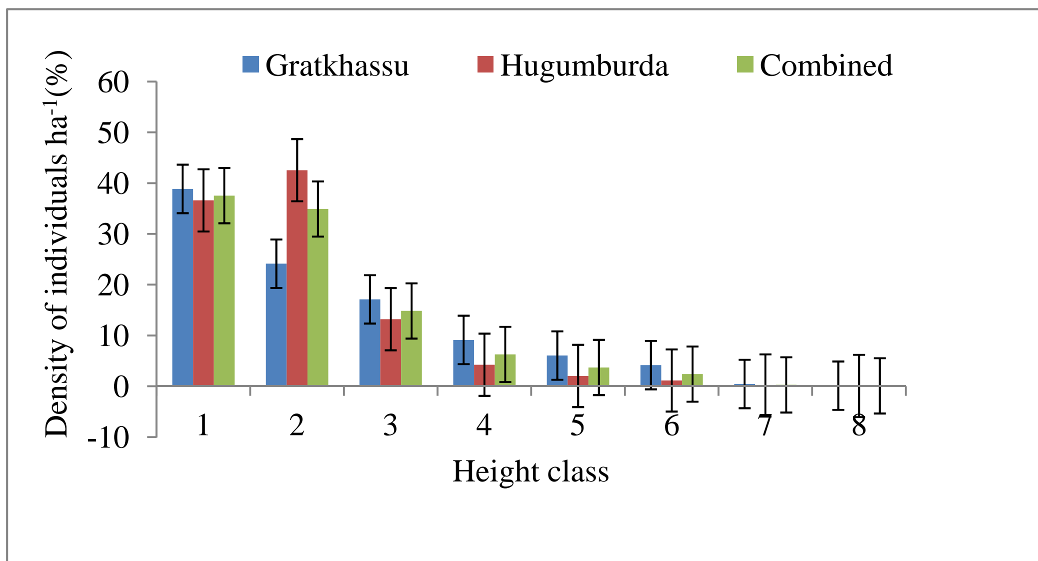

Fig. 13 Cumulative frequency distribution by height class of woody species. Height class: 1 = ≤ 2 m, 2 = 2.01 – 6 m, 3 = 6.01 – 10 m, 4 = 10.01 – 14 m, 5 = 14.01 – 18 m, 6 = 18.01 – 22 m, 7 = 22.01 – 26 m and 8 > 26 m (bars show density ± standard error)

The density distribution of woody species by DBH class showed a pattern similar to that seen for height classes (Fig. 14). For the whole data set, 38% of the individuals were found in class one (≤ 2 cm) while class two (2.01 – 7 cm) accounted for 26.16% (Fig. 14). The rest of the individuals (36.27%) were found in the higher classes (class 3-10). Generally, individuals from the upper diameter class (greater than 32 cm) contributed less

than 2% in both forest remnants. The third, fourth, fifth, sixth and seventh diameter classes contributed 15.4, 10.7, 7.5, 4.1 and 2.6 per cent of the individuals in Gratkhassu Forest fragments, respectively. While in Hugumburda, 14.5, 8.1, 3.7, 2.5 and 1.3 per cent of the individuals were found in diameter class 3, 4, 5, 6 and 7, respectively (Fig. 14).

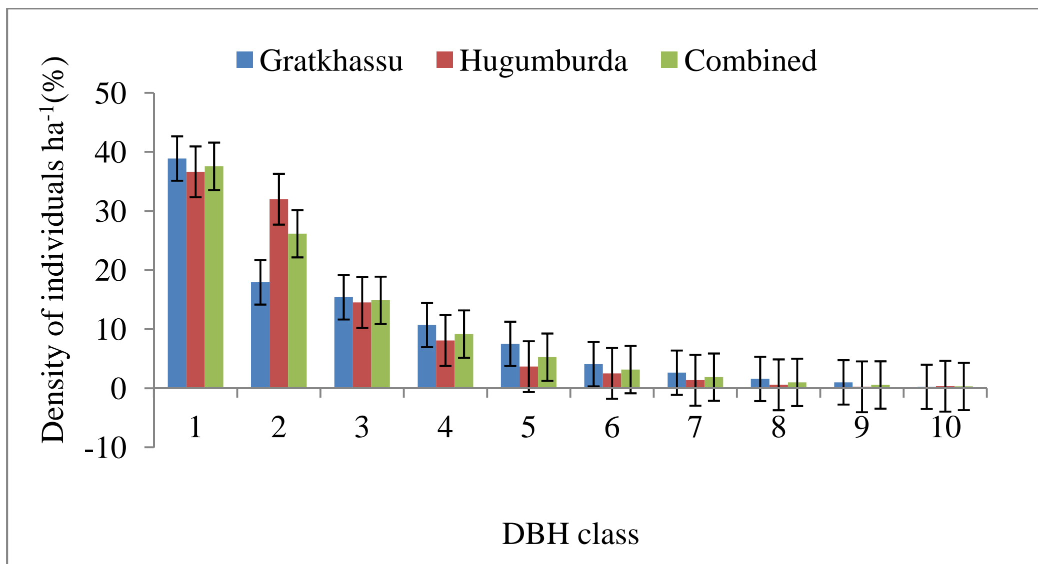

Fig. 14 Cumulative frequency distribution by diameter class of woody species. Diameter at breast height (DBH) class: 1 = ≤ 2 cm, 2 = 2.01 – 7 cm, 3= 7.01 – 12 cm, 4 = 12.01 – 17 cm, 5 = 17.01 – 22 cm, 6 = 22.01 – 27 cm, 7 = 27.01 – 32 cm, 8 = 32.01 – 37 cm, 9 = 37.1 – 42 cm and 10 > 42.01 cm (bars show density ± standard error)

Population structure of selected tree species

The population structure of selected tree species followed six general diameter class distribution patterns (Fig. 15). These include 1) inverted-J-shape, which showed that the species frequency was highest in the lower diameter classes and decreased gradually towards the higher classes e.g., *Podocarpus falcatus* (Fig. 15a), 2) broken inverted J-shape, e.g., *Ekebergia capensis* (Fig. 15b), 3) bell-shape, which showed a type of frequency distribution in which the number of individuals in the middle diameter classes was high and lower in the lower and higher diameter classes, e.g., *Maytenus arbutifolia* (Fig. 15c), 4) U-shape, which showed a high number of individuals in the lowest and highest diameter classes but a very low number in the intermediate classes, e.g., *Teclea simplicifolia* (Fig. 15d), 5) broken J-shape, which showed that few individuals in the lower diameter class and individuals were absent in the intermediate diameter class and relatively higher number of individuals in the highest diameter class. Example for such distribution was, *Hagenia abyssinica* (Fig. 15e), 6) upward ‘F’, which showed that individuals concentrate in the first (lower) and intermediate (third and fourth) diameter classes. The rest of the classes had almost equally few numbers of individuals, e.g., *Osyris quadripartita* (Fig. 15f).

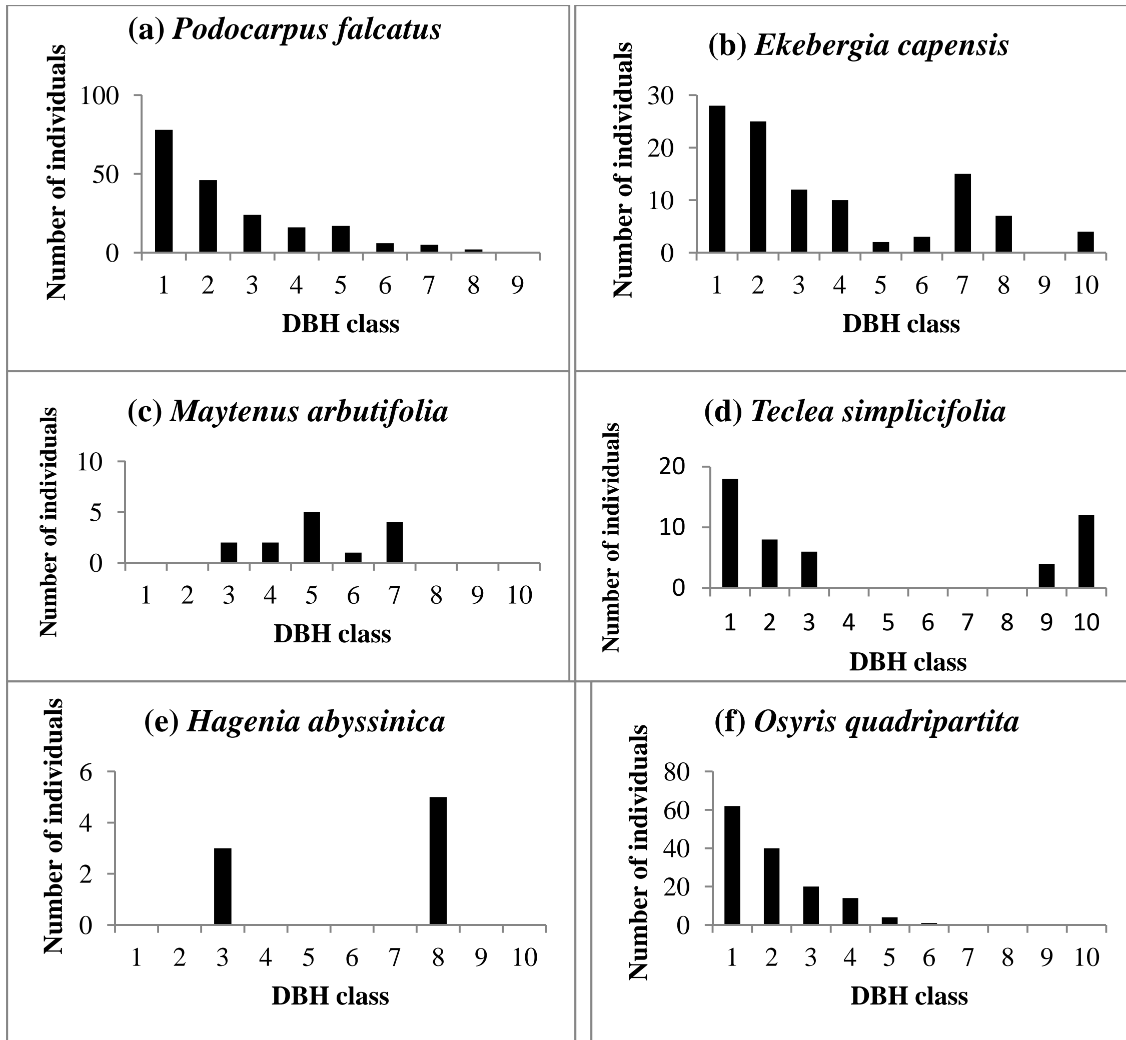

Fig. 15 Six representative patterns of population structure based on DBH. DBH class: 1 = ≤2 cm, 2 = 2.01 – 7 cm, 3= 7.01 – 12 cm, 4 = 12.01 – 17 cm, 5 = 17.01 – 22 cm, 6 = 22.01 – 27 cm, 7 = 27.01 – 32 cm, 8 = 32.01 – 37 cm, 9 = 37.1 – 42 cm and 10 >42.01 cm.

4.1.7.3. Basal area and density of woody species

The mean basal area per hectare of Gratkhassu and Hugumburda Forest remnants was shown in Fig. 16. As it was indicated in Fig. 16, Hugumburda Forest fragments had higher basal area per hectare (37.34 m^2) than that of Gratkhassu (30.92 m^2) Forest fragments. This difference was statistically significant at $P < 0.05$ level of significance.

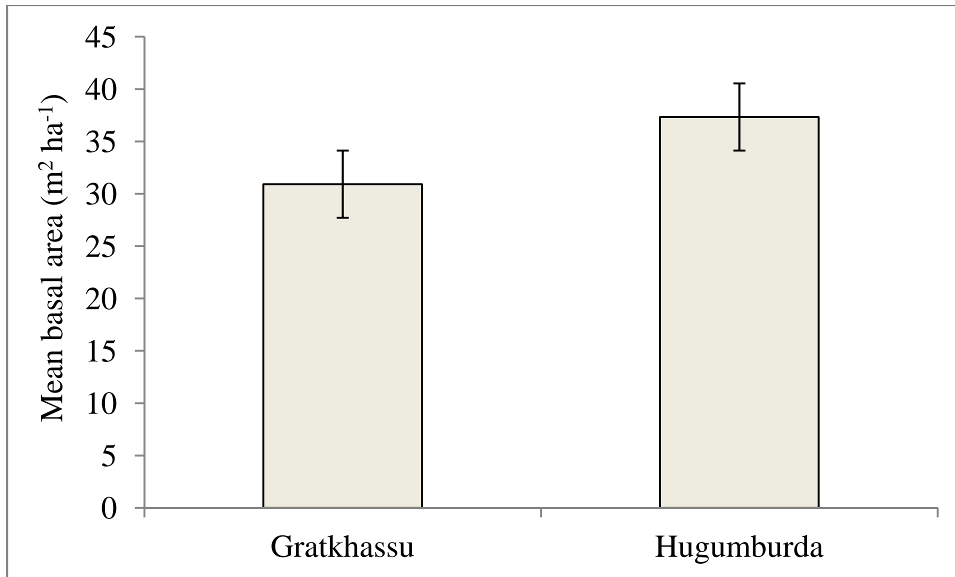

Fig. 16 Mean basal area per ha of Gratkhassu and Hugumburda Forest remnants (bars show mean ±standard error)

Similarly, the two forest remnants significantly differed in their densities of woody plants (Table 12). Higher woody plant species density per hectare (2,773 individuals per ha) was recorded in Hugumburda Forest fragments while in Gratkhassu, 2,135 individual per ha was recorded (Table 12). Mean density per plot recorded in Hugumburda was 111 individuals while in that of Gratkhassu it was 86 (Table 12).

Table 12 Density of woody plants in Gratkhassu and Hugumburda forest remnants

Characteristics	**Gratkhassu**	**Hugumburda**	**F value**	**P value**
Total quadrats	37	40		
Total density of woody plants	3,162	4,440	26.828	0.004
Mean Density per quadrat	86	111	80.075	0.001
Density ha^{-1}	2,135	2,773	37.463	0.004

4.1.8. Regeneration status of Hugumburda-Gratkhassu Forest

Regeneration status analysis result of Hugumburda-Gratkhassu Forest on densities of age classes of woody species is shown in Fig. 17. A total of 11,000 ± 1,500 seedlings, 8,000 ± 1090 saplings and 3,500 ± 477 ha^{-1} mature plants were recorded for all woody species. From the total seedlings, 6,982, 3,711 and 107 ha^{-1} belonged to shrubs, trees and lianas respectively. Of the total saplings recorded in the Forest, 5,224, 2,583 and 217 ha^{-1} were for shrubs, trees and lianas respectively. Similarly, from the total mature plants, 2,466, 927 and 103 ha^{-1} belonged to shrubs, trees and lianas respectively (Fig. 17).

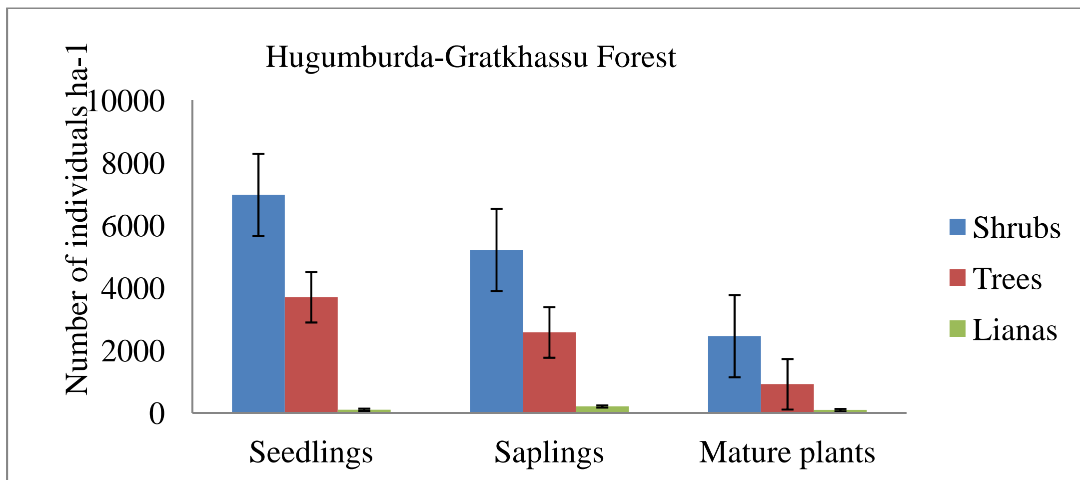

Fig. 17 Age class distribution of woody species in Hugumburda-Gratkhassu Forest (bars show density ± standard error)

4.1.8.1. Comparison of regeneration status among the forest remnants

When result of the two forest remnants was analyzed separately, in Gratkhassu a total of 55 woody species of which 24 were tree species, 26 were shrubs and 5 liana species distributed in 44 genera and 33 families were analyzed. A total of 5,000 ± 700 seedlings, 3,600 ± 490 saplings and 1,500 ± 200 ha^{-1} mature plants were recorded for the above growth forms. From the total seedlings, 3,376, 1,394 and 78 ha^{-1} belonged to shrubs, trees

and lianas respectively. Of the total saplings recorded in Gratkhassu, 2,607, 955 and 72 ha^{-1} were for shrubs, trees and lianas respectively. Similarly, from the total mature plants, 1,160, 286 and 43 ha^{-1} belonged to shrubs, trees and lianas respectively (Fig. 18a).

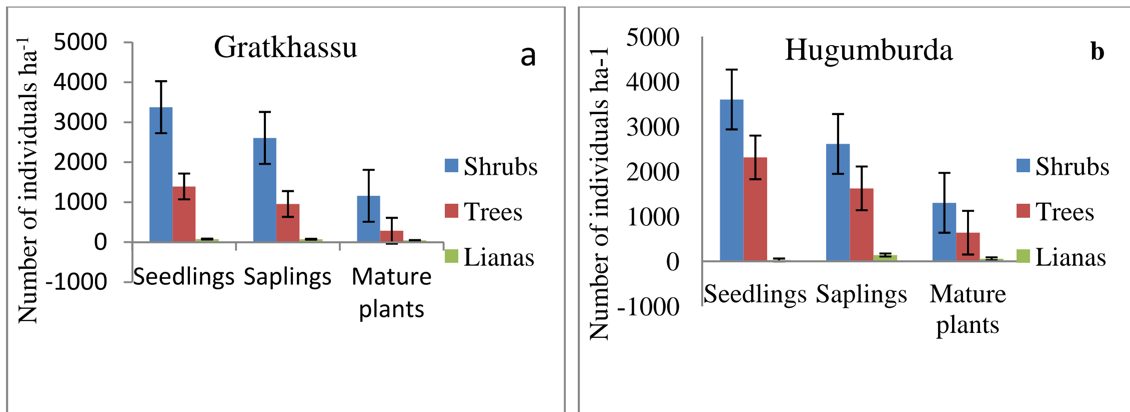

Fig. 18 Age class distribution of woody species in Gratkhassu (a) and Hugumburda (b) forest remnants (bars show density ± standard error)

In Hugumburda, a total of 80 woody species of which 30 were tree species, 40 were shrubs and 10 liana species distributed in 72 genera and 46 families were analyzed. A total of 6,000 ± 800 seedlings, 4,400 ± 600 saplings and 2,000 ± 270 ha^{-1} mature plants were recorded for the above growth forms. From the total seedlings, 3,600, 2,317 and 30 ha^{-1} belonged to shrubs, trees and lianas respectively. Of the total saplings recorded in Hugumburda, 2,617, 1,628 and 145 ha^{-1} were for shrubs, trees and lianas respectively. Similarly, from the total mature plants, 1,306, 641 and 60 ha^{-1} belonged to shrubs, trees and lianas respectively (Fig. 18b).

When the seedling and sapling density of the two forest remnants analyzed together Fabaceae was the richest family represented by 15 species and followed by Lamiaceae

consisting of 6 species. Correspondingly, when the seedling and sapling density of the two forest remnants was analyzed separately Fabaceae was still the top species rich family in both forest remnants (Table 13). In Hugumburda the second species richest family was Asteraceae while in Gratkhassu Anacardiaceae takes the second place containing four species (Table 13).

Table 13 Families represented by two or more species in the seedling and sapling population of Gratkhassu and Hugumburda Forest remnants

	Gratkhassu		**Hugumburda**		
Family	**Number of species**	**Number of genera**	**Family**	**Number of species**	**Number of genera**
Fabaceae	11	6	Fabaceae	11	9
Anacardiaceae	4	2	Asteraceae	5	3
Lamiaceae	3	3	Lamiaceae	4	4
Asclepiadaceae	2	2	Oleaceae	4	2
			Anacardiaceae	3	1

Comparison of regeneration status in the two forest remnants were made based on the ratios of the different age classes of tree species. Ratio/proportion of seedlings to saplings showed the status of regeneration of a given forest ecosystem. Ratios of the different age classes were presented in Table 14.

Table 14 Ratios of different size classes in the two forest remnants

Forest remnants	Seedlings density (a)	Saplings density (b)	Mature plants density (c)	(a/b)	(a/c)	(b/c)
Gratkhassu	1,393.57	955.07	285.81	1.50	4.88	3.34
Hugumburda	2,316.96	1,627.98	641.25	1.42	3.61	2.54

Variation in the number of seedlings to saplings in Gratkhassu and Hugumburda forest remnants were statistically significant ($P < 0.05$) which was reflected in their ratios of seedlings to saplings. The ratio of seedlings to saplings was high and almost similar in both forest fragments. The ratio of seedlings to mature plants was high in Gratkhassu (4.88) and low in Hugumburda (3.61). Similarly, the ratio of saplings to mature plants was high in Gratkhassu (3.34) and low in Hugumburda (Table 14).

To apply regeneration analysis for conservation priority setting, tree and selected shrub species recorded in the study area were grouped into three categories, based on the density of seedlings (Table 15) following Simon Shibru and Girma Balcha (2004) and Getinet Masresha (2014). Therefore, those species with no or one seedling were grouped under category I, whereas those species with seedlings between one and 15 were categorized under category II and lastly species with seedling number greater than or equal to 15 were grouped under category III.

In Gratkhassu Forest fragments six species, *Acacia asak, Senna sinqueana, Clerodendrum myricoides, Clutia abyssinica, Hypericum quartinianum* and *Schinus molle* were included in category I, whereas in category II eight species, *Acacia seyal,*

Bersama abyssinica, Celtis africana, Ehretia cymosa, Grewia mollis, Rhus glutinosa, Rhus natalensis and *Teclea simplicifolia* were included.

In Hugumburda Forest fragments eight species, *Cadaba farinosa, Colutea abyssinica, Cordia africana, Discopodium penninervium, Hagenia abyssinica, Phytolacca dodecandra, Pinus patula* and *Rhus glutinosa* were placed in category I and seven species, *Acokanthera schimperi, Erica arborea, Ficus palmata, Ficus sur, Podocarpus falcatus, Vernonia amygdalina* and *Vernonia rueppellii* were included in category II.

When the whole data set of the entire study area is considered (Table 15) seven species, *Buddleja polystachya, Cordia africana, Euphorbia ampliphylla, Hagenia abyssinica, Hypericum quartinianum, Phytolacca dodecandra* and *Vernonia rueppellii* were included in category I, whereas in category II seventeen species, *Acacia asak, Acacia seyal, Acokanthera schimperi, Becium grandiflorum, Senna sinqueana, Discopodium penninervium, Erica arborea, Ficus palmata, Ficus sur, Grewia ferruginea, Grewia mollis, Hypericum revolutum, Maytenus senegalensis, Pinus patula, Podocarpus falcatus, Rhus retinorrhoea* and *Vernonia amygdalina* were included and the remaining species were placed in category III (Table 15).

Table 15 Tree and shrubby tree species categories for conservation priorities in Hugumburda-Gratkhassu Forest

Regeneration status		
Category I	**Category II**	**Category III**
Seedling density (0-1)	**Seedling density (2-14)**	**Seedling density (≥15)**
Buddleja poylstachya	*Acacia asak*	*Acacia abyssinica*
Cordia africana	*Acacia seyal*	*Acacia etbaica*
Euphorbia ampliphylla	*Acokanthera schimperi*	*Acacia tortilis*
Hagenia abyssinica	*Becium grandiflorum*	*Allophylus abyssinicus*
Hypericum quartinianum	*Senna sinqueana*	*Balanites aegyptiaca*
Phytolacca dodecandra	*Discopodium penninervium*	*Bersama abyssinica*
Vernonia rueppellii	*Erica arborea*	*Cassipourea malosana*
	Ficus palmata	*Celtis africana*
	Ficus sur	*Dovyalis abyssinica*
	Grewia ferruginea	*Ekebergia capensis*
	Grewia mollis	*Juniperus procera*
	Hypericum revolutum	*Maytenus undata*
	Maytenus senegalensis	*Myrica salicifolia*
	Pinus patula	*Nuxia congesta*
	Podocarpus falcatus	*Olea europaea*
	Rhus retinorrhoea	*Olinia rochetiana*
	Vernonia amygdalina	*Osyris quadripartita*
		Pittosporum viridiflorum
		Rhus natalensis
		Teclea simplicifolia

4.1.8.2. Regeneration pattern of selected woody species

On the basis of frequency distribution of the different size classes, the existing regeneration behavior of tree and selected shrub species of the study area were represented by six regeneration patterns (Fig. 19a-f). The first pattern was represented by *Nuxia congesta*, the pattern that exhibited inverted J-shaped where densities of seedlings and saplings exceeded densities of mature trees (Fig. 19a). This pattern represents good regeneration and recruitment. The second pattern was represented by *Ekebergia capensis*, the pattern that exhibited U-shaped where densities of seedlings and mature trees exceeded densities of saplings (i.e. the intermediate class contains few numbers of individuals) (Fig. 19b). The third pattern was represented by *Maytenus undata*, the pattern that exhibited Bell-shape where densities of saplings exceeded densities of seedlings and mature trees (Fig. 19c). The fourth pattern was represented by *Heteromorpha trifoliata*, the pattern that exhibited J-shaped where densities of mature trees and saplings exceeded densities of seedlings (Fig. 19d). The fifth pattern was represented by *Cordia africana*, the pattern that exhibited broken J-shaped, where densities of mature trees exceeded densities of seedlings and saplings (Fig. 19e). The sixth pattern was represented by *Ehretia cymosa*, the pattern that exhibited upward F-shaped, where individuals concentrate in the first (seedlings) and second (saplings) size class and few or no individuals in the matured size class (Fig. 19f).

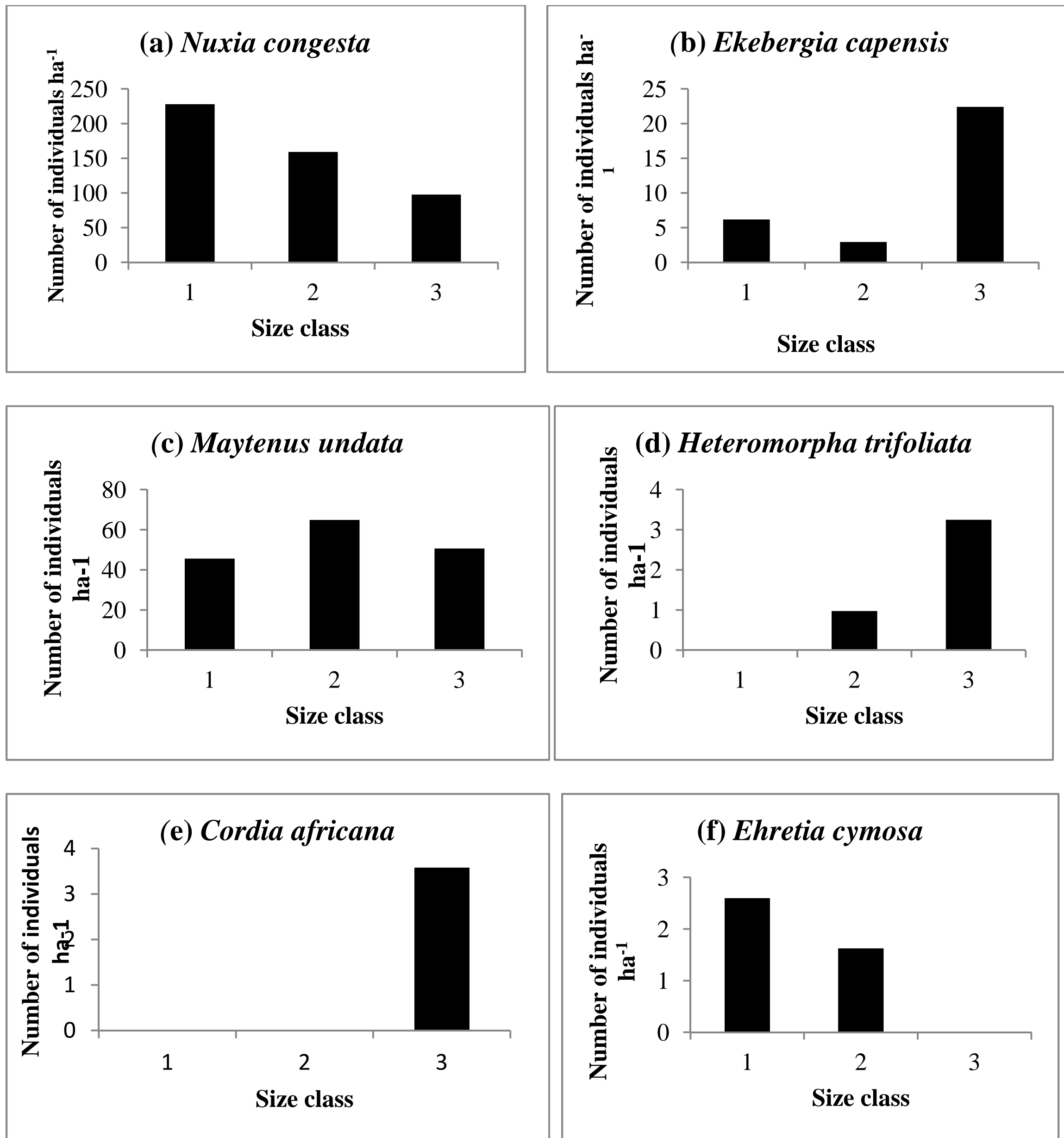

Fig. 19 Six representative regeneration patterns in Hugumburda-Gratkhassu Forest

4.2. Edge effect on plant diversity and population structure

4.2.1. Edge effect on species composition and abundance along the edge - intermediate - interior gradient

A total of 288 plant species belonging to 214 genera and 73 families were recorded from the three studied edge - intermediate - interior habitats. Asteraceae was the most dominant family with 35 species followed by Fabaceae and Poaceae (27 species each), in the studied edge - intermediate - interior gradient of the forest (Fig. 20). The most frequent species in the edge - intermediate - interior gradient study were *Juniperus procera* with a relative frequency of 86.74%, followed by *Hypoestes forskaolii* and *Carissa spinarum* with a relative frequency of 85.85% and 80.95% respectively. Of the total 288 species observed in this study, 39.24% (113 species) were common in all the three studied forest habitats while 13.89% (40 species), 10.42% (30 species), and 4.51% (13 species) occurred exclusively to the edge, intermediate and interior habitats, respectively (Appendix 5).

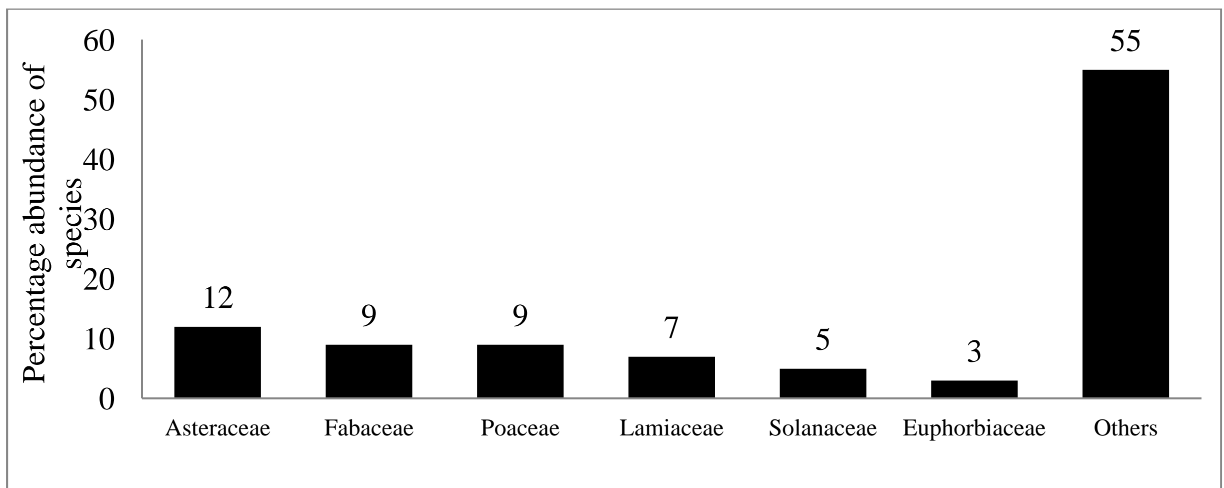

Fig. 20 Percentage frequencies of species in the top six species rich families recorded in the edge - intermediate - interior gradient

4.2.2. Plant species diversity along the edge - intermediate - interior gradient

As indicated in Table 16 all Fisher α, Shannon diversity index and Simpson index were high in the intermediate habitat than in the edge and interior habitats. This showed that the forest intermediate habitat was the most diverse community with significantly higher Shannon diversity index (4.31), Fisher‟s α (60.72) and Simpson Inv Mean (42.15) compared to edge and interior habitats, which did not differ significantly ($P > 0.05$). Though evenness did not differ significantly between habitats, the edge community had relatively higher evenness (0.58) than the other habitats (Table 16).

Table 16 Various diversity parameters calculated for edge - intermediate - interior gradient

Diversity measurement	Edge	Intermediate	Interior
Fisher‟s α	56.31	60.72	54.14
Shanon diversity (H')	4.24	4.31	4.24
Evenness index (E')	0.576	0.561	0.33
Simpson Inv Mean	39.75	42.15	41.22

4.2.3. Species richness and accumulation curves along the edge - intermediate - interior gradient

Rarefaction curve is useful to test whether there is significance difference in species richness between different sites or not. As it was depicted in Fig. 21, the observed species accumulation curve of interior and intermediate habitats was outside of the 95% confidence interval of edge habitat revealing that they had significantly higher species richness than that of the edge habitat (Figs. 21 and 22).

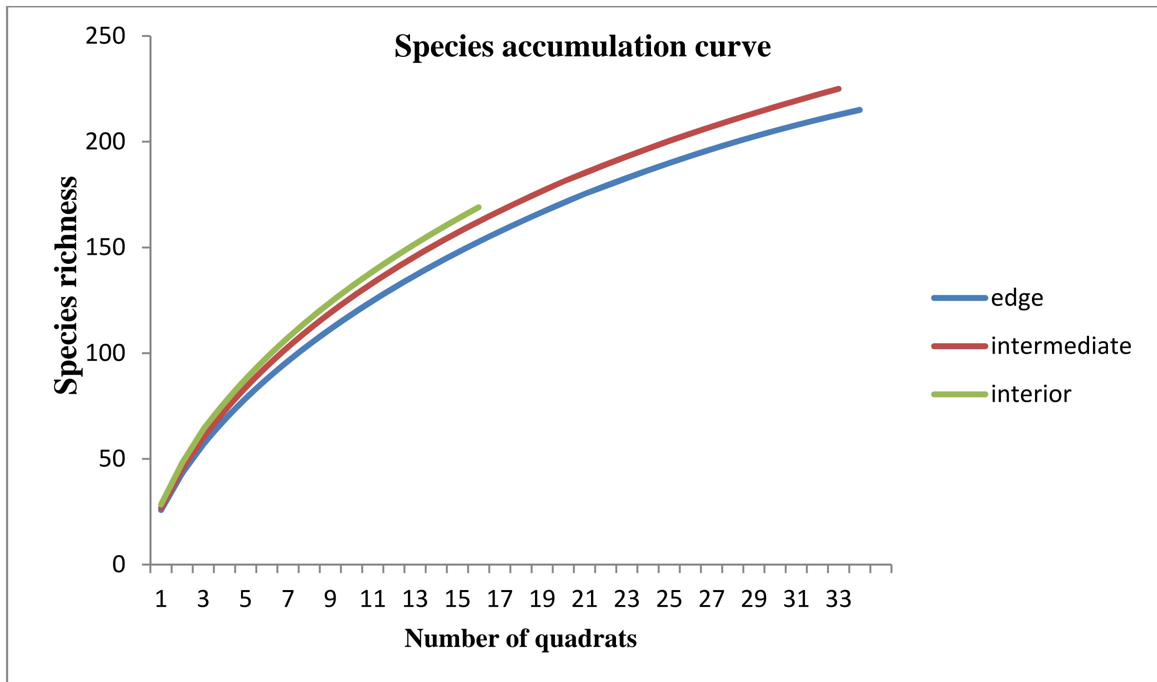

Fig. 21 Species accumulation (rarefaction) curves and 95% confidence intervals along the edge - intermediate - interior gradient

As it is shown in Table 17, the second order Jackknife species richness estimator projected higher species richness in each studied habitat than the other estimators. The species richness was significantly higher in the intermediate habitat compared to edge and interior habitats, which did not differ significantly. Bootstrap estimator gave the lowest species richness estimate for each studied habitat (Table 17).

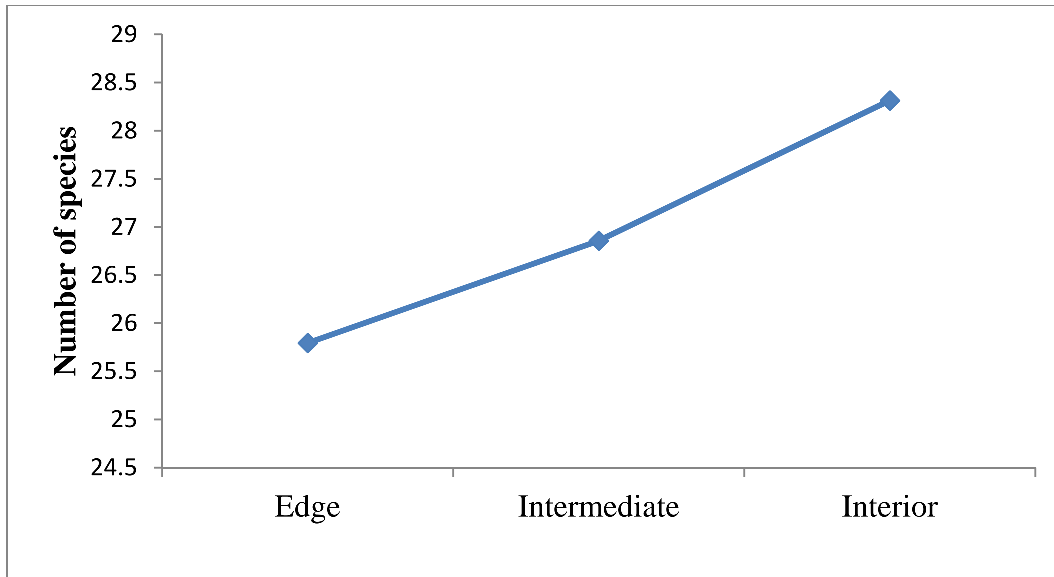

Fig. 22 Number of species recorded along the edge - intermediate - interior habitats

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Table 17 Species richness estimation (based on different estimators) of the edge - intermediate -interior gradient

Estimator	Edge	Intermediate	Interior
Number of samples	34	33	16
Sobs	215	225	174
ACE	282.69	326.75	274.88
Chao 1	266.84	332.6	272.59
Chao 2	256.1	318.37	282.44
Jackknife 1	289.74	313.24	251.5
Jackknife 2	298.32	359.6	302.37
Bootstrap	252.47	264.94	204.63
Species collection degree	72.1 - 83.95%	62.56 - 84.92%	57.54 - 85.1%

4.2.4. Density and basal area of woody species along the edge - intermediate - interior gradient

The density of woody plant species between studied forest habitats varied significantly ($P < 0.05$) ranging from 1,368 individual per ha (at the edge) to 1,898 individual per ha (at the interior) with an overall average of 1,552 individual per ha when all habitats are pooled together (Fig. 23). Post hoc Tukey's HSD test ($P < 0.05$) confirmed the interior habitat to have significantly higher density than the edge and intermediate, which did not differ considerably ($P > 0.05$) in their woody species densities.

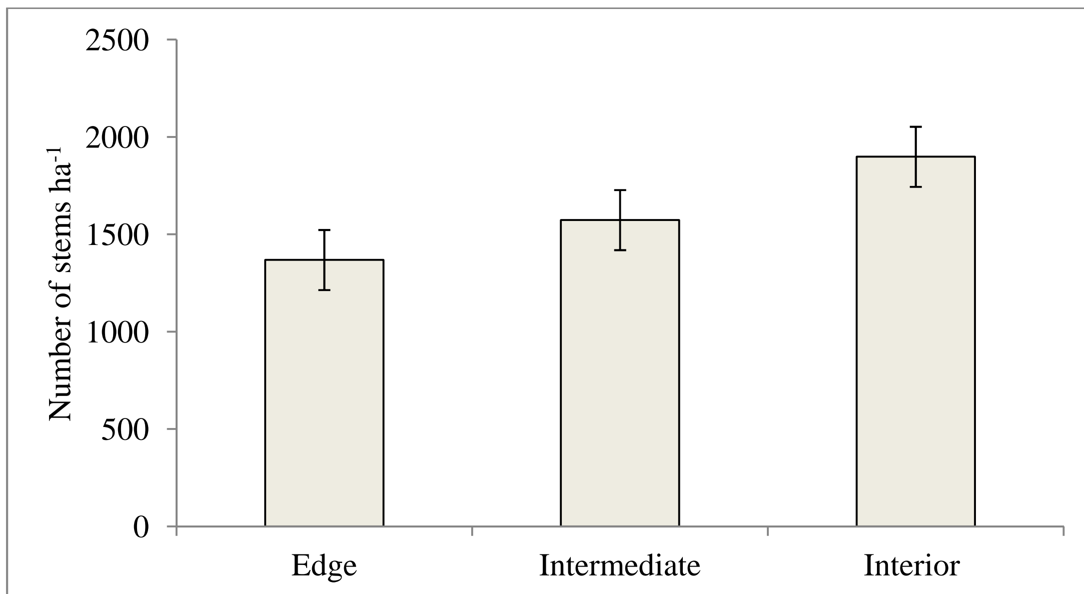

Fig. 23 Mean woody species density per ha of the edge - intermediate - interior habitats (bars show mean ±standard error)

The basal area of woody species (Fig. 24) differed significantly ($P < 0.05$) between the studied three habitats ranging from 21.97 m^2 per hectare (at the edge) to 33.62 m^2 per hectare (at the interior). Post hoc Tukey's HSD test ($P < 0.05$) revealed the interior of the forest to have a significantly higher basal area compared to the edge and intermediate habitats, which also did not differ considerably ($P > 0.05$) in their basal areas. The species

with the highest basal areas were *Juniperus procera* and *Nuxia congesta* with a relative basal area of 5.08% and 1.96%, respectively.

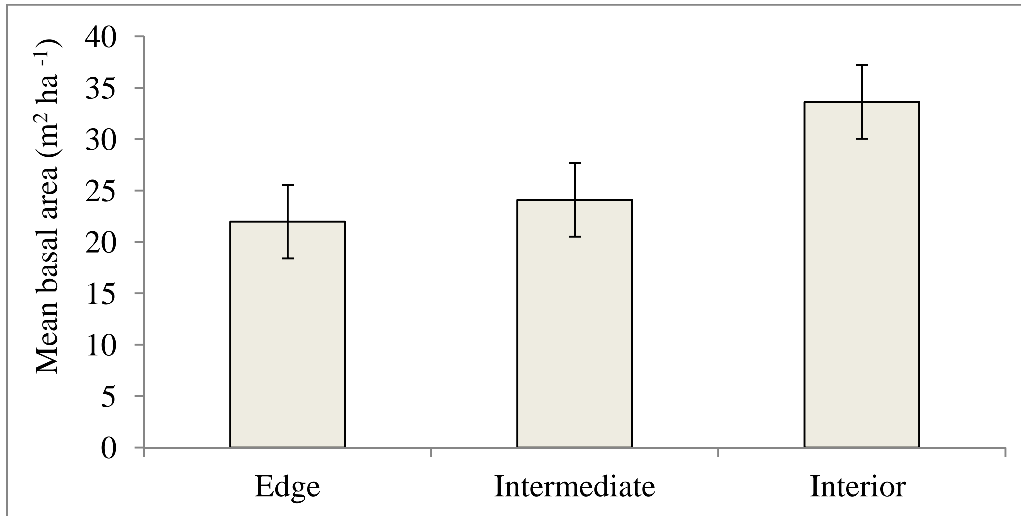

Fig. 24 Mean basal area per ha of the edge - intermediate - interior habitats (bars show mean ±standard error)

The numbers of dead and fallen trees were strongly influenced by proximity to the edge (Fig. 25). The number of standing dead trees (snags) and fallen trees (logs) (Appendix 10) decreased with distance from edge to interior habitat. These patterns were due primarily to snags (Fig. 25), whose number declined from high levels (36 stems per ha) close to the edge to very low levels (5 stems per ha) at the interior. Correspondingly, along the distance into the interior of the forest the canopy cover was most open at the edge than the intermediate and interior habitats. The edge habitat had a higher percentage of canopy openness and the effects of conditions at the edge were different as compared to the interior.

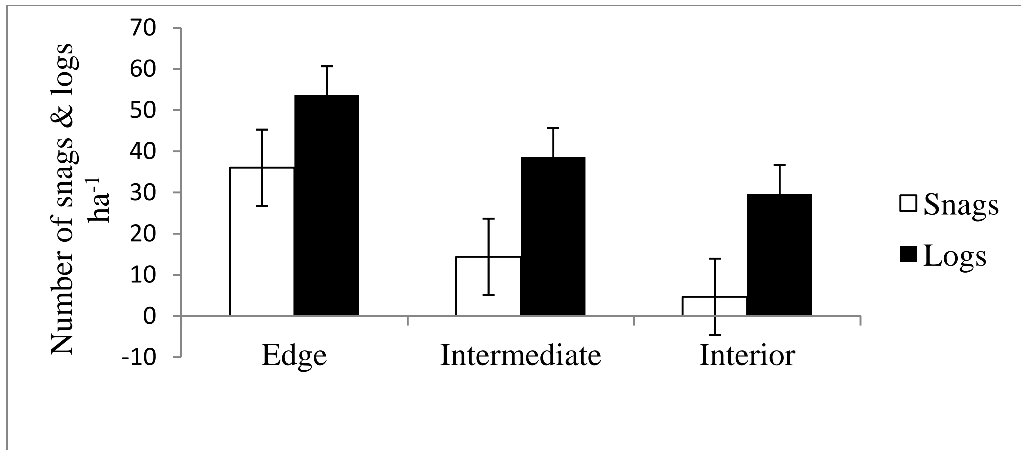

Fig. 25 Mean numbers of standing dead trees (snags) and fallen boles (logs) per ha in the edge - intermediate - interior habitats (bars show mean±standard error)

The numbers of seedlings (≤ 100 cm) and saplings (100 -200 cm) significantly differed (at $P < 0.05$) with distance from the edge into the interior habitat (Fig. 26). The numbers of seedlings and saplings increased with distance, ranging from 2,739 seedlings and 2,275 saplings per hectare (at the edge) to 3,542 seedlings and 2,806 saplings per hectare (at the interior).

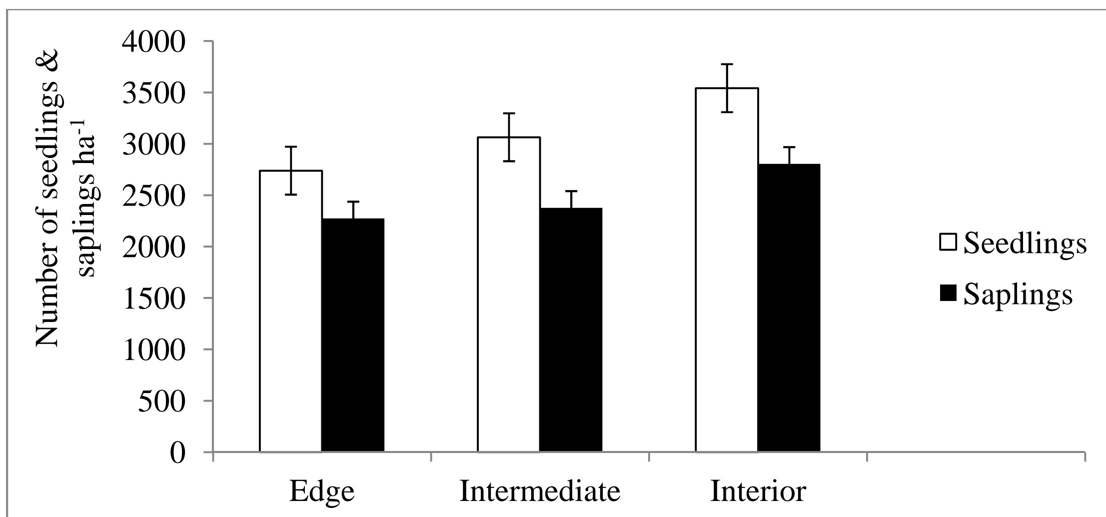

Fig. 26 Mean numbers of seedlings and saplings per ha in the edge - intermediate - interior gradient (bars show mean±standard error)

4.3. Dynamics in Land use\Land cover Types

4.3.1. Accuracy assessment

Producer accuracy, user accuracy, overall accuracy and kappa coefficient were used to indicate how much the classified image represented the actual features on the ground. Finally, overall classification accuracy of 90.53% and overall Kappa statistics 0.88 was achieved (Table 18), which was feasible for further application. The reasons for the errors may include the similarity of reflectance of settlement, cultivated areas and bare lands. Moreover, there was also an error in the classification of areas covered by forest, shrub land and grassland due to the reflectance proximity.

Table 18 The overall classification accuracy and Kappa statistics

Class Name	Reference Totals	Classified Totals	Number Correct	Producers Accuracy	Users Accuracy	Kappa Index
Forest	12	13	12	100.00%	92.31%	0.91
Shrub land	28	28	26	92.86%	92.86%	0.89
Grassland	15	15	13	86.67%	86.67%	0.84
Cultivated land	19	20	18	94.74%	90.00%	0.88
Settlement	5	4	3	60.00%	75.00%	0.74
Bare land	16	15	14	87.50%	93.33%	0.92
Totals	95	95	86			
Overall Classification Accuracy = 90.53%						
Overall Kappa Statistics = 0.8816						

4.3.2. Land use\land cover classification maps and outputs

The land use\land cover types of the study area derived from the maps of 1973, 1986 and 2011 were categorized as: forest, shrub land, grassland, cultivated land, settlement and bare land. The land use\land cover distribution map for the year 1973 (Fig. 28) and area coverage for each land use\land cover (Table 19), showed forest land accounted for 7,353 ha (30.4%) from the total study area (24,176 ha). Shrub land and grasslands occupied about 5,518 ha (23%) and 3,193 ha (13%), respectively.

Table 19 Land use\land cover classes and their corresponding areas for 1973, 1986 and 2011

Landuse\ landcover types	**1973 Landcover (ha)**	**% of landcover 1973**	**1986 Landcover (ha)**	**% of landcover 1986**	**2011 Landcover (ha)**	**% of landcover 2011**
Forest	7352.8	30.4	8042.9	33.3	8992.6	37.2
Shrub land	5518.3	22.8	5618.6	23.2	6040.4	24.9
Grassland	3192.6	13.2	2893.7	12.0	2179.6	9.1
Cultivated land	4719.7	19.5	4889.0	20.2	5676.0	23.5
Settlement	1533.9	6.3	1003.8	4.2	820.7	3.4
Bare land	1858.4	7.7	1727.8	7.1	466.5	1.9
Total	24,175.8	100	24,175.8	100	24,175.8	100

The land use\land cover classification for 1986 from TM satellite image (Fig. 28) showed that forest and shrub land accounted for 8043 ha (33%) and 5618 ha (23%) respectively, while grassland and cultivated land took the share of 2,894 ha (12%) and 4,889 ha (20%) respectively (Table 19). The remaining area was covered with settlement and bare land. Most portions of the land use\land cover class was forest land during this period.

The land use\land cover distribution map and the area coverage for each land use\land cover types for the year 2011 are presented in Fig. 28 and Table 19. As shown in Table 19, the areal coverage of forest and shrub land accounted for 8,993 ha (37%) and 6,040 ha (25%) respectively from the total study area. The grassland, cultivated land, settlement and bare land occupied about 9%, 24%, 3% and 2%, respectively (Table 19).

The results of land use\land cover map (Table 19 and Fig. 28) showed that the area of forest, shrub land and cultivated land increased in both periods whereas grassland, settlement and bare land decreased in both periods. This was just the general impression of the landcover dynamics based on comparison of individual landcover maps. The trend in area coverage of each land use\land cover types for each image is shown in Fig. 27.

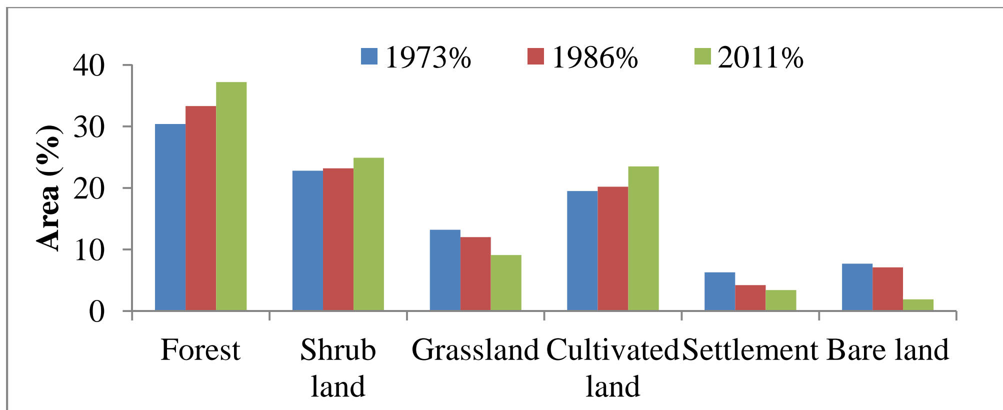

Fig. 27 Trends in land use\land cover extent during the reference periods

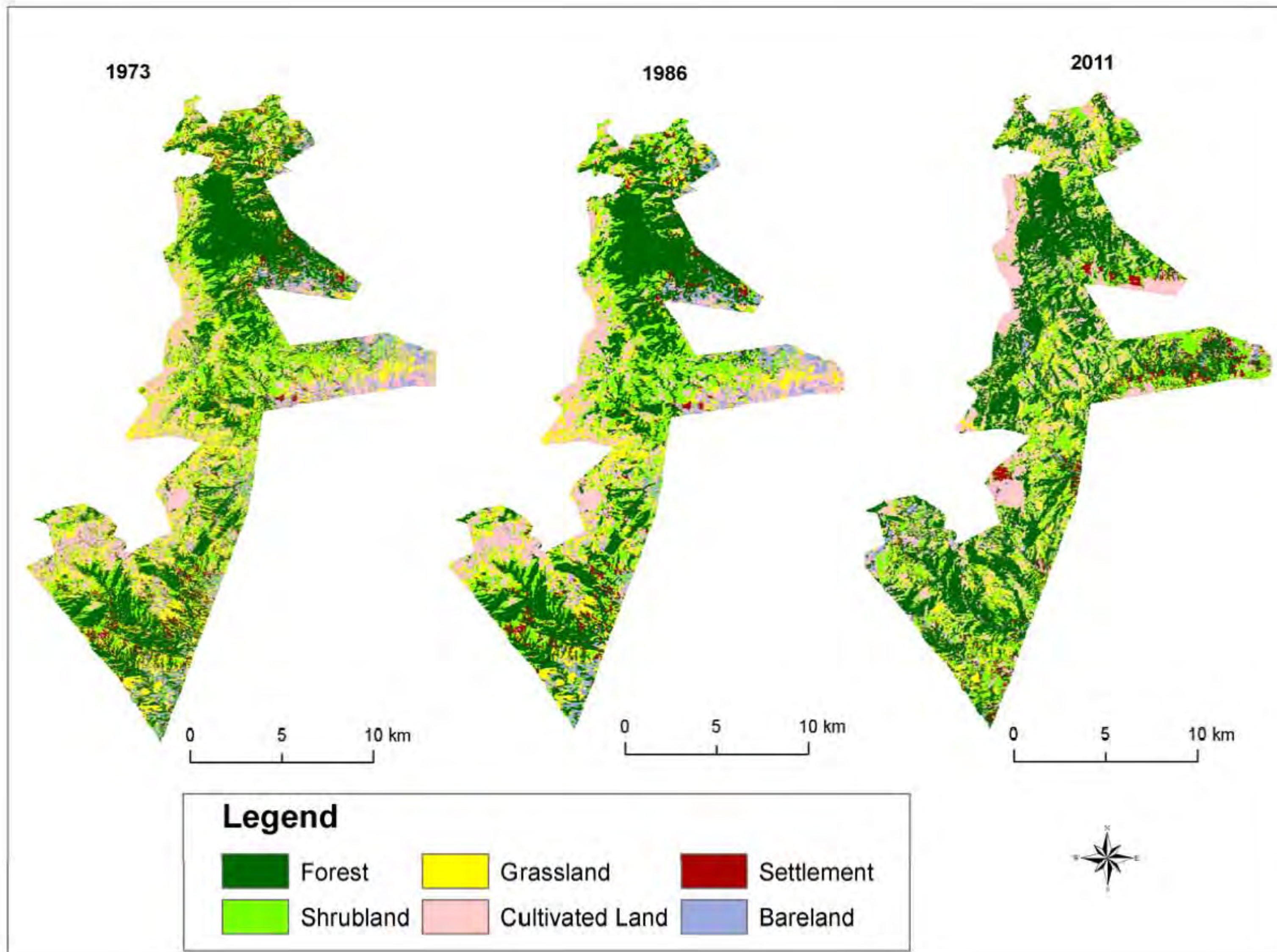

Fig. 28 Land use\land cover classification map of 1973, 1986 and 2011

4.3.3. Rate of land use\land cover change

The rates of change in land use\land cover types are indicated in Table 20. In the first reference period (1973 - 1986), positive rates of change were observed for forest, cultivated land and shrub land with a rate of 53, 13 and 8 ha/year respectively. Whereas, negative rates of change were observed for settlement and grassland, which decelerated with a rate of 41 and 23 ha/year respectively. In the second reference period (1986 - 2011) maximum positive rate of change was observed for forest with 38 ha/year followed by cultivated land 31 ha/year whereas maximum deceleration rate was observed for bare land 50 ha/year and grassland 29 ha/year.

Table 20 Rate of change in land use\land cover types

				Rate of change in cover (ha/y)	
Class name	**1973 area/ha**	**1986 area/ha**	**2011 area/ha**	**From 1973 - 1986**	**From 1986 - 2011**
Forest	7352.8	8042.9	8992.6	53.08	37.99
Shrub land	5518.3	5618.6	6040.4	7.72	16.87
Grassland	3192.6	2893.7	2179.6	-22.99	-28.56
Cultivated land	4719.7	4889.0	5676.0	13.02	31.48
Settlement	1533.9	1003.8	820.7	-40.78	-7.32
Bare land	1858.4	1727.8	466.5	-10.05	-50.45
Total	24,175.8	24,175.8	24,175.8		

4.3.4. Land use\land cover change detection

Change detection is used to find out which land use\land cover is changed to the other type of land use\land cover class. This process involves a pixel comparison of the study year image through overlay analysis. The land use\land cover change matrix represents the direction of change and the landuse type that remains as it is at the end of the period.

Change detection was made based on the classified maps of 1973 and 1986, and 1986 and 2011 (Table 21 and 22) respectively. The values on the diagonals (bold and underlined numbers) are the landcover types that had not changed during the study period. As well, image differences were observed for all land use\land cover types with different extent throughout the study period. Comparison between 1973 and 1986 land use\land cover types showed that higher class changes were observed for shrub land, grassland, cultivated land, settlement and forest among which positive changes were observed for forest, shrub land and farm land whereas negative changes were observed for the remaining land use\land cover types (Table 21).

In the first reference period (1973 - 1986) 672.8 ha of shrub land, 116.1 ha of grassland, 225.7 ha of farm land and 323.7 ha of settlement were converted to forest contributing to its development. Substantial area of grassland (592 ha), cultivated land (410.2 ha) and forest (378.8 ha) were converted to shrub land. Moreover, other land use\land cover types also showed considerable change from one to another during the study period which means that no land use\land cover type was static (Table 21).

Table 21 Landcover/landuse change detection matrix showing class to class changes, between 1973 and 1986

		Initial year (1973)						
Final year (1986)		**Forest**	**Shrub land**	**Grassland**	**Cultivated land**	**Settlement**	**Bare land**	**Class total**
	Forest	**6594.1**	672.8	116.1	225.7	323.7	110.5	8042.9
	Shrub land	378.8	**3732.9**	592.0	410.2	358.3	146.4	5618.6
	Grassland	59.0	390.9	**1914.1**	315.0	93.4	121.4	2893.7
	Cultivated land	150.1	461.3	418.7	**3587.5**	43.1	228.3	4889.0
	Settlement	101.1	140.6	56.4	14.2	**617.4**	74.1	1003.8
	Bare land	69.7	120.0	95.4	167.2	98.0	**1177.7**	1727.8
	Class total	7352.8	5518.3	3192.6	4719.7	1533.9	1858.4	0.0
	Class Change	758.7	1785.4	1278.5	1132.2	916.5	680.7	0.0
	Image difference	690.1	100.3	-298.9	169.3	-530.1	-130.6	0.0

Assessment of 1986 and 2011 land use\land cover classification revealed that forest, shrub land and cultivated land showed higher positive changes while grassland and bare land showed higher negative change. Therefore, large amount of shrub land (838 ha), grassland (552 ha) and cultivated land (390 ha) were converted to forest. As well, extensive amount of bare land (657 ha), grassland (435 ha) and shrub land (287 ha) were changed to farm land contributing for its increment in the period (Table 22).

Table 22 Landcover\landuse change detection matrix showing class to class changes, between 1986 and 2011

Final year (2011)		Initial year (1986)						
		Forest	Shrub land	Grassland	Cultivated land	Settlement	Bare land	Class total
	Forest	**6871.1**	837.5	551.9	390.2	166.2	175.7	8992.6
	Shrub land	663.9	**4086.8**	584.2	292.6	116.1	296.8	6040.4
	Grassland	139.7	245.8	**1234.9**	158.3	59.8	341.2	2179.6
	Cultivated land	236.0	286.9	435.4	**3895.5**	165.7	656.5	5676.0
	Settlement	48.4	79.6	68.8	58.7	**460.4**	104.8	820.7
	Bare land	83.7	82.1	18.4	93.7	35.7	**152.8**	466.5
	Class total	8042.8	5618.5	2893.7	4889.0	1003.8	1727.9	0.0
	Class change	1171.7	1531.8	1658.8	993.5	543.4	1575.1	0.0
	Image difference	949.8	421.8	-714.2	787.0	-183.0	-1261.4	0.0

4.4. Effect of habitat fragmentation on plant species richness

4.4.1. Fragmentation measures

4.4.1.1. Spatial landscape metrics at class and landscape level

From LULC maps of 1973, 1986 and 2011, the following spatial landscape metrics (Tables 23 and 24) were computed at class and landscape levels. Number of patches was calculated both at the class level (Table 23) and the landscape level (Table 24). On average, at the class level, the shrub land, the grassland, settlement and cultivated land were the LULC classes with large number of patches in the study area, while the forest and bare land had the lowest number of patches.

At the landscape level the number of patches was highest in 1973 and lowest in 1986 (Fig. 29), and showed a negative relationship with mean patch area (Table 23). At the

class level, such negative relationship of number of patches with patch area was also observed for all LULC classes (Table 23).

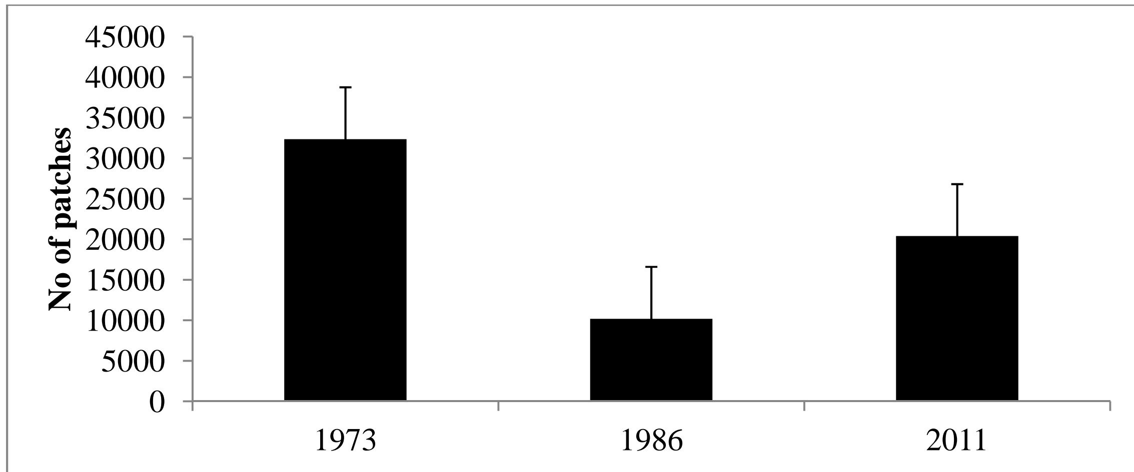

Fig. 29 Number of patches (NP #) at the landscape level, for 1973, 1986 and 2011

Class area measures how much of the landscape was comprised of a particular LULC type (Table 23). The percentage of landscape (PLAND) was an alternative to express the total class area (CA), which facilitates the comparison of proportions of each LULC class among study years. In general, the most abundant LULC classes in the study area were forest, shrub land, cultivated land and grassland; the less abundant were settlement and the bare land.

AREA_MN is a function of the area and number of patches. AREA_MN was evaluated at the class level (Table 23) and at the landscape level (Table 24). The LULC classes that exhibited the largest mean patch sizes were forest, cultivated land and shrub land classes. Whereas, the LULC classes with the smaller mean patch sizes were settlement and bare land (Table 23).

Forest land presented as the dominant class of the study area because its percentage of total area was largest (Table 23). The statistic of forest class showed that the percentage

of landscape (PLAND) index increased from 9% to 12.1%, and the number of patches (NP) decreased from 3,020 to 2,953 during the whole study period from 1973 to 2011. This combination result suggested dominance of the forest class in the study area. This was supported by the mean patch area (AREA_MN), which increased from 2.4 ha (in 1973) to 3.4 ha (in 2011). The values of mean shape index (SHAPE_MN) of the forest class for the 1973 and the 2011 were more than 1, exposing that the average forest patch shape was non-square and there was no significant difference in shape index values.

The increasing of mean proximity index (PROX_MN, defined by 500 m) from 635.8 m (1973) to 1,040.2 m (2011) indicated that forested patches in 2011 became more contiguous in spatial distribution than in 1973. This was asserted once again by the decreasing of mean nearest neighbour distance (ENN_MN) from 76.1 m to 73.8 m and the interspersion and juxtaposition index (IJI) from 81.1 to 78.4 indicating decrease in number of patches. In regards to cultivated land during the whole study period (1973-2011), percentage of landscape (PLAND) index increased from 5.8 to 7.3, the mean patch area (AREA_MN) increased from 0.9 to 1.1 and the largest patch index (LPI) increased from 0.4 to 0.9. These indexes evidenced the expansion of cultivated area in the study landscape (Table 23).

The decreasing of mean proximity index (PROX_MN) from 60.5 m to 41.4 m and decreased number of patches (NP) from 5,085 to 4,712 (Fig. 29) indicated that cultivated patches in 2011 became more contiguous in spatial distribution than in 1973. This was supported again by the decreasing of mean nearest neighbor distance (ENN_MN) from 76.3 to 73.8 and the interspersion and juxtaposition index (IJI) from 82.3 to 78.4 (Table 23).

The spatial analysis of shrub land showed significant increase of the percentage of landscape index (PLAND) from 5,518.7 to 6,040.4 and the mean patch area (AREA_MN), increased from 0.7 ha to 0.9 ha. These indexes showed the expansion of shrub land vegetation throughout the study period. Finally, the decreasing of mean proximity (PROX_MN) from 40.8 m to 18.6 m and of the interspersion and juxtaposition index (IJI) from 87.2 to 71.6 indicated that shrub land class distributed a more landscape configuration in 2011 than in 1973.

The Mean Euclidean nearest neighbour distance (ENN_MN) metric is a measure of the patch dispersion in the landscape and it measures the distance to the nearest neighbouring patch of the same type. At the class level (Table 23) on average, the class with the larger values for this index was the bare land class with 150.2 m of distance between patches, and the class with the smaller value for ENN_MN was the shrub land class with an average of 66.8 m between patches of the same LULC class.

The large distances among patches of bare land indicated the low abundance (low number of patches - 796) and small mean patch size (0.6 ha) of this LULC class. The few patches of this class were not close to each other. On the other hand, the low ENN_MN between patches of shrub land class (Table 23) was most likely due to this LULC class being very abundant in the study area and with the higher number of patches (7,700) and were, in general, closer to each other.

Table 23 Metrics of landscape structure for selected indices at the class level, 1973, 1986 and 2011

Class /year	CA (ha)	PLAND (%)	NP (#)	LPI (%)	AREA_ MN (ha)	SHAPE_ MN (ha)	PROX_ MN (m)	ENN_ MN (m)	IJI (%)
1973									
Forest	7352.6	9.0	3020	4.0	2.4	1.3	638.5	76.1	81.1
Shrub land	5518.7	6.7	7700	0.5	0.7	1.3	40.8	66.8	87.2
Grassland	3192.6	3.9	6511	0.1	0.5	1.3	6.4	74.3	80.0
Cultivated Land	4719.7	5.8	5085	0.4	0.9	1.3	60.5	76.3	82.3
Settlement	1533.4	1.9	6141	0.0	0.3	1.2	1.8	82.5	81.5
Bare land	1858.7	2.3	3883	0.1	0.5	1.2	7.5	87.0	90.9
1986									
Forest	8042.8	9.8	1034	4.5	7.8	1.3	613.9	101.8	80.0
Shrub land	5618.7	6.8	2497	1.2	2.3	1.3	103.6	87.8	85.7
Grassland	2893.6	3.5	2215	0.2	1.3	1.2	8.9	101.3	80.0
Cultivated Land	4889.4	5.9	1698	0.7	2.9	1.3	86.4	102.8	82.6
Settlement	1003.8	1.2	1561	0.0	0.6	1.2	1.8	130.4	81.2
Bare land	1727.4	2.1	1184	0.1	1.5	1.2	9.3	119.6	91.5
2011									
Forest	8992.6	12.1	2953	4.0	3.4	1.3	1040.2	73.8	78.4
Shrub land	6040.4	7.5	5395	0.2	0.9	1.3	18.6	72.8	71.6

Grassland	2179.6	2.0	2731	0.0	0.6	1.2	3.3	96.8	64.1
Cultivated Land	5676.0	7.3	5689	0.9	1.1	1.3	41.4	73.9	75.0
Settlement	820.7	1.2	2834	0.1	0.3	1.1	1.9	99.5	70.2
Bare land	466.5	0.6	796	0.1	0.6	1.2	1.6	150.2	59.6

At the landscape level (Table 24) the largest value for ENN_MN was 104.88 m for 1986 and the smallest was 76.08 m for 1973. The high mean Euclidean nearest neighbour distance for 1986 was perhaps due to the result of similar isolated patches joined to one another. Such aggregation of similar patches takes place when corridors are eliminated between similar patches, their connectivity increased and their edges joined together. On the other hand, in 1973 low values for ENN_MN at the landscape level, indicated the LULC composition were not so dominated by just one class and there were more number of patches of all classes, then the LULC classes were more interspersed and the average for ENN_MN was low.

Table 24 Metrics of landscape structure for selected indices at the landscape level, 1973, 1986 and 2011

Year	**NP (#)**	**LPI (%)**	**AREA_ MN (ha)**	**SHAPE_ MN (ha)**	**PROX_ MN (ha)**	**ENN_ MN (m)**	**IJI (%)**	**SHDI**	**SHEI**
1973	32341	70.48	2.54	1.26	81.37	76.08	84.93	1.10	0.56
1986	10190	70.64	8.08	1.27	105.36	104.88	84.85	1.08	0.55
2011	20399	70.68	4.06	1.26	167.82	83.21	73.80	1.02	0.53

The Interspersion and juxtaposition index (IJI) was used to assess the neighbourhood relations between patches. IJI expresses observed interspersion over the maximum possible interspersion for the given number of LULC classes. At the class level (Table 23), on average the values for IJI were very similar among classes (small range of variation among the mean values of classes). The maximum mean value of IJI was for bare land class (91.5) in 1986. This was anticipated as this class had a tendency to have small mean patch size and a low number of patches (1,184), and additionally it occupied low proportions of the landscape (7%). In all the study years at the landscape level there was a lower mean patch size and a higher number of patches; thus these high values at the landscape level were good expression of the interspersion and juxtaposition of the LULC classes in the study area.

The comprehensive analysis in landscape level confirmed the evidence of fragmentation of the landscape with the increasing of total number of patches from 10,190 (in 1986) to 20,399 (in 2011) while the mean patch area decreased from 8.08 to 4.06 in 2011 (Table 24). In general, it was observed that the LULC class that exhibited the largest mean patch size (forest land) had low number of patches next to bare land. On the other hand, the shrub land class had low AREA_MN and high number of patches. The grassland class also exhibited this pattern. In addition, both Shannon's diversity and evenness indices in 2011 were lower than in 1973, showing the slight decrease of the landscape heterogeneity and evenness.

Decrease of mean nearest-neighbour distance (ENN_MN) from 104.88 to 83.21 and increase of mean proximity index (PROX_MN) (Table 24 and Fig. 30) revealed that patches tended less isolated during the whole study period (1973 - 2011). Besides, interspersion and juxtaposition index (IJI) decreased from 84.93 to 73.80, indicating a trend towards the uniform configuration of the landscape (Table 24).

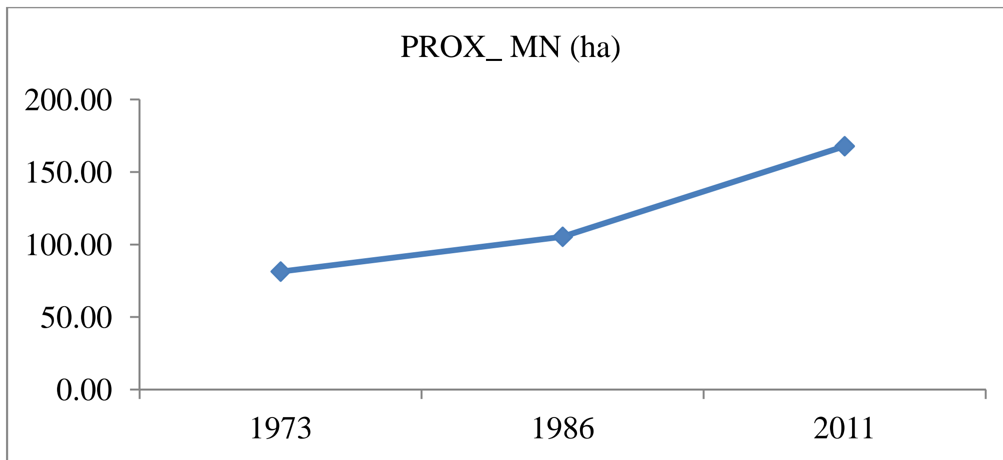

Fig. 30 Mean proximity index (PROX_MN) at the landscape level, for 1973, 1986 and 2011

The computed landscape level diversity metrics indicated that the study area had become less diverse. This was reflected in decreased values for the Shannon diversity (SHDI) indices of the whole study period (Fig. 31 and Table 24). This metrics was a function of the number of patch classes (PR) and the total number of patches (NP).

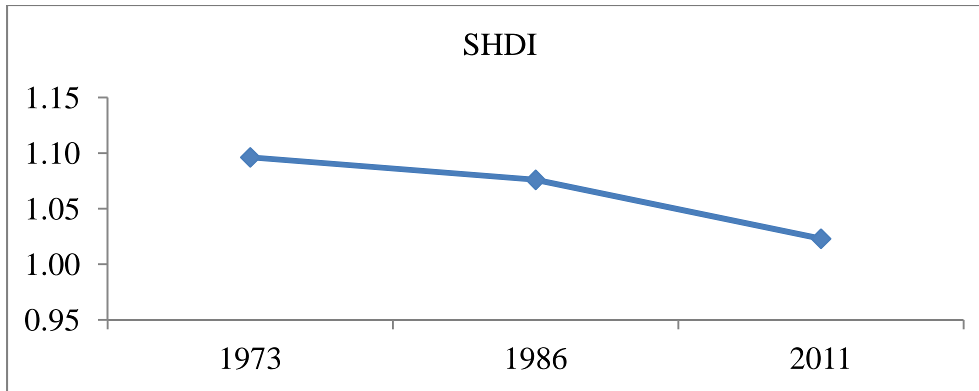

Fig. 31 Shannon diversity (SHDI) index at the landscape level, for 1973, 1986 and 2011

Given that patch richness remained constant between study years, the decrease observed in SHDI values (Fig. 31) can be attributed directly to the decrease in NP. Decreases in this landscape diversity metrics indicated that the study area became less heterogeneous in 2011 compared to 1973. The Shannon evenness index also decreased during the whole study period (Fig. 32). The maximum value of SHEI is 1, indicating a perfectly even distribution of area among the different classes of patches (Table 24).

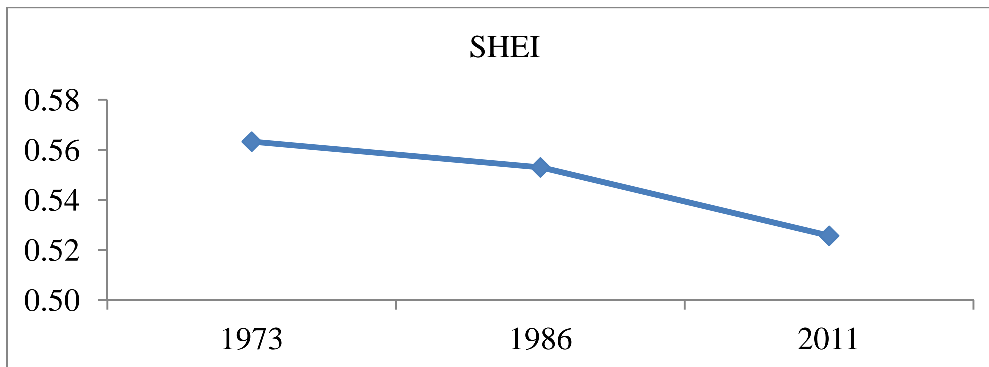

Fig. 32 Shannon evenness index (SHEI) at the landscape level, for 1973, 1986 and 2011

Table 25 Main features of the 18 forest patches (Fig. 33) selected for evaluation of patch area and species richness. Core area was measured at a distance of 100 m to the patch edge

PID	AREA	SHAPE	FRAC	CORE	CAI
P1	154.17	4.12	1.19	110.52	71.68
P2	47.88	1.85	1.09	37.35	78.0
P3	214.29	6.64	1.26	129.96	60.64
P4	451.35	9.51	1.29	276.57	61.27
P5	46.8	4.36	1.22	21.15	45.19
P6	140.22	6.93	1.27	70.56	50.32
P7	61.56	5.30	1.25	26.46	42.98
P8	51.93	4.42	1.22	23.94	46.10
P9	5.04	1.93	1.12	1.62	32.14
P10	7.02	2.05	1.13	2.61	37.17
P11	1.08	1.28	1.05	0.18	16.66
P12	1.98	1	1.01	0.72	36.36
P13	24.48	3.78	1.21	9.81	40.07
P14	50.94	4.87	1.24	22.77	44.69
P15	0.99	1.28	1.06	0.10	9.09
P16	0.54	1.4	1.08	0.04	0.08
P17	0.72	1.5	1.10	0.06	0.09
P18	0.63	1.33	1.09	0.08	0.06

4.4.2. The relation of patch area and fractal dimension with plant species richness

The study used LULC map of 2011, to compute three patch metrics, namely, patch size, patch fractal dimension and core area to evaluate the relationship between patch area and species richness of the studied patches. The analysis was facilitated through the use of patch analysis extension of ArcGIS and FRAGSTATS software 4.1. Eighteen patches were identified and Fig. 33 showed the spatial distribution of these patches within the study area. Table 25 summarizes the computed total patch area, fractal dimension and core area for each patch.

The average extent and the complexity of patches were evaluated by computing patch size and shape complexity:

4.4.2.1. Patch size (area) and plant species richness

As indicated in Table 25, the patches that exhibited the largest patch sizes were P4, P6, P3 and P7. While the patches with the smallest patch sizes were P16, P18, P17 and P15. These reduced area results were probably the patches were devoted to more intervention and disturbance, which can be attributed to deforestation in the past and the current illegal logging activities.

As it is indicated in Table 26, the patches that showed the largest sizes were also having high species richness. Species numbers increased significantly ($P < 0.05$) with increasing habitat area. This confirmed a general positive relationship between species richness and patch area. The only exception was P13 where slightly more species were recorded in the smallest, than in the medium-sized patch (Table 26).

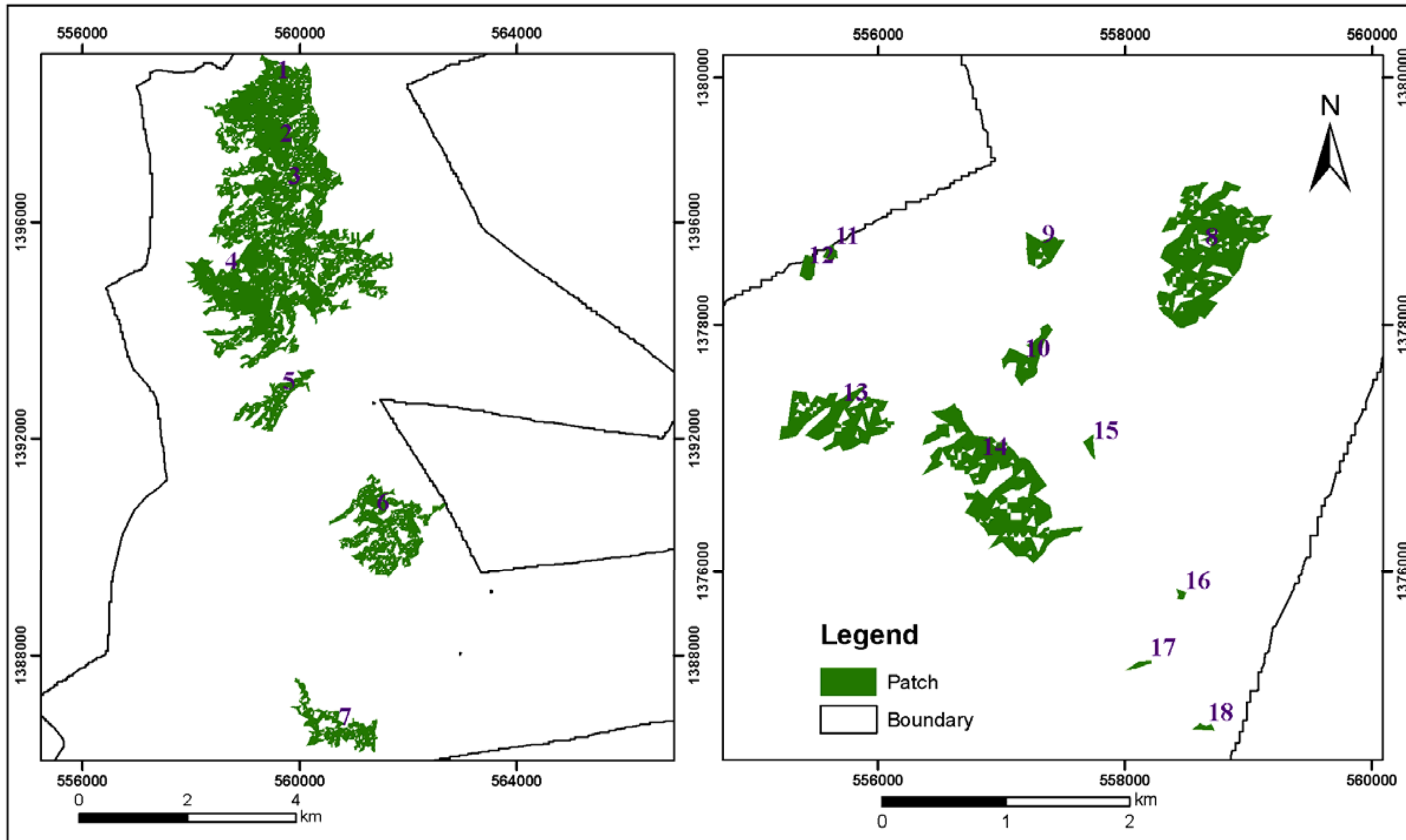

Fig. 33 Location map of the patches selected for evaluation of patch area and species richness. Patch extent was based on land use\land cover classification image of 2011. Labels referred to patch Id. in Table 25

As shown in Fig. 33, forest fragmentation was mostly concentrated along the patches having the smallest patch sizes, and which were nearest to towns and farming communities, where access to the forest was facilitated. In addition, total core area (Table 25) was related to both patch area and patch shape, and fragmentation of core forests directly affects habitat suitability for particular species.

Table 26 Number of species recorded from the selected patches

Patch ID	No of plots	Observed No of species
P1	6	45
P2	4	40
P3	6	53
P4	6	62
P5	4	37
P6	6	42
P7	4	35
P8	5	24
P9	4	23
P10	3	21
P11	6	17
P12	5	33
P13	6	44
P14	5	41
P15	3	17
P16	4	19
P17	3	29
P18	3	31

4.4.2.2. Fractal dimension (FRAC) and species richness

The computed patch fractal dimension index (FRAC) for all patches showed values that were greater than 1.0 (Table 25), indicating complex patch shape. Patch number P4, P6, P3 and P7 showed highest computed values of FRAC, which was supported by the values of shape index (SHAPE) and this can be attributed to their irregularity in shape. On the other hand, low values of FRAC and SHAPE index were observed for P12, P11, P2 and P15 (Table 25).

CHAPTER FIVE

5. DISCUSSION, CONCLUSION AND RECOMMENDATIONS

5.1. Discussion

5.1.1. Plant diversity studies in Hugumburda-Gratkhassu NFPA

5.1.1.1. Floristic composition

Results of floristic analysis showed that Hugumburda-Gratkhassu Forest was rich in species composition as shown by the presence of 326 species distributed in 238 genera and 88 families. The observed diverse flora of Hugumburda-Gratkhassu Forest was in line with the general pattern of rich species presence in East African montane forests. Generally, tropical forests are among ecosystems that have high species diversity of the world (Gentry, 1995). East African forests are also considered center of botanical endemism (Lovett, 1998). According to Coetzee (1978) and Tamrat Bekele (1993), among the Afromontane forest regions, East African Mountains (found in Ethiopia, Kenya, Tanzanya and Uganda) have the richest and most diverse flora. Comparison of species richness in tropical and subtropical dry forests is difficult due to the inherent heterogeneity of the forests (Murphy and Lugo, 1986) and difference in forest size, data collection methodology and objectives of the studies. However, to give a general picture of the species richness of Hugumburda-Gratkhassu Forest, the results of the present study were compared with results from other forests in Ethiopia.

Accordingly, Hugumburda-Gratkhassu Forest had higher species richness than some other forests in Ethiopia such as Chilimo Forest (213 species) (Teshome Soromessa and Ensermu Kelbessa, 2013), Sire Beggo (185 species) (Abyot Dibaba *et al*., 2014), Masha

(130 species) (Abreham Assefa *et al.,* 2013), Komto forest (103 specie) (Fekadu Gurmessa *et al.,* 2012), Dense Forest (158 species) (Ermias Lulekal, 2014), Jibat Forest (131 species) (Tamrat Bekele, 1993), Yayu Forest (220 species) (Tadesse Woldemariam *et al*., 2008) or Dello Menna (171 species) (Motuma Didita *et al*., 2010). However, its species richness was found to be less when compared to other forests, e.g., Borena Sayint National Park (397 species), (Husen Adal, 2014), Simien Mountains National Park (532 species) (Getinet Masresha, 2014). All these differences in species richness among these sites could mainly be attributed to the dissimilarities of the sites in terms of location, altitude, human impact, rainfall, and other biotic and abiotic factors. Variation in species composition over different forests could be attributed to topographic difference among the forests compared, as well as the degree of available gradients in the respective forests (Chen *et al.,* 2005).

The highest representation of species from the family Asteraceae (38 species, 11%), Poaceae (31 species, 10%) and Fabaceae (31 species, 10%) in Hugumburda-Gratkhassu Forest in general (Table 4) could be related to the fact that they are species rich families in the flora area, Ethiopia and Eritrea (Mesfin Tadesse, 2004). Asteraceae was also shown to be well-represented in other montane forests in Ethiopia including Dense Forest (20 species, 27%) (Ermias Lulekal, 2014), Komto forest (17 species, 9.44%) (Fekadu Gurmessa *et al.,* 2012) and Adelle and Boditi forest (39 species, 17%) (Haile Yineger *et al*., 2008). Similarly, the top species rich families of Hugumburda-Gratkhassu Forest were more or less the same with other forests even if there was difference in their order. Some of these families, for instance Asteraceae, Euphorbiaceae and Fabaceae were said to be always among the top 10 species-rich families in many Neotropical forests (Gentry,

1995). Hence, the dominance of these families in this study was in agreement to their general dominance in tropical forests.

Successful colonization of these families can be attributed to their pollination, morphological and dispersal mechanism that facilitate to move for long distance and set down in the non occupied parts of the Forest. According to Hedberg (1964), wind dispersal is one of the main mechanisms affecting long distance dispersal for Afromontane landscapes. Moreover, many species of Asteraceae are having umbrella shape structures adapted for air dispersal (Hedberg, 1964), and increase their opportunity to successful establishment of their populations. Physiological and genetic adaptation such as augmented power of germination and ability to endure severe climatic circumstances and biotic competition for limited resources might facilitate for their dominance (Ahmed Abdikadir, 2013). There were also very poorly represented families, with only one or two genera and only a single or few species each (Appendix 2). Species of these families were either rare or restricted in their distribution.

Results for growth form analysis also showed that herbaceous species outnumber (57%) in Hugumburda-Gratkhassu NFPA (Fig. 12) when compared to other growth forms. This could be related partly to disturbance and presence of canopy gap in the Forest. These observations are in agreement with similar patterns of dominance of herbaceous species in other Ethiopian montane forests (Haile Yineger *et al.,* 2008; Fekadu Gurmessa *et al.,* 2012; Ermias Lulekal, 2014) and elsewhere in other tropical forests (Dhillion *et al*., 2003).

Out of the total plant species (326 species) collected from the study area, 15 species were endemic to Ethiopia and Eritrea (Table 6). Out of this nine species were included in the IUCN Red List Categories of Ethiopia and Eritrea (Vivero *et al*., 2005). Based on the IUCN criteria of level of threat, one species as vulnerable (VU) and two species had been evaluated as near threatened (NT). Six species had been categorized as species of least concern (LC) and the remaining seven species not evaluated. This was in line with the typical feature of Afromontane forests that house numerous endemic species (White, 1983). According to this author, the Afromontne region is an important center of endemism housing over 3,000 endemic plant species. The highest presence of endemic species in Eastern Africa montane forests was also mentioned as one of the major reasons to recognize the whole region as a biodiversity hotspot. It is an area that will fast loose the endemic and other taxa along with the natural habitat beyond the 70% it has already scarified (Mittermeier *et al.,* 2005). Thus, conservation measures in Hugumburda-Gratkhassu Forest by policy makers, natural resources managers or stakeholders need to take into account special protection and monitoring of the endemic and threatened species of the Forest in order to avoid further extinction and to conserve the natural habitat status.

Results of plant species richness and diversity of the two Forest remnants showed, higher species richness and diversity for Gratkhassu Forest fragments. This difference was also confirmed using rarefaction curve and found to be statistically significant (Fig. 8). Hugumburda Forest fragments are located on plain topography with limited altitudinal range (2100 - 2505 m) than that of Gratkhassu Forest fragments (1814 -2805 m) and this difference in altitude coupled with slope and disturbance could influence its abundance of species. Different types of environmental factors affect the processes that can both

enhance or extinct diversity (Sagar *et al*., 2003). Altitude and slope are important environmental variables to influence species richness and dispersal activities of plants (Tamrat Bekele, 1994; Ermias Aynekulu, 2011). Both of the Forest remnants were not intact but composed of fragmented patches and the intensity and main causes for disturbance were not the same. In Hugumburda, there was high pressure of disturbance on edge habitats from cutting wood for fuel, construction material and uncontrolled grazing inside the Forest than that of Gratkhassu. Therefore, these activities can affect species diversity when they are undertaken unsustainably.

Higher number of species resulted in higher Shannon diversity. Accordingly, the higher species richness at Gratkhassu gave higher Shannon diversity index when it is compared with that of Hugumburda (Table 7). According to Magurran (2004), presence of greater evenness or greater richness or both in an area can result in increased Shannon diversity. In addition, the rarefaction curve of Gratkhassu was outside of the 95% confidence interval of the rarefaction curve of Hugumburda showing that Gratkhassu had significantly higher species richness (Figs. 7 & 8).

Species richness as a measure of diversity is fundamental to assess community situation and quantitative evaluation of biodiversity. But, as it is stated by Chiarucci *et al*. (2003), entire inventory of total number of species for extensive study of plant population is not easy. Therefore, estimating species richness using various numerical methods is mandatory in studies of biodiversity (Magurran, 2004). This can help to examine changes in ecological consistency and as necessary tool to make conversant decisions in conservation. Based on different non-parametric species estimators (Table 8), maximum number of species estimated was (262 species) in Hugumburda Forest fragments and

(293 species) that of Gratkhassu Forest remnants. In addition the overall species richness of Hugumburda-Gratkhassu NFPA was estimated to be in the range of 347 to 383 species and showed 86 - 95% of the species found in the area were collected. According to Heck *et al*. (1975), collecting 50-75% of the total number of species of a given area is satisfactory; so, in this study sufficient species of the NFPA was recorded.

Analysis of incidence and abundance based similarity indices (Table 9) indicated high similarity between the two forest remnants of the study area. The similarity of the species composition of the two Forest remnants ranged between 55 to 82% (Table 9). This suggests presence of high similarity in geographic location and/or altitudinal range, intensity and type of disturbance experienced.

5.1.1.2. Plant community types

From the result of cluster analysis and ordination (multivariate techniques), five different plant communities (Fig. 9) were identified in Hugumburda-Gratkhassu NFPA. Dissimilarity in environmental factors such as, altitudinal variation, difference in soil texture, pH, EC, slope, aspect, etc., together with disturbance and species environmental necessities might have involved in their formation. These plant communities were found to be composed of their own indicator species. For example, many species were restricted only to Gratkhassu forest fragments, and these comprise *Myrica salicifolia, Acacia etbaica, Orobanche ramosa, Dombeya torrida, Acacia tortilis, Balanites aegyptiaca, Grewia mollis, Acacia seyal, Themeda triandra, Helinus mystacinus, Acacia mearnsii, Monechma debile, Kalanchoe lanceolata, Acacia asak, Verbascum sinaiticum, Cupressus lusitanica, Crepis rueppellii, Caylusea abyssinica,* etc. (Table 10). Similarly, there were characteristic species of Hugumburda Forest fragments which include *Erica arborea,*

Cynanchum abyssinicum, Ficus sur, Podocarpus falcatus, Allophylus abyssinicus, Pinus patula, Ekebergia capensis, Pavetta oliveriana, Cordia africana, Pterolobium stellatum, Laggera crispata, Pelargonium alchemilloides, Achyranthes aspera, etc. that were recorded only in Hugumburda Forest fragments (Table 10).

Gratkhassu forest remenants had three plant community types (i.e. community types 1, 2 and 4). Community Type one (*Juniperus procera - Nuxia congesta* Community Type), was composed of 13 plots. In this community, some of the common species, besides the dominant species, were *Cassipourea malosana, Cupressus lusitanica, Calpurnia aurea, Carissa spinarum, Clutia abyssinica, Dovyalis verrucosa, Asparagus racemosus*, *Solanum benderianum, Vigna membranacea,* etc. In most of its stands, introduced exotic species of *Cupresus lusitanica, Eucalyptus camaldulensis* and *Eucalyptus globulus* were observed. The stands sampled in this community were located in an area having shallow soils with medium human interference in the form of firewood collection and selective cutting. This might be due to its being near to human settlement and agricultural encroachment. Although most areas of this stands were highly affected before, about 35 years being used as farming land, by now it is in good regeneration status.

Community Type 2 (*Cassipourea malosana* - *Myrica salicifolia*) was from the disturbed part of Gratkhassu Forest fragments and it included 15 plots and 107 species. Topographically this community had steeper slope and higher mean altitude (2562 m) than the other communities. The characteristic species of this community were *Dovyalis abyssinica, Bersama abyssinica, Acacia abyssinica, Osyris quadripartita, Clutia abyssinica*, *Aloe debrana, Clerodendrum myricoides, Crotalaria lachnophora, Hypericum quartinianum, Otostegia fruticosa, Rosa abyssinica* and *Rumex nervosus.* In

addition to these species, there were also abundant herbaceous species in the field layer including *Verbascum sinaiticum, Andropogon abyssinicus, Crepis rueppellii, Caylusea abyssinica, Euphorbia schimperiana, Silene macrosolen, Cymbopogon pospischilii, Themeda triandra,* etc. in this community.

Hugumburda Forest remenants comprised two plant community types (i.e. community types 3 and 5). These communities might be the result of difference in disturbance rate and other biotic and abiotic factors or combination of these factors. Community Type 3 (*Psydrax schimperiana* - *Podocarpus falcatus* community type) comprised of 15 plots which were located in specialized habitats such as along river courses at the middle of Hugumburda Forest fragments and it was characterized by a gentle slope, deep soil and higher soil moisture. Thus, it's being grazed by cattle and also its human impact was found to be low. Regenerating species of *Podocarpus falcatus* were common here. The characteristics species of this Community was dominated by Afromontane species and these include *Ficus sur, Podocarpus falcatus, Allophylus abyssinicus, Ekebergia capensis, Phytolacca dodecandra*, *Celtis africana, Cordia africana, Maytenus arbutifolia, Teclea simplicifolia, Pterolobium stellatum, Berberis holstii, Abutilon longicuspe, Debregeasia bicolar, Gomphocarpus fruticosus, Meriandra bengalensis,* etc. (Table 10).

Community Type 4 (*Acacia etbaica* - *Balanites aegyptiaca* Community Type) comprised of nine plots which were all from Gratkhassu Forest fragments. The stands of this community were from an area, which was at lower altitudes, having shallow soil and that received lower amount of annual rainfall with higher temperature. The common species

of this community include *Grewia mollis, Acacia seyal, Cymbopogon pospischilii, Acacia abyssinica, Acokanthera schimperi, Acacia asak, Helinus mystacinus, Clematis hirsuta, Rhus retinorrhoea, Rosa abyssinica, Ziziphus spina-christi, Calotropis procera, Euclea schimperi, Grewia tembensis, Themeda triandra, Kalanchoe lanceolata, Arthraxon prionodes, Hyparrhenia hirta, Pennisetum thunbergii,* etc. (Table 10). This community was highly influenced by people collecting firewood, charcoal making and grazing animals. This was due to its being nearby to Alamata Town and having species of plants suitable for charcoal making and firewood.

In Community Type 5 (*Olea europaea - Cadia purpurea* Community Type) 25 plots and 148 species were associated with this community. All the plots belonging to this community were drawn from the moderately disturbed part of Hugumburda Forest fragments. The characteristic species of this Community were *Cadia purpurea, Olea europaea* ssp. *caspidata, Rhus glutinosa, Erica arborea, Discopodium penninervium, Pittosporum viridiflorum, Teclea simplicifolia, Juniperus procera, Carissa spinarum, Dodonaea angustifolia, Pterolobium stellatum, Opuntia ficus-indica, Psydrax schimperiana, Ficus palmate,* etc. (Table 10). This Community had very dense shrub layer and abundant herbs when compared to the other communities. Ground species of the community include *Cynodon dactylon, Heracleum abyssinicum, Hypoestes forskaolii, Bidens biternata, Bidens pilosa, Argemone mexicana, Carduus nyassanus, Cynoglossum lanceolatum,* etc. This Community was surrounded by upper slope and bare rock area, thus little disturbance by human and cattle was encountered.

The five plant communities differed in their species richness, diversity and evenness. Community types 2, 4 and 5 were the richest in respect to species richness and diversity and Community Types 1 and 3 the lowest (Table 11). The greater diversity in any community can be attributed to the amount of disturbance in the quadrats. Thus, higher plant diversity in the above three community types may be due to higher site productivity created by such intermediate disturbance and mild climate and deep soils (Flombaum and Sala, 2008) compared to the higher disturbance, shallow soil and steeper slope where Community Type one occurs. Disturbance affects species composition and diversity by affecting certain species and size classes and by changing the light environment of the understory (Slik, 2004; Engelbrecht *et al.,* 2007). Species evenness shows the relative proportional abundance of a species in quadrats. Low evenness value implies the dominance of the environment by few species. Based on this, communities 2 and 4 had the highest evenness value. Community 5 had the least evenness value, which indicates dominance by few species.

Generally, it is evidenced from the result that the five communities are diverse, rich and with even distribution of species. This is most likely because at present improved Forest Administration is practiced in the study area. At this time disturbance in large parts of the forest is intermediate or reduced. This reduced or intermediate disturbance may give possibility for less competent species to coexist together. This in turn may add to the high evenness and diversity of the community types.

Identified species composition, plant communities, altitudinal ranges, type of dominant species in the upper canopies of the Forest and climate reading of the area indicated that Hugumburda-Gratkhassu Forest belongs to the category of dry evergreen Afromontane

forest and grassland complex. According to Friis *et al*. (2011) and Zerihun Woldu (1999), dry evergreen forests lie between altitudinal ranges of 1800-3000 m a.s.l. and consist of canopies dominated by *Podocarpus falcatus, Juniperus procera* and *Olea europea* ssp. *cuspidata*, which all found to be true for Hugumburda-Gratkhassu Forest (Appendix 9).

As it was stated by Urban *et al*. (2000), altitude, climate, soil, and disturbance variations and the response of plants to these gradients determine patterns of vegetation composition in the environment. Incidence of an altitudinal zonation in the vegetation of Ethiopia was described by many researchers (e.g., Hedberg, 1964; Lissanework Nigatu and Mesfin Tadesse, 1989; Ermias Aynekulu, 2011). Elevation gradients are multifaceted and comprise diverse interrelated factors, for instance landscape, soil texture, moisture and climate (Hedberg, 1964; Austin *et al.,* 1996). Consequently, they control growth and development of plants and the patterns of vegetation distribution in an area. According to Körner (2000), among these composite variables, temperature and other climatic factors appear to be most significant for describing species richness by means of altitudinal gradient.

The five plant community types in this study showed little overlap in either the NMDS or CCA ordinations, indicating that they are largely distinctive in terms of their species composition and environmental factors. The NMDS ordination bi-plot (Fig. 10) showed the association of species in five distinctive plant communities along the environmental gradients. The NMDS axis 1 is significantly positively correlated to elevation, aspect, grazing and human disturbance, while negatively correlated to slope and EC (Fig. 10). Similarly, CCA of stands (Fig. 11) showed the relationship between vegetation data and

environmental variables. Axis 1 reflects aspect, grazing and human disturbance, while axis 2 reveals a gradient of altitude. The canonical correspondence analysis coefficients revealed that altitude, slope and disturbance were most significant in determining variation in species composition. In total, altitude was the most statistically significant environmental variable in determining general species distribution and association (Fig. 11), followed by slope, aspect and disturbance at $P < 0.05$. However, pH, EC and silt were non-significant in explaining the total species distribution patterns in the Forest.

In Ethiopia, ecological study by Tamrat Bekele (1994) and Ermias Aynekulu (2011), reported importance of altitude and slope in separation of forest communities. They ultimately influence species distribution by affecting soil texture and micro-climatic factors (Bolstad *et al.*, 1998). According to Hodgson (1978), moderate slope at lower altitude helps the formation of profound soils with a better water retaining ability; in the study area Community Type five was found to occupy such habitats. Thus, the higher plant richness in Community Type five may be due to higher site productivity created by such mild climate and deep and moist soils (Flombaum and Sala, 2008). Whereas, community types one and two are found on the gentle slopes, and are clearly separated along the CCA axis 1, which is associated with higher human and grazing disturbance. Therefore, low level of species richness and diversity in communities one and three (Table 11) is due to higher grazing and human disturbance in these communities. Disturbance affects species composition and diversity by affecting certain species and size classes, and by changing the light environment of the understory (Engelbrecht *et al.,*

2007). On the other hand, occurrence of intermediate disturbance in the area of *Acacia etbaica - Balanites aegyptiaca* Community Type contributes for higher species diversity.

5.1.1.3. Species population structure

We found that, the species population structure of the study area was represented by an inverted J-shape distribution pattern (Figs. 13 & 14). This confirms a general pattern of normal population structure where the number of individuals decreases as the height and DBH class increases, indicating the Forest is in a stage of secondary regeneration. Tamrat Bekele (1993) and Fekadu Gurmessa *et al.* (2012) had reported similar results from the different Afromontane forests of Ethiopia. This type of population structure is typical of many tropical forest tree species (Cesar, 1992), and is a sign of good regeneration ability. However, this pattern does not show trends of population dynamics and recruitment processes of a given species. Investigation of population structures for each individual tree species could offer more sound and detailed information for conservation actions.

DBH measurements of some selected tree species revealed different patterns of population structure, showing diverse population dynamics between species (Fig. 15). An inverted J-shaped distribution pattern was shown by *Podocarpus falcatus* (Fig. 15a), where the species frequency was highest in the lower diameter classes and decreased gradually towards the higher classes, confirmed good regeneration. Conversely, different distribution patterns for instance broken inverted J-shaped (*Ekebergia capensis*); bell-shaped (*Maytenus arbutifolia*); U-shaped (*Teclea simplicifolia*); broken J-shaped (*Hagenia abyssinica*) and upward 'F' (*Osyris quadripartita*) were also investigated (Fig. 15). Therefore, different biotic and/or abiotic factors could have contributed for the absence of individuals in some DBH classes. According to Alemayehu Wassie (2007),

livestock-induced disturbances might be among the major factors constraining regeneration and recruitment of woody species and contributing, ultimately, to the decline of woody species populations in forests. Pueyo *et al*. (2006) reported that community structure was proved to be more sensitive to grazing effects.

Generally, distribution of population structure which deviates from inverted J-shaped pattern demonstrates historical disturbance of the vegetation by natural or manmade attributes. The species with irregular distributions were trees that are highly required by the local people. Comparable results to the aforementioned tree species population structure were also investigated in different Afromontane Forests of Ethiopia by Tamrat Bekele (1994), Feyera Senbeta (2006), Tadesse Woldemariam *et al*. (2008) and Getinet Masresha (2014).

With regard to basal area analysis, Hugumburda Forest fragments had shown higher values than that of Gratkhassu Forest fragments (Fig. 16). Density, competition and/or site productivity have an effect on the basal area of a forest. Density of woody plants (Table 12) in Hugumburda Forest remnants was higher than that of Gratkhassu. In addition, maturation stage, intensity of disturbance and altitudinal gradients of the two forest remnants were not the same, which might have contributed to the total basal area difference.

Kohyama (1992) described, growth form composition of communities can significantly affect the processes that are responsible for the vertical arrangement in the forest. Ecological behavior of the species and their site conditions are closely linked to growth forms (Raunkiaer, 1934). But, factors causing disparity in species richness may vary

between growth forms. In terms of growth form comparison, Gratkhassu Forest remnants had shown highest species richness compared to Hugumburda Forest remnants (Fig. 12). Higher percentage proportion of herbaceous species and diversity in this forest remnants possibly relate to the diversity of habitats and altitudinal gradients. Whereas, presence of ferns and epiphytes in Hugumburda Forest remnants indicated higher rainfall and humidity gradients than Gratkhassu Forest remnants. This corresponds with the findings of Zotz *et al.* (1999), who reported epiphyte richness as an indicator of moisture gradients in tropical rainforests. Generally, herbaceous growth forms were abundant in Hugumburda-Gratkhassu Forest indicating a diversity of local habitats due to the difference moisture regimes, disturbance and edaphic gradients (Whittaker *et al*., 2001).

5.1.1.4. Regeneration status of Hugumburda-Gratkhassu Forest

Densities of age classes (seedlings, saplings and adult plants) of forest species are indications of present and future status of regeneration of species and they provide evidence or basic information for conservation priorities of forest species. When the forest has high number of seedlings but most of them do not attain the next age classes, there could be disturbances in the form of trampling or grazing which selectively removes juvenile stages.

Analysis of regeneration status of woody species in the entire Hugumburda-Gratkhassu Forest showed that the frequency distribution of age classes successively decreased from lower to higher age classes (Fig. 17). Separate analysis of age classes for tree, shrubs and liana growth forms showed considerable variations in the trend of their frequency distribution. In all cases normal frequency distribution of age classes were observed

where the number of seedlings exceeded the number of saplings and the number of saplings was greater than the number of adult plants (Fig. 17). However, in liana species frequency distribution of individuals in the lower age class (seedlings) was lower than the higher age class (saplings) suggesting that regeneration of lianas was not as good as that of shrubs and trees.

Separate analysis of regeneration status of woody species in the two forest remnants showed that the frequency distribution of age classes successively decreased from lower to higher age classes (Fig. 18a and b). However, separate investigation of growth forms demonstrated significant variation in the extent of frequency distribution of individuals at each class. In Gratkhassu, all growth forms follow more or less normal distribution even though their density proportion was different. Higher density of individuals was contributed by shrubs as compared to trees and lianas (Fig. 18a). In Hugumburda, shrubs and trees showed normal frequency distribution of age classes, where the number of seedlings exceeded the number of saplings and the number of saplings was greater than the number of adult plants (Fig. 18b). However, in liana species frequency distribution of individuals in the lower age class was lower than the higher age class, suggesting that regeneration of lianas was not as good as that of shrubs and trees.

Analysis of seedling and sapling density of the two forest fragments indicated that both forest fragments were found to be dominated by few species (Table 13). For instance Fabaceae the most species rich (11 species) family in each forest fragment and the other families were represented by few species (less than 5 species) in the seedling and sapling population of each forest remnant.

Result of regeneration status of the two forest remnants (Gratkhassu and Hugumburda) indicated that the ratio of seedlings to saplings was high in both forest remnants (Table 14). Similarly, the ratio of saplings to mature plants was high in Gratkhassu (3.44) and low in Hugumburda (Table 14), demonstrating that the two forest remnants were in a good reproduction potential corresponded with recruitment status, putting the forest into healthy or good regeneration status.

Seedling density of trees and selected shrub species was used in the priority setting for conservation (Table 15). On the basis of the result, species were grouped into three categories or priority classes. The number of species placed in each category of the two forest remnants was different. This showed that the level of management or disturbance of the forest remnants might be different. Although there was variation in each category of the two forest remnants, in the whole study area seven species were placed in category I, and seventeen species were grouped in category II and the remaining species were placed in category III (Table 15). Species under category I and II be supposed to get fist priority for conservation actions, as they were found without or represented by few seedlings. On the other hand, species in category III need monitoring management. Comparable studies carried out in different forests of Ethiopia (Shiferaw Belachew, 2006; Fekadu Gurmessa *et al.,* 2012; Getinet Masresha, 2014) reported results that verify some tree species to be given immediate conservation attention and large number of species require monitoring management.

Woody species regeneration pattern can be represented by patterns of regeneration of selected tree species (Fig. 19a-f). The first regeneration pattern was an inverted J-shape (Fig. 19a pattern) which showed the highest density in the lower size classes with gradual

decrease towards the bigger size classes. This pattern showed healthier regeneration status of the species where the large number of seedlings established, adequate number of which reach the sapling and enough amount of saplings developed to adult stage which in turn would produce seeds for the continuity of the generation (Silverton and Lovett-Doust, 1993; Tamrat Bekele, 1994; Demel Teketay and Granstrom, 1997). Such pattern of regeneration reflected minimum disturbance of the vegetation. *Nuxia congesta, Juniperus procera* and *Olinia rochetiana* were examples of species exhibiting this pattern.

The second regeneration pattern exhibited maximum density of seedlings and adult plants compared to densities of saplings (Fig. 19b). Such pattern reflects good reproduction potential of the species interrupted by selective disturbance of saplings which may be due to selective removal or most seedlings perished before reaching the stage of saplings due to unfavorable environmental conditions. The species exhibiting such pattern was *Ekebergia capensis.* The third type of regeneration pattern (Fig. 19c) was seen by *Maytenus undata,* having few numbers of seedlings and adult plants compared to saplings. This pattern indicated a poor reproduction and recruitment of species which may be associated with intense competition from the surrounding trees. Feyera Senbeta *et al*. (2007) and Woldeyohaness Enkossa (2008) reported similar reasons for bell-shaped population structure.

The fourth pattern was J-shape curve (Fig. 19d), where the densities of mature trees and saplings exceeded densities of seedlings frequency, which reflected poor reproduction potential but good recruitment status. Such pattern indicates presence of disturbance in the early stage of growth or unfavorable microenvironment for germination. The species

exhibiting such pattern was *Heteromorpha trifoliata.* With the fifth pattern of regeneration (Fig. 19e), species were represented by adult individuals with no or few number of individuals in the lower classes. This pattern showed poor reproduction and hampered regeneration (Bhuyan *et al.,* 2003; Khumbongmayum *et al.,* 2006). Species which represent such pattern was *Discopodium penninervium.* The sixth pattern was represented by *Ehretia cymosa*, the pattern that exhibited upward 'F' where individuals concentrate in the first (seedlings) and second (saplings) size class and few or no individuals in the matured size class (Fig. 19f).

Reasons for inadequate seedlings and saplings of some tree species in the NFPA might be many, including grazing and browsing, seed predation, lack of safe sites for seed recruitment, or the need of dormancy period for seeds of certain trees. Besides, moisture stress, pathogens and possession of alternative adaptations for propagation other than seed germination could also be the cause for lack of sufficient seedlings.

5.1.2. Edge effect on plant diversity and population structure

Tropical forests are known to harbor more than half of the global species richness, and they are often subjected to increasing anthropogenic pressure, which create a great threat to existing biodiversity (Myers *et al.,* 2000). The East African montane forests are known to be rich in species diversity as they are located within the region of high rainfall and habitat heterogeneity (Madoffe *et al*., 2006). This study had shown Asteraceae, Fabaceae and Poaceae to be the most dominant families in the edge - intermediate - interior habitats. Structural, physiological and genetic adaptation such as increased power of

germination as well as ability to withstand the vagaries of extreme climatic conditions and biotic competition may have contributed to the successful establishment of these dominant families in the edge - intermediate - interior habitats.

The most frequent species in the edge gradient study were *Juniperus procera* with a relative frequency of 86.74%, followed by *Hypoestes forskaolii* and *Carissa spinarum* with a relative frequency of 85.85% and 80.95% respectively. Of the total 288 species observed in this part of the study, 13 species were recorded solely in the interior habitat, 30 species in the intermediate habitat, and 40 species in the edge habitat. One hundred thirteen species were found to be common to the three studied habitats. Therefore, these plant species (Appendix 5) can be categorized based on their frequency of occurrence into three groups: (1) forest specialists (intermediate and interior species), (2) edge specialists and (3) generalist species respectively.

Results show that the species accumulation curves displayed an increasing trend, which suggest that increasing the sampling effort could have increased the species richness observed in each habitat. This is due to the fact that the larger the forest area sampled is the more environmentally heterogeneous the sampling area becomes and hence the higher the possibility of having many species. The observed trend in rarefaction curves (Fig. 21) coincided with the second order Jackknife species richness estimator (Table 17), which provides higher species richness estimates than the observed (Fig. 22) in each studied habitat. As it was depicted in Fig. 19, the observed species accumulation curve of edge habitat was outside of the 95% confidence interval of the intermediate and interior habitat revealing the edge habitat had low species richness compared to the intermediate and interior habitats. Therefore, this observation provides a clear indication that the interior

and intermediate habitats were rich and had a high recruitment rate than the edge habitat. The observation can possibly be linked with enduring anthropogenic disturbances observed at the edge habitat whereby the number of dead stems (snags and logs) was significantly higher in this habitat compared to the forest interior habitats. Large numbers of snags and fallen trees were observed in the edges (Fig. 25), a pattern that was common in other studies too (Ranney, 1977; Williams-Linera, 1990; Kacholi, 2014).

Results had shown a clear pattern of changes in the plant species community whereby the species diversity, richness, density and basal area increased towards the forest interior (Table 16, Fig.s 23 and 24). In all cases, the forest intermediate and interior habitats had significantly higher values than the edge habitat. Therefore, the result indicated the edge effect can penetrate to a distance of 100 m from the forest margin in the study area. Also, these findings suggested that the forest edge and the intermediate/interior were contrasting habitats. Besides, the present findings are comparable to other works done in Tanzania, Brazil and other countries, in which the forest interior was found to be rich in species diversity and the penetration of edge effects was reported to vary from 15 m to 200 m (Williams-Linera, 1990; Benitez-Malvido, 1998; Gehlhausen *et al.,* 2000; Kacholi, 2014).

The presence of less species diversity, richness, density, and basal area at the edges compared to intermediate and interior habitats can be linked to the fact that these habitats were easily accessible by local people while fetching for their basic needs like firewood, construction poles, farm implements and traditional medicines. The regular illegal timber harvest in the edge habitat contributes to the observed low values, especially on woody species density and basal area on the edge habitat, as it was easier for local people to cut

and carry logs during the night and nonappearance of forest guards. The grazing and livestock trampling (Appendix 11) can also be factors for the observed low density of seedlings (2,739 per ha) and saplings (2,275 per ha) at edges compared to 3,542 seedlings and 2,806 saplings per hectare (at the interior) (Fig. 26). Since, these disturbances can kill the sensitive species and affect their regeneration (Saunders *et al.,* 1991). Different abiotic conditions, vegetation structure, tree mortality rate, and high predation, loss of pollinators, and seed dispersers, at edges, were reported to be the common cause of differences in species composition between forest edge and interior in various studies done elsewhere (Williams-Linera, 1990; Benitez-Malvido, 1998; Gehlhausen *et al.,* 2000; Oosterhoorn and Kappelle, 2000).

The presence of less woody species density at the edge habitats may also be the result of the interaction of different factors. These could be due to those factors that reduce the possibility of seedling establishment (Fig. 26), as seedling is the first size class to be affected by edge effects (Saunders *et al.,* 1991; Benitez-Malvido, 1998) and reduction in seedling recruitment at edges due to uprooting and breakage due to wind turbulence, seedling damage caused by an increasing disturbance near forest edges and easy accessibility to edges by locals and their livestock (Laurance *et al.,* 1998; Cochrane and Laurance, 2002). In addition, the low basal area at the edge and intermediate habitats (Fig. 22) could be the result of enduring anthropogenic activities especially illegal logging done by unfaithful locals. As various authors had indicated, the presence of harsh micro environmental conditions such as soil and air temperatures, high light transmittance, lower relative humidity (Didham and Lawton, 1999), increased wind forces, lower availability of soil carbon, total nitrogen and phosphorus, and lower soil and

litter moisture (Jose *et al.,* 1996) near edges may contributed to significant changes in the abundance and distribution of plant species in the forest.

In a study of edge effects on vegetation structure and regeneration in Panama, Williams-Linear (1990), observed that the number of seedlings and saplings was much higher near the edge than in the interior forest habitat. Similar results have been documented elsewhere (Palik and Murphy, 1990; Brothers, 1993). However, in the present study it was found the reverse. The number of seedlings and saplings were lower near the edge than the intermediate and interior of the forest habitat (Fig. 26) a result that is consistent with prior studies (Chen and Franklin, 1992; Murcia 1995; Goldblum and Beatty, 1999; Gehlhausen *et al*., 2000; Sampaio and Scariot, 2011), probably because the edge environment had less favorable moisture and temperature conditions for seedling establishment and survival (Chen *et al.,* 1990) or seedling damage caused by disturbance near forest edges and easy accessibility to edges by locals and their livestock (Laurance *et al.,* 1998; Cochrane and Laurance, 2002) in the study area.

5.1.3. Land use\land cover dynamics

Efficient executive of natural resources, especially forest priority areas require accurate and up-to-date information to guide administrative bodies in making appropriate decision. This study provides such information using GIS and remote sensing technologies to guide policy development in managing the NFPA. By utilizing remote sensing technologies and implementing GIS mapping techniques, major land use\land cover types of the study area, over the past 38 years, had been identified and quantified. The accuracy level of the study was well over the minimum requirement of 85% as set by

Anderson *et al.,* (1976) revealing that the land use\land cover classification of the study area was valued.

Ecosystems are not static but exposed to a certain natural and human-induced disturbance regimens that caused temporal changes. In consequence, changes in disturbance regimes are likely to affect diversity, composition, community distribution, structure and functioning (Jentsch *et al*., 2002). Such responses get stronger with increasing magnitude and frequency of the disturbances. In this study, sequence of categorization of the land use\land cover types changed through the study period, due to shift of one category to another. This shift showed dynamicity of landcover categories over time.

This study used comprehensive approach to understand the past, present and future conditions of the study area, by employing remote sensing and GIS tools supported with field work. Finally, the study comes up with the following major findings:

- Generating the thematic landcover maps for change detection and comparisons using landsat images,
- Analyses of landcover dynamic factors such as rate of landcover change and conversion matrix, and
- Investigate the different landuse structural change using basic statistical indicators and indices for a better design of decision support system in sustainable natural resource management of the study area.

Moreover, findings of the study reveal that the study area has been under continual land use\land cover changes since 1973. Deforestation due to population growth and the associated expansion of farming and increasing demand for resources were imposing

threat on biodiversity of the area. In the study area, based on the change detection matrix, robust transformation of land use\land cover types were observed. In the first reference period, 72.9% of the total area of Hugumburda-Gratkhassu NFPA remained unchanged and 27.1% of the area changed from one category to another whereas in the second reference period 69% of the total area of the NFPA remained unchanged and 31% of the area changed from one category to another.

In the first reference period (1973 - 1986), a noticeable transformation was made from grassland, settlement and bare land to farm land. During this time, there was also extension of forest and shrub land, and tapering of certain cover types. The second period (1986 - 2011) is also characterized by reduction of grassland and expansion of forest, shrub land and cultivated land. The evident increase in forest and shrub land may be due to regulation of desertion of selective logging through government intervention in recent times. Throughout the study period forest and shrub land were the most dominant landcover types (Table 19).

On the other hand, increase of cultivated land from 4,719.7 ha in 1973 to 5,676 ha in 2011 was merely population induced and it was mainly at the expense of grassland, settlement and bare land. This is in agreement with the result of Pahari and Marai (1999) and Abate Shiferaw (2011). This showed that the impact of population pressure on Hugumburda-Gratkhassu NFPA and surrounding area at recent years had been increasing and it needs serious attention and appropriate solution. Land use\land cover dynamics is a result of complex interactions between several biophysical and socio-economic conditions. The effects of human activities are immediate and often radical, while the

natural effects take a relatively longer period of time. The difference in increase by households and landcover change indicates the pressure on vegetation cover and related biodiversity. This implies that population pressure was believed to be one of the major driving forces for the changes in the study area. In order to make relevant conclusions and recommendations of an area, one has to properly consult situations of the past and present, i.e. socio-economic and biophysical aspects of the area. Hence, in the case of this analysis, the major driving force to changes in LULC was increased population change. Such impact of population pressure on dynamics of land use\land cover was also reported by Abate Shiferaw (2011) and Getinet Masresha (2014).

Trends in land use\land cover during the two reference periods showed, the total area covered by green vegetation (forest, shrub land and grassland) in each of the study years (1973, 1986 and 2011) was greater than the total area covered by the remaining landuse types (cultivated land, settlement and bare land) which was highest in 2011 (71%) and low in 1973 (66%). However, considerable amount of one land use\land cover type was transformed to another. While considering the whole range of time, the rate of landcover changes clearly showed that cultivated land and forest land showed great change as compared to the other landcover classes, (i.e. forest and cultivated land showed the highest positive change, and grassland and bare land showed the highest negative change) whereas settlement showed modest change (Table 20). Increase in cultivated land can result in the reduction of natural forest vegetation of the study area which in turn would affect status of biodiversity. Thus, appropriate protection and conservation actions are required. Generally, both quantifiable and non-quantifiable socio-economic factors can

be major causes of vegetation cover change in the study area. This resource has been changed due to population growth (with other variables) such as demand for forest products for construction, fire wood and expansion of various types of agricultural activities and built-up areas along the margin. This circumstances lead to further depletion of forest resources in the study area.

Darbyshire *et al*. (2003) noted that, Northeastern Ethiopia including the study area had a long history of settlement. This long history of settlement has been followed by population growth. There has been little regulation of family size since children are considered an asset in the struggle for survival, as well as security in old age. Therefore, population pressure definitely was the most driving force in the observed land use\land cover dynamics. Population pressure is often considered to be an important driving force of deforestation (Pahari and Marai, 1999). This is because population growth created demand of land for various purposes. Environmental consequence of population growth thus have been a reduction in fallow periods and soil exhaustion, cultivation of shallow soils and steep slopes (Abate Shiferaw, 2011) followed by accelerated erosion and over-exploitation of forest.

According to Badege Bishaw (2009), lack of consistency in land tenure system during the different government regimes of Ethiopia made farmers lose their confidence in the security of their rights to the land. This led farmers to seek short-term needs rather than long-term conservation for sustainable utilization of the land resources which resulted in ecological damage. Another related driving force for land use\land cover change in the

study area was overstocking of livestock. Failing of rain-fed agriculture due to uncertainty of rainfall and increased intensity of drought resulted in massive food shortages (Girma Tadesse, 2001) which influence farmers to change the landuse pattern. Following this, farmers are shifting to animal husbandry (Galvin *et al.,* 2001). In Ethiopia, particularly in highlands like the study area, livestock have become a means of income diversification and a drought-coping strategy. Therefore, livestock grazing in high densities and with static grazing patterns may alter composition, reduce biodiversity, increase soil compaction and in extreme cases eliminate vegetation cover altogether (White *et al*., 2002).

5.1.4. Effect of habitat fragmentation on plant species richness

Fragmentation of a landscape is of greater concern, because it creates a natural imbalance in terms of size, shape and distribution of mosaic of patches found within the human dominant landscapes (Riiters *et al*., 2000). The significance of this is that it influences the dynamics of species and material in the landscape (Forman, 1995), giving various ecosystems their unique structure and function. Landscape metrics have been helpful tools in understanding the ecological implications of fragmentation and human-activities on the landscape. In this study, the effects of landuse change on the landscape had been quantified using various metrics (Tables 23 and 24).

The result showed an increase in forest and shrub landcover throughout the study period (from 1973 to 2011), indication of aggregation than fragmentation in the study landscape. Forest aggregation can be demonstrated through decrease in number of patches, an increase in their mean area, decrease in shape complexity and decrease in the distance

between or degree of isolation of patches (Table 23). Thus, the study confirmed an increase in forest and shrub landcover between 1973 and 2011 (Tables 23 and 24). On the other hand, the overall intensity of landscape fragmentation fluctuated throughout the period of study and suggests that the study area sustained varying levels of pressure, and this point to an overall increase in fragmentation over the period of study.

Increase in NP and decrease in mean patch area (Table 23) between 1986 and 2011 indicates that fragmentation strengthened during this period, giving rise to a complex assemblage of isolated and diverse landscape patches and ecological processes. Associated with this were increased edge habitats and their effects (Couvillion, 2005), and greater loss of connectivity. This is because edges have the tendency to change the biological and physical conditions around patch boundaries and within adjacent patches (Harper *et al*., 2005). The result also indicated presence of several settlement (820.7 ha) and cultivated areas (5676 ha) within the study area. This can facilitate more fragmentation and deforestation, the most important threats to the area's biota (Pahari and Marai, 1999; Abate Shiferaw, 2011), that is, the conversion of the forest area for agriculture, and illegal logging for timber and construction.

Table 23 demonstrated, forest land, shrub land and cultivated land were, in general, similar, occupying larger proportion of the landscape (or larger PLAND), having large mean patch sizes (AREA_MN), small mean Euclidean nearest neighbor distances and small values of the interspersion and juxtaposition index. On the other hand, grassland, settlement and bare land, were, in general, similar, having a high number of patches (Table 23), small mean patch sizes, large mean Euclidean nearest neighbour distances

and high values of the interspersion and juxtaposition index (Table 23). This may be explained by the fact that the study area was in good reforestation activity but with significant disturbance which could be related to more fragmentation of the natural LULC classes that could be an indication of human intervention.

A lot of small patches do not present the same habitat as a single larger patch, mainly for plants that require interior habitat. Therefore, formation of more simple or fractal patches in year 2011 indicated variability in habitat opportunity. Habitat conditions therefore became more heterogeneous in year 2011 than in 1986. The 100 percent increase in NP (10,190 to 20,399) from 1986 to 2011 (Table 24) could be attributed to increased human activities. Much of such activities could be credited to increase in impervious layers across the landscape. This increase could be attributed to increase in population and agricultural activities. These have been found to have positive relationship with increase in NP and hence increased landscape mosaics (Xu *et al.*, 2010).

The observed decrease in NP (10,190) during 1973 - 1986 does not indicate a reduction in human activities on the landscape, but could rather be the result of similar isolated patches joined to one another. Such aggregation of similar patches takes place when corridors are eliminated between similar patches, their connectivity increased and their edges joined together. Smaller parcels of agricultural fields may have aggregated together by either, clearing and putting under cultivation more forest patches existing between the plots, or such patches replaced by other landuse classes such as through afforestation. This assertion could be attributed to the land laws, especially those on agricultural land and forest law that came into effect during this period, hence the 68 percent drop in NP

between 1973 and 1986. Results for Largest Patch Index (LPI) clearly go with this assertion, between 1973 and 1986, LPI increased from 70.48 to 70.64.

Analysis for patch area and species abundance indicated that, the patches with the largest patch sizes were also having high species richness. Species numbers increased significantly with increasing habitat area (Table 26). This confirmed the general validity of species - area relationships (Bruun, 2000; Honnay *et al.,* 2010). There was a general positive relationship between species richness and fragment size. The only exception was P13 where slightly more species were recorded in the smallest, than in the medium-sized fragment (Table 26).

When the species-area correlations were investigated the role of patch area remained highly significant. This supports the importance of patch area in explaining the plant species richness. However, the computed patch fractal dimension index (FRAC) for all patches showed values that were greater than 1.0 (Table 25), which indicated complexity in shape among the forest patches. For most patches, this observed complexity in shape was critical because this implies an increased likelihood of contact between interior species and edge species (Bagarinao, 2011). Though species have differential responses to edge conditions, increased contact of interior species to edge species increases stress and risks (Forman and Godron, 1986; Bagarinao, 2011). Similarly, an increase in the FRAC and SHAPE index suggests a general increase in fragmentation of the forest core area. This is important because total core forest area is related to both patch area and

patch shape (Table 25) and fragmentation of core forests will directly affect habitat suitability for particular species.

Core forest areas were closely related to the least disturbed areas of forest (i.e., in the center of forest patches). Consideration of change in core forest cover can therefore give a better indication of the effect of forest loss and forest health. Change in patch area (particularly of effective core forest area) had been found to be the most important variable in determining change in species number and community composition (Jha *et al.,* 2005).

There are many possible reasons for the absence of some species from the smaller patches (e.g. P11, P15 and P16). For instance, small fragments can only support small populations, which easily succumb to environmental, demographic and genetic changes and thus, can easily go extinct locally (Hobbs and Yates, 2003). Also, the larger fragments (P4, P6, P3 and P7) were more likely to have a greater variety of microhabitats due to the existence of a wide range of microclimatic conditions. This in effect, increases the extent of the resource spectrum and survival of species in larger patches compared to smaller ones (Begon *et al.,* 1996).

The isolation of patches from one another may result in the disruption of plant-animal mutualisms causing imbalances of pollinators, dispersers, herbivores and predators and the loss of viable populations (Burkey, 1989). Plant species do not all respond uniformly to the different matrices surrounding remnant patches because they are bound to be differences in susceptibility depending on life history traits related to dispersal, establishment and persistence (Ewers and Didham, 2006). This may be responsible for the absence or presence of some species at particular sites. Therefore, the maintenance of

large continuous patches with greater core area appears to be critical to the survival of species in the study area.

5.1.5. Threats and management practices in the forest

Alongside long history of human inhabitation, natural resources of Hugumburda-Gratkhassu NFPA have been influenced. Fast population growth in the area and its surrounding demanded extra land for agriculture, grazing and settlement. Moreover, demand for construction, timber or fuel wood and farm implement also increased. These pressures resulted in fragmentation of the natural remnant vegetation in the study area which in turn affected status of plant diversity. Findings of the study indicated that both habitat fragmentation and edge effect were impacting plant diversity of the area. Thus, appropriate conservation measures are mandatory. At present, conservation activities are taking place in the national forest priority area. There are guards to restrict exploitation of the forest from the local inhabitants and their livestock. Nursery activity focused on the restoration of some indigenous woody species that show poor regeneration statuses are taking place, soil conservation practice through terracing are also taking place and public awareness with regard to the significances of the forest is taking place. However, these conservation activities are not adequate compared to the extent of habitat fragmentation and edge effect in the forest priority area.

5.2. Conclusion

Hugumburda-Gratkhassu NFPA is a very important ecosystem for biodiversity conservation in the Region. However, the efforts to conserve this forest has not been as desired, where great disturbance resulted in fragmentation and patchiness in the past. Consequently, information on the existent status of the landscapes, and plant species diversity, structure and regeneration are inadequate. As seen from the result and discussion, in the study area, plant species diversity, richness and composition have been found to be high. Such features have resulted from habitat heterogeneity due to topographic variables which suit different plant species association. The NFPA was found to be home for some endemic and near endemic plant species due to differences in altitude and variable landscape features. The presence of endemic and near endemic plant species in the forest indicates the potentiality of the area for biodiversity conservation. Some of the species are newly recorded in the floristic region from the study area. It can be inferred that further exhaustive botanical exploration would result in additional new records in the floristic area.

Altitude and slope were the major environmental factors in division of the vegetation into plant community types. Result of frequency distribution of woody species showed high proportion of small sized individuals in the lower diameter classes indicating good recruitment potential of the forest patches and rare occurrence of large individuals. Such trend was probably caused by past disturbance of the original vegetation resulting in succession of secondary vegetation. In addition, the analysis of species population structure indicated that some tree species had abnormal population structure with no or

few individuals at a lower size classes. Moreover, assessment of regeneration status on the basis of age classes indicated that significant proportion of woody species were represented by few or no seedlings, entailing that they were under threat.

Land use\land cover change detection and analysis of habitat fragmentation revealed size of forest, shrub land and cultivated land increased in the two study periods. Expansion of agricultural landuse was found to be major reason for fragmentation and patchiness of the remnant natural vegetation and thus exposing plant species to various environmental effects at edge habitats. As a result of this, the forest interior habitats were found to possess higher species richness, diversity, tree density and basal area compared to the edge habitats. In addition, the edge habitats were characterized by high anthropogenic activities by having more snags and logs per ha than the forest interior and intermediate habitats.

The degree of landscape fragmentation, patch area and fractal dimension computed indicated, intensity of deforestation and forest fragmentation fluctuated throughout the period of study and suggest that the forest sustained varying levels of pressure, and this pointed to an overall increase in fragmentation over the period of study. Thus, to adequately conserve and ensure species survival and maintain species diversity in the study area, these fragments should be maintained and well managed. If the current fragmentation process in the study area remains unabated, it would endanger the survival of species and isolated patches will have a high rate of local extinction of species. Therefore, this condition calls for the implementation of strategies that can mitigate

ecological effects of fragmentation and patch isolation and requires the establishment of corridors to enhance patch-to-patch connectivity. Forest patches of the study area, which serve as important habitats, must be connected to allow greater area for species survival.

The proliferation of illegal logging and farming activities must be managed to curtail further habitat fragmentation and thus, reduce isolation and disturbance effects. Reforestation activities must be done to connect isolated patches that serve as important habitat for vulnerable species. However, they must be implemented in a manner that sufficient interior core areas are created. The ability to provide such options depends on an understanding of landscape processes and the ability to use this understanding to develop appropriate strategies. This study therefore proved the capability of remote sensing and GIS to quantify changes in natural resource over time. By considering the change in landscape structure over time, it was possible to identify areas of a landscape that were more fragmented and vulnerable to human intervention.

Hugumburda-Gratkhassu NFPA is a major reserve of a number of plant species in the Region. Although, it is rich in plant diversity, the efforts to conserve this forest has not been as desired. If the NFPA continues being fragmented, there will be an increase of edge related habitats, which will cause structural and floristic composition changes due to increased edge effect and the forest will face a great threat of losing its original biota especially the rare species. The landuse changes, habitat changes and fragmentation in the study landscape represent the most direct effects of humans on this important landscape. In time to come, rapid landuse changes are expected in the study landscape and they

would result into severely altered current matrix and further fragmentation. Therefore, the situation calls for wide range of sustained governmental support to promote an overall complementary in situ and ex situ conservation strategy for plant diversity of the Region. Accordingly, to prevent further fragmentation and to maximize survival of species in the study area the following recommendations were forwarded.

5.3. Recommendations

Outcome of this research showed that the NFPA is still under disturbance and even in the future, it would be hard to make the forest area free from disturbance. Therefore, to develop appropriate management actions which ensure sustainable utilization of the natural resources is required. Hence, the following recommendations were forwarded based on the findings and the conclusions drawn:

- The study area includes unique combination of plant diversity with endemic and near endemic elements and rare species. From the endemic species few are registered in the IUCN Red List Categories. Hence, responsible bodies and decision makers shall devise appropriate conservation measures to save these valuable plant species and their habitats.
- Substantial numbers of forest species were found to have irregular population structure and are in reduced regeneration status. To prevent local extinction of these species, present efforts of nursery establishment and plantation of indigenous species in the forest should be strengthened and extended.
- Illegal grazing and logging mainly in the edge habitat of the forest were found to be the principal drivers affecting ecological processes in the forest. Therefore, the Forestry Administration shall develop a long term plan and strategy for a

long term reduction of livestock density with in and around the forest area so as to meet sustainable utilization of natural resources.

- The need for conservation and management of the forest and natural resources requires influencing the uses of land and the sizes and distributions of populations to maintain specific habitat conditions. Accordingly, minimizing access and reducing settlement and cultivation within the protected area is perhaps the most effective means of conserving the remaining forest patches. Practically the priority area should be free from permanent human settlement and other destructive human interferences. Through participatory way and compensating for displacing households resettlement program should be taking on.
- Hugumburda-Gratkhassu NFPA has variable topographic land features varying from flat terrain to steep escarpment. Due to the difficulty of the land escapes botanical investigation was restricted to relatively accessible areas in the forest. Therefore, extensive botanical investigation, mainly focused on inaccessible parts is recommended.
- Reforestation activities in the study area must produce patches with shapes that create sufficient interior habitat and interconnected to each other. Creating sufficient interior habitat and corridors are important to protect plant species and maintain biodiversity of the area.
- Focusing on environmentally benign activities - since all economic activities could not affect the natural forest, those activities should be carefully studied and encouraged. The activities like bee keeping should be given attention and

conditions should be facilitated in order to maximize the economic return. To fulfill this, the traditional way of bee keeping should be replaced by modern honey production. It is by assuming that, if the modern honey production is going to reduce economic pressure of the population, the degree of impact on natural forest will be reduced. Therefore, focusing on and encouraging such economic activities which are environmentally benign is very important.

- Studies related to climate change through ecological modeling, soil seed bank and carbon sequestration are recommended.
- Long-term research to study micro environmental factors such as light availability, air and soil temperature, humidity and nutrients along the edge-interior gradient in the forest in order to determine their influence on plant species richness, composition and structure is obligatory.
- Resource managers, scientific community, NGO's and local communities need to come together for participatory research and action oriented management interventions. Also they need to adopt a paradigm shift in view of practicing an adaptive as well as landscape management approach to conservation.
- Further research on understanding the relationship between the spatial settings of the landscape ecosystems and the conservation of their functions and biodiversity is needed for sustainable management and development of the priority area.

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APPENDICES

Appendix 1 List of plant species recorded in Hugumburda-Gratkhassu NFPA

No	Scientific Name	Family	Vernacular Name (Tigregna)	Habit
1	*Abutilon hirtum* (Lam.) Sweet	Malvaceae	Necha	Shrub
2	*Abutilon longicuspe* Hochest. ex A.Rich.	Malvaceae	Thaeda-necha	Shrub
3	*Acacia abyssinica* Hochst.ex Benth.	Fabaceae	Cheae	Tree
4	*Acacia asak* (Forssk.)Willd.	Fabaceae	Sabansa	Tree
5	*Acacia etbaica* Schweinf.	Fabaceae	Seraw	Tree
6	*Acacia mearnsii* De Wild.	Fabaceae		Tree
7	*Acacia seyal* Del.	Fabaceae	Wancho	Tree
8	*Acacia tortilis* (Forssk.) Hayne	Fabaceae	Karora	Tree
9	*Achyranthes aspera* L.	Amaranthaceae	Mechalo	Herb
10	*Acokanthera schimperi (*A.DC.) Schweif.	Apocynaceae	Meroz	Tree
11	*Actiniopteris diamorpha* Pic. Serm*	Actiniopteridaceae		Herb
12	*Actiniopteris semiflabellata* Pic.Serm.*	Actiniopteridaceae		Herb
13	*Adiantum incisum* Forssk.*	Adiantaceae		Herb
14	*Aerva lanata* (L.) Juss. ex Schultes	Amaranthaceae		Herb
15	*Ajuga integrifolia* Buch-Ham. ex D.Don.	Lamiaceae	Ezniater	Herb
16	*Allophylus abyssinicus* (Hochst.) Radlkofer	Sapindaceae	Meara	Tree
17	*Aloe debrana* Christian	Aloaceae	Ere	Shrub
18	*Amaranthus caudatus* L.	Amaranthaceae	Hamlitalian	Herb
19	*Anarrhinum forskaohlii* (Gmel.) Cufod. subsp. *abyssinicum* (Jaub. & Spach) D.A. Sutton	Scrophulariaceae		Herb
20	*Andropogon abyssinicus* Fresen.	Poaceae	Saeribala	Herb
21	*Anthemis tigreensis* J.Gay ex A.Rich.	Asteraceae		Herb
22	*Anthericum angustifolium* Hochst. exA. Rich.	Anthericaceae		Herb
23	*Anthospermum herbaceum* L.f.	Rubiaceae		Herb

24	*Argemone mexicana* L.	Papaveraceae	Buru	Herb
25	*Argyrolobium arabicum* (Decne.) Jaub. & Spach	Fabaceae		Herb
26	*Arthraxon prionodes* (Steud.) Dandy	Poaceae		Herb
27	*Arundo donax* L.	Poaceae		Shrub
28	*Asparagus africanus* Lam.	Asparagaceae		Climber
29	*Asparagus racemosus* Willd.	Asparagaceae	Kestensto	Climber
30	*Asplenium abyssinicum* Fee	Aspleniaceae		Herb
31	*Asplenium aethiopicum* (Brum.f.) Bech.*	Aspleniaceae		Herb
32	*Astragalus atropilosulus* (Hochest.) Bunge subsp. *abyssinicus* (Hochst.) Gillett	Fabaceae	Tetem-agazen	Shrub
33	*Azadirachta indica* A. Juss*	Azadirachta	Nim	Tree
34	*Balanites aegyptiaca* (L.) Del.	Balanitaceae	Bedano	Tree
35	*Barleria eranthemoides* R.Br.	Acanthaceae		Herb
36	*Barleria orbicularis* Hochst. ex T. Anders.	Acanthaceae	Gurbia	Shrub
37	*Barleria ventricosa* Hochest. ex Nees	Acanthaceae		Herb
38	*Becium grandiflorum* (Lam.) Pic.serm.	Lamiaceae	Tebeb	Shrub
39	*Berberis holstii* Engl.	Berberidaceae	Mucha-eff	Shrub
40	*Bersama abyssinica* Fresen.	Melianthaceae	Mirkuz-zibe	Tree
41	*Bidens biternata* (Lour.) Merr. & Sherff	Asteraceae		Herb
42	*Bidens pilosa* L.	Asteraceae	Chegogot	Herb
43	*Buddleja polystachya* Fresen.	Loganiaceae	Metere	Tree
44	*Cadaba farinosa* Forssk.	Capparidiaceae		Shrub
45	*Cadia purpurea* (Picc.) Ait.	Fabaceae	Shilaen	Shrub
46	*Calotropis procera* (Ait) Ait.f.	Asclepiadaceae		Shrub
47	*Calpurnia aurea* (Ait.) Benth	Fabaceae	Hitsawits	Shrub
48	*Campanula edulis* Forssk.	Campanulaceae		Herb
49	*Canthium pseudosetiflorum* Bridson	Rubiaceae		Shrub
50	*Cardiospermum corindum* L.*	Sapindaceae		Climber
51	*Carduus nyassanus* (S.Moore) R.E.Fries	Asteraceae		Herb
52	*Carissa spinarum* L.	Apocynaceae	Agam	Shrub
53	*Cassipourea malosana* (Baker) Alston	Rhizophoraceae	Keyh-om	Tree
54	*Caylusea abyssinica* (Fresen) Fish. & Mey	Resedaceae		Herb

55	*Celosia anthelminthica* Asch. in Schwinf	Amaranthaceae		Herb
56	*Celsia scrophulariifolia* Hochest. ex A.Rich.	Scrophulariaceae		Herb
57	*Celtis africana* Burm. f.	Ulmaceae	Moto-koma	Tree
58	*Ceratostigma abyssinicum* (Hochst.) Schweinf & Aschers.*	Plumbaginaceae		Herb
59	*Cheilanthes farinosa* (Forssk) Kaulf*	Sinopteridaceae		Herb
60	*Chenopodium murale* L.	Chenopodiaceae		Herb
61	*Chenopodium schraderianum* Schult.	Chenopodiaceae		Herb
62	*Chloris gayana* Kunth	Poaceae		Herb
63	*Chrysopogon aucheri* (Boiss.) Stapf	Poaceae		Herb
64	*Cissus quadrangularis* L.	Vitaceae		Climber
65	*Citrus aurantifolia* (Christm.) Swingle	Rutaceae	Lemin	Shrub
66	*Clematis hirsuta* Perr. & Guill	Ranunculaceae	Hareg-hazo	Climber
67	*Clematis simensis* Fresen.	Ranunculaceae	Hareg-thirae	Climber
68	*Clerodendrum myricoides* (Hochest.) Vatke.	Lamiaceae	Shewha	Shrub
69	*Clutia abyssinica* Jaub. and Spach.	Euphorbiaceae	Hirtmtmo	Shrub
70	*Colutea abyssinica* Kunth & Bouche*	Fabaceae	Que-queta	Shrub
71	*Combretum collinum* Fresen	Combretaceae		Tree
72	*Commicarpus pedunculosus* (R. Rich.) Cuford.	Nyctaginaceae	Eznitaewa	Herb
73	*Convolvulus kilimandschari* Engl.	Convolvulaceae		Climber
74	*Conyza aegyptiaca* (L.) Dryand ex Ait	Asteraceae		Herb
75	*Conyza hochstetteri* Sch.Bip. ex A.Rich.	Asteraceae		Shrub
76	*Conyza hypoleuca* A.Rich.	Asteraceae	Tsaeda-kotsilo	Herb
77	*Cordia africana* Lam.	Boraginaceae	Awhi	Tree
78	*Cotula abyssinica* Sch.Bip. ex A. Rich.	Asteraceae		Herb
79	*Crassocephalum crepidioides* (Benth.) S. Moore	Asteraceae		Herb
80	*Crassocephalum macropappum* (Sch.Bip. ex A.Rich.) S.Moore	Asteraceae		Herb

81	*Crassocephalum rubens* (Juss. ex Jacq.) S.Moore	Asteraceae		Herb
82	*Crassula schimperi* Fisch & Mey	Crassulaceae		Herb
83	*Crepis rueppellii* Sch.Bip.	Asteraceae		Herb
84	*Crotalaria incana* L.	Fabaceae		Herb
85	*Crotalaria lachnophora* Hochst. ex A.Rich.	Fabaceae		Shrub
86	*Crotalaria pycnostachya* Benth.*	Fabaceae		Herb
87	*Cupressus lusitanica* Mill.	Cuppressaceae	Tsihdi-ferenji	Tree
88	*Cyanotis barbata* D. Don	Commelinaceae	Gemale	Herb
89	*Cyathula uncinulata* (Schrad) Schinz*	Amaranthaceae		Herb
90	*Cymbopogon pospischilii* (K. Schum.) C.E. Hubb	Poaceae	Saeriadi	Herb
91	*Cynanchum abyssinicum* Decne.	Asclepiadaceae	Hareg-gumgumo	Climber
92	*Cynanchum altiscandens* K. Schum	Asclepiadaceae		Climber
93	*Cynodon dactylon* (L.) Pers.	Poaceae	Tehag	Herb
94	*Cynoglossum lanceolatum* Forssk.	Boraginaceae	Teneg	Herb
95	*Cyperus alternitolius* L.	Cyperaceae	Senti	Herb
96	*Cyperus fischerianus* A. Rich.	Cyperaceae		Herb
97	*Cyphostemma cyphopetalum* (Fresen.) Desc. ex Willd & Drummond*	Vitaceae		Herb
98	*Datura stramonium* L.	Solanaceae	Mestenagir	Herb
99	*Debregeasia bicolar* (Roxb.) Wedd.	Urticaceae	May-awalie	Shrub
100	*Dichrostachys cinerea* (L.) Wight & Arn.	Fabaceae	Harshmersha	Shrub
101	*Dicrocephala chrysanthemifolia* (Bl.) DC.	Asteraceae		Herb
102	*Dicrocephala integrifolia* (L.f.) O. Kuntze	Asteraceae		Herb
103	*Digitaria abyssinica* (Hochest. ex A. Rich.) Stapf	Poaceae		Herb
104	*Discopodium penninervium* Hochst.	Solanaceae	Alhim	Tree
105	*Dodonaea angustifolia* L.f.	Sapindaceae	Tahsos	Shrub
106	*Dombeya torrida* (J.F.Gmel.) P.Bamps	Sterculiaceae	Buyak	Shrub
107	*Dovyalis abyssinica (*A.Rich.)	Flacourtiaceae	Mengolhats	Tree

	Warb.			
108	*Dovyalis verrucosa (*Hochst.) Warb.	Flacourtiaceae	Tuemtenay	Shrub
109	*Dyschoriste radicans* Nees	Acanthaceae		Herb
110	*Echinops macrochaetus* Fresen.	Asteraceae	Dander	Herb
111	*Echium plantagineum* L.	Boraginaceae	Ayni-ater	Herb
112	*Ehretia cymosa* Thonn.	Boraginaceae	Tuwlaga	Shrub
113	*Ekebergia capensis* Sparm.	Meliaceae	Kot	Tree
114	*Epilobium hirsutum* L.	Onagraceae	Embabagereb	Herb
115	*Equisetum ramosissimum* Desf.	Equisetaceae		Herb
116	*Eragrostis papposa* (Roem. & Schult) Steud.	Poaceae		Herb
117	*Eragrostis tenuifolia* (A. Rich.) Steud.	Poaceae		Herb
118	*Dichanthium foveolatus* (Del.) Roberty	Poaceae	Keyhsaeri	Herb
119	*Erica arborea* L.	Ericaceae	Hasti	Shrub
120	*Erythrococca trichogyne* (Muell. Arg.) Prain	Euphorbiaceae		Shrub
121	*Eucalyptus camaldulensis* Dehnh.	Myrtaceae	Keyh-bahirzaf	Tree
122	*Eucalyptus globulus* Labill*	Myrtaceae	Thaeda-kelamitos	Tree
123	*Euclea schimperi* (A.DC.) Dandley*	Ebenaceae	Kuliow	Shrub
124	*Euphorbia ampliphylla* Pax	Euphorbiaceae	Qulqwal	Tree
125	*Euphorbia hirta L.*	Euphorbiaceae		Herb
126	*Euphorbia monacantha* Pax	Euphorbiaceae	Hamashiro	Herb
127	*Euphorbia petitiana* A. Rich	Euphorbiaceae		Herb
128	*Euphorbia schimperiana* Scheele.	Euphorbiaceae	Tsebadimu	Herb
129	*Ficus palmata* Forssk.	Moraceae	Beles	Shrub
130	*Ficus sur* Forssk.	Moraceae	Shanfa	Tree
131	*Ficus sycomorus* L.*	Moraceae		Tree
132	*Flueggea virosa* (Willd.) Voigt.	Euphorbiaceae		Shrub
133	*Galinsoga quadriradiata* Ruiz & Pavon	Asteraceae	Shermuta	Herb
134	*Galium spurium* L.	Rubiaceae	Keshaeta	Herb
135	*Geranium arabicum* Forssk	Geraniaceae		Herb
136	*Gerbera piloselloides* (L.) Cass.	Asteraceae		Herb

137	*Glycinne wightii* (Wight and Arn.) Verdc.	Fabaceae		Climber
138	*Gomphocarpus fruticosus* (L.) Ait.f.	Asclepiadaceae	Enchie	Shrub
139	*Gomphocarpus purpurascens* A. Rich.	Asclepiadaceae		Herb
140	*Grewia ferruginea* Hochst. ex A.Rich.	Tiliaceae	Meleglega	Shrub
141	*Grewia mollis* A. Juss.	Tiliaceae	Reway	Tree
142	*Grewia tembensis* Fresen.	Tiliaceae	Chaka	Shrub
143	*Guizotia scabra* (Vis.) Chiov.	Asteraceae	Embabayohaness	Herb
144	*Hagenia abyssinica* (Bruce) J.F.Gmel.	Rosaceae	Habi	Tree
145	*Haplocarpha schimperi* (Sch. Bip.) Beauv.	Asteraceae	Hug-hugo	Herb
146	*Hebenstretia angolensis* Rolfe	Scrophulariaceae		Herb
147	*Helichrysum nudifolium* (L.) Less.	Asteraceae		Herb
148	*Helinus mystacinus* (Ait.) E. Mey. ex Steud.	Rhamnaceae		Climber
149	*Heliotropium cinerascens* DC. & A.DC.	Boraginaceae	Am-am-gimel	Shrub
150	*Heracleum abyssinicum* (Boiss.) Norman	Apiaceae		Herb
151	*Heteromorpha trifoliata* (Wendel.) Eckl. & Zeyh.	Apiaceae	Memowets akeytay	Shrub
152	*Hibiscus ovalifolius* (Forssk.) Vahl	Malvaceae		Herb
153	*Hibiscus panduriformis* Burm. f.	Malvaceae		Herb
154	*Hybanthus enneaspermus* (L.) F. Muell	Violaceae		Herb
155	*Hybanthus puberulus* M.Gilbert	Violaceae		Herb
156	*Hyparrhenia hirta* (L.) Stapf	Poaceae	Alenjega	Herb
157	*Hypericum quartinianum* A.Rich	Hypericaceae		Shrub
158	*Hypericum revolutum* Vahl.	Hypericaceae	Abedi	Shrub
159	*Hyperthelia dissoluta* (Steud.) W. D. Clayton	Poaceae		Herb
160	*Hypoestes forskaolii* (Vahl) R.Br.	Acanthaceae	Girbidia	Herb
161	*Indigofera costata* Guill. & Perr.	Fabaceae	Acasha	Herb
162	*Indigofera dendroides* Jacq.	Fabaceae	Egug	Herb

163	*Indigofera hochstetteri* Bak.	Fabaceae		Herb
164	*Indigofera spicata* Forssk.	Fabaceae		Herb
165	*Ipomoea sinensis* (Desr.) Choisy	Convolvulaceae		Climber
166	*Jasminum abyssinicum* Hochst. ex DC.	Oleaceae		Climber
167	*Jasminum dichotomum* Vahl	Oleaceae		Climber
168	*Jasminum grandiflorum* L. subsp. *f1oribundum* (R.Br. ex Fresen.) P.S.Green	Oleaceae	Tselim-habi	Climber
169	*Juniperus procera* Hochst. ex Endl.	Cuppressaceae	Tsihdi-adi	Tree
170	*Justicia caerulea* Forssk.	Acanthaceae		Herb
171	*Kalanchoe lanceolata* (Forssk.) Pers.	Crassulaceae	Saenihibey	Herb
172	*Kalanchoe marmorata* Bak.	Crassulaceae	Are-are	Herb
173	*Kniphofia pumila* (Ait.) Kunth*	Asphodelaceae	Ashenda	Herb
174	*Lactuca inermis* Forssk	Asteraceae		Herb
175	*Laggera crispata* (Vahl) Hepper and Wood	Asteraceae		Herb
176	*Laggera elatior* R.E. Fries	Asteraceae		Shrub
177	*Laggera tomentosa (*Sch. Bip. ex. A. Rich.) Oliv. & Hiern	Asteraceae	Kanskanso	Herb
178	*Lantana viburnoides* (Forssk.) Vahl.	Verbenaceae		Shrub
179	*Leonotis ocymifolia* (Brum.f.) Iwarsson	Lamiaceae		Herb
180	*Lepidium bonariense* L.	Brassicaceae		Herb
181	*Leucas abyssinica* (Benth.) Briq	Lamiaceae	Shumakerni	Shrub
182	*Leucas martinicensis* (Jacq.) R.Br.*	Lamiaceae		Herb
183	*Lippia adoensis* Hochst.ex.Walp.	Verbenaceae	Kusihe	Shrub
184	*Lycopersicon esculentum* Mill.	Solanaceae		Herb
185	*Maerua angolensis* DC.	Capparidiaceae	Kormediet	Shrub
186	*Maesa lanceolata* Forssk.	Myrsinaceae	Saweria	Tree
187	*Malva verticillata* L.	Malvaceae	Lhtit	Herb
188	*Maytenus arbutifolia* (A. Rich.) Wilczek.	Celastraceae	Dawija	Tree
189	*Maytenus senegalensis* (Lam.) Excell	Celastraceae	Kebkeb	Tree

190	*Maytenus undata* (Thunb.) Blakelock	Celastraceae	At-at	Tree
191	*Medicago polymorpha* L.	Fabaceae	Gurdiumahya	Herb
192	*Mentha longifolia* (L.) Hudson	Lamiaceae	Setisemhal	Herb
193	*Mentha spicata* L.	Lamiaceae	Naenae	Herb
194	*Meriandra bengalensis* (Konig ex Roxb.) Benth	Lamiaceae	Mesaguh	Shrub
195	*Meriandra dianthera* (Roth ex Roem & Schult.) Briq.	Lamiaceae	Mesagu	Herb
196	*Minuartia filifolia* (Forssk.) Mattf.	Caryophyllaceae		Herb
197	*Momordica foetida* Schumach.	Cucurbitaceae		Herb
198	*Monechma debile* (Forssk.) Nees	Acanthaceae		Herb
199	*Myrica salicifolia* A.Rich.	Myricaceae	Shihnet	Tree
200	*Myrsine africana* L.	Myrsinaceae	Kechemo	Shrub
201	*Nepeta azurea* R.Br.ex Benth.	Lamiaceae	Kuseha	Herb
202	*Nicandra physaloides* (L.) Gaertn	Solanaceae		Herb
203	*Nuxia congesta* R. Br. ex Fresen.	Loganiaceae	Tekarie	Tree
204	*Ocimum americanum* L.	Lamiaceae	Seseg	Herb
205	*Ocimum forskolei* Benth	Lamiaceae	Tebeb	Herb
206	*Ocimum lamiifolium* Hochest. ex Benth.	Lamiaceae	Ayni-birsin	Herb
207	*Ocimum suave* Willd	Lamiaceae		Shrub
208	*Ocimum urticifolium* Roth	Lamiaceae	Demekelbi	Herb
209	*Olea europaea* L. ssp. *caspidata* (Wall. ex G.Don.)	Oleaceae	Awlie	Tree
210	*Olinia rochetiana* A.Juss.	Oliniaceae	Ale-ale	Tree
211	*Oncocalyx kelleri* (Engl.) M. Gilbert	Loranthaceae		Herb
212	*Oncocalyx schimperi* (A. Rich) M. Gilbert	Loranthaceae	Dikala-om	Herb
213	*Opuntia ficus-indica* (L.) Miller	Cactaceae	Qulqwal bahri	Shrub
214	*Orobanche ramosa* L.	Orobanchaceae	Akenchira	Herb
215	*Osteospermum vailliantii* (Decne.) T.Norl.	Asteraceae		Herb
216	*Osyris quadripartita* Decn.	Santalaceae	Kerets	Tree
217	*Otostegia fruticosa* (Forssk.) Schweif. ex Penzig	Lamiaceae	Chamo	Shrub

218	*Otostegia integrrifolia* Benth	Lamiaceae	Chi-endog	Shrub
219	*Oxalis corniculata* L.	Oxalidaceae		Herb
220	*Oxygonum sinuatum* (Meisn.) Dammer	Polygonaceae		Herb
221	*Panicum hochstetteri* Steud.	Poaceae		Herb
222	*Panicum hygrocharis* Steud.	Poaceae		Herb
223	*Panicum maximum* Jacq.	Poaceae		Herb
224	*Panicum subalbidum* Kunth	Poaceae		Herb
225	*Pappea capensis* Eckl. & Zeyh.	Sapindaceae	Kolashihnet	Tree
226	*Parthenium hysterophorus* L.	Asteraceae	Kinche	Herb
227	*Pavetta oliveriana* Hiern	Rubiaceae	Shumeja	Shrub
228	*Pelargonium alchemilloides* (L.) Ait	Geraniaceae	Dander	Herb
229	*Pelargonium multibracteatum* Hochst. ex A.Rich.	Geraniaceae	Chewamrakut	Herb
230	*Pennisetum macrourum* Trin.	Poaceae		Herb
231	*Pennisetum polystachion* (L.) Schult.	Poaceae	Saerishife	Herb
232	*Pennisetum setaceum* (Forssk) Chiov.	Poaceae		Herb
233	*Pennisetum sphacelatum* (Nees) Th. Dur. & Schinz	Poaceae		Herb
234	*Pennisetum thunbergii* Kunth	Poaceae		Herb
235	*Pennisetum unisetum* (Nees.) Benth.	Poaceae		Herb
236	*Pennisetum villosum* Fresen.	Poaceae	Saerigama	Herb
237	*Peperomia abyssinica* Miq.	Piperaceae		Herb
238	*Periploca linearifolia* Quart.-Dill. & A. Rich.	Asclepiadaceae		Climber
239	*Phagnalon abyssinicum* Sch. Bip. ex A.Rich.	Asteraceae	Bosena	Herb
240	*Physalis peruviana* L.	Solanaceae	Comedere	Herb
241	*Phytolacca dodecandra* L.Herit.	Phytolaccaceae	Shimti	Shrub
242	*Pimpinella hirtella* (Hochst.) A.Rich.	Apiaceae	Shembekoseitan	Herb
243	*Pinus patula* Schiede ex Schltdl. & Cham.	Pinaceae	Bush	Tree
244	*Pittosporum viridiflorum* Sims	Pittosporaceae	Mayliho	Tree
245	*Plectranthus lanuginosus* (Hochst. ex Benth.) Agnew	Lamiaceae		Herb
246	*Poa leptoclade* Hochest. ex A.	Poaceae		Herb

	Rich.			
247	*Podocarpus falcatus* (Thun) Mirb.	Podocarpaceae	Zigba	Tree
248	*Poecilostachys oplismenoides* (Hack.) W. D. Clayton	Poaceae		Herb
249	*Polygala obtusissima* Chod.	Polygalaceae		Climber
250	*Polygala sphenoptera* Fresen.*	Polygalaceae		Herb
251	*Polypogon viridis* (Gouan) Breister	Poaceae		Herb
252	*Portulaca oleracea* L.	Portulacaceae		Herb
253	*Portulaca quadrifida* L.	Portulacaceae		Herb
254	*Pseudognaphalium oligandrum* (DC.) Hilliard & Burtt	Asteraceae		Herb
255	*Psiadia punctulata* (DC.)Vatke	Asteraceae	Saeriranda	Herb
256	*Psidium quajava* L.*	Myrtaceae	Zeitun	Tree
257	*Psydrax schimperiana* (A.Rich.) Bridson	Rubiaceae	Tsehag	Shrub
258	*Pteris dentata* Forssk	Pteridaceae		Herb
259	*Pterolobium stellatum* (Forssk.) Brenan	Fabaceae	Konteftefe	Shrub
260	*Rhoicissus tridentata* (L. f.) Wild & Drummond	Vitaceae	Keyh-hareg	Climber
261	*Rhus glutinosa* A.Rich.	Anacardiaceae	Tetaelo	Tree
262	*Rhus natalensis* Krauss	Anacardiaceae	Atam	Tree
263	*Rhus retinorrhoea* Oliv.	Anacardiaceae	Nefasito	Shrub
264	*Rhynchosia elegans* A. Rich.	Fabaceae		Herb
265	*Rhynchosia malacophylla* (Spreng.) Boj.	Fabaceae		Herb
266	*Ricinus communis* L.*	Euphorbiaceae	Guley	Shrub
267	*Rosa abyssinica* Lindley	Rosaceae	Kaga	Shrub
268	*Rubia cordifolia* L.	Rubiaceae		Climber
269	*Rubus steudneri* Schweinf.	Rosaceae	Mengolel	Shrub
270	*Rumex abyssinicus* Jacq.	Polygonaceae	Mokmoko	Herb
271	*Rumex nepalensis* Spreng.	Polygonaceae		Herb
272	*Rumex nervosus* Vahl	Polygonaceae	Enbacho	Herb
273	*Sageretia thea* (Osbeck) M. C. Johnston	Rhamnaceae	Kenchelchele	Shrub
274	*Sanicula elata* Buch.-Ham. ex D.Don	Apiaceae		Herb
275	*Sarcostemma viminale* (L.) R. Br.	Asclepiadaceae	Tsibtsibo	Shrub

276	*Satureja biflora* (Ham. ex Don) Briq.	Lamiaceae		Herb
277	*Satureja punctata* (Benth.) Briq	Lamiaceae	Shitalemin	Herb
278	*Scabiosa columbaria* L.	Dipsacaceae		Herb
279	*Schinus molle* L.*	Anacardiaceae		Tree
280	*Selaginella abyssinica* Spring*	Selaginellaceae		Herb
281	*Senecio hadiensis* Forssk.	Asteraceae	Suhumatali	Shrub
282	*Senecio lyratus* Forssk.	Asteraceae		Herb
283	*Senna occidentalis* (L.) Link	Fabaceae		Herb
284	*Senna sinqueana* (Del.) Lock	Fabaceae	Hambahambo	Shrub
285	*Silene burchellii* DC.	Caryophyllaceae		Herb
286	*Silene macrosolen* A. Rich.	Caryophyllaceae	Saerisaero	Herb
287	*Silene yemensis* Defl.	Caryophyllaceae		Herb
288	*Silybum marianum* (L.) Gaertn.	Asteraceae		Herb
289	*Snowdenia petitiana* (A. Rich) C. E. Hubb.	Poaceae		Herb
290	*Snowdenia polystachya* (Fresen) Pilg.	Poaceae	Mugia	Herb
291	*Solanum adoense* Hochst. ex A.Rich.	Solanaceae	Alalmo kelbi	Climber
292	*Solanum benderianum* Schimper. ex Dammer	Solanaceae		Climber
293	*Solanum campylacanthum* Hochst. ex A. Rich.	Solanaceae	Ingola	Herb
294	*Solanum incanum* L.	Solanaceae	Engule	Herb
295	*Solanum marginatum* L.f.	Solanaceae	Geberenbuyeh	Shrub
296	*Solanum nigrum* L.	Solanaceae		Herb
297	*Solanum schimperianum* Hochst. ex A.Rich.	Solanaceae	Berbereawald	Herb
298	*Sparmannia ricinocarpa* (Eckl. & Zeyh.) O.Ktze.	Tiliaceae		Shrub
299	*Spermacoce sphaerostigma* (A.Rich.) Vatke	Rubiaceae		Herb
300	*Sporobolus pyramidalis* P. Beauv.*	Poaceae		Herb
301	*Tagetes minuta* L.	Asteraceae	Dihanekel	Herb
302	*Teclea simplicifolia* (Engl.) Verdoom	Rutaceae	Salih	Tree
303	*Tephrosia uniflora* Pers.	Fabaceae		Herb
304	*Themeda triandra* Forssk	Poaceae	Hangalsaeri	Herb
305	*Thymus schimperi* Ronniger	Lamiaceae	Tesne	Herb

306	*Toddalia asiatica* (L.) Lam.	Rutaceae	Hareg	Climber
307	*Torilis arvensis* (Hudson) Link	Apiaceae		Herb
308	*Trachydium abyssinicum* (Hochst.) Hiern	Apiaceae		Herb
309	*Tragia cinerea* (Pax) Gilbert & Radcl-Smith.	Euphorbiaceae		Climber
310	*Trifolium baccarinii* Chiov.	Fabaceae		Herb
311	*Trifolium rueppellianum* Fresen	Fabaceae		Herb
312	*Triumfetta pilosa* Roth	Tiliaceae		Shrub
313	*Urtica simensis* Steud.	Urticaceae		Herb
314	*Verbascum pedunculosa* (Steud. & Hochst. ex Benth.) Kuntze	Scrophulariaceae		Herb
315	*Verbascum sinaiticum* Benth.	Scrophulariaceae	Tirnaka	Herb
316	*Verbesina encelioides* (Cav.) A. Gray*	Asteraceae		Herb
317	*Vermifrux abyssinica* (A. Rich.) Gillett	Fabaceae		Herb
318	*Vernonia amygdalina* Del.	Asteraceae	Girawa	Tree
319	*Vernonia bipontini* Vatke	Asteraceae	Mechalo	Tree
320	*Vernonia rueppellii* Sch.Bip. ex Walp	Asteraceae	Tetaso	Shrub
321	*Vernonia schimperi* DC.	Asteraceae		Shrub
322	*Veronica anagallis-aquatica* L.*	Scrophulariaceae		Herb
323	*Vigna membranacea* A. Rich.	Fabaceae	Haregefi	Climber
324	*Withania somnifera* (L.) Dunal	Solanaceae	Gineharas	Herb
325	*Ziziphus mucronata* Willd.	Rhamnaceae	Kunkura-hido	Shrub
326	*Ziziphus spina-christi* (L.) Desf.	Rhamnaceae	Kunkura(Geba)	Shrub

*= spcies collected out of quadrats

Appendix 2 Families, genera and species distribution of plants collected in the study area

	Family	Genera	Species			Family	Genera	Species
1	Acanthaceae	5	7		46	Melianthaceae	1	1
2	Actiniopteridaceae	1	2		47	Moraceae	1	3
3	Adiantaceae	1	1		48	Myricaceae	1	1
4	Aloaceae	1	1		49	Myrsinaceae	2	2
5	Amaranthaceae	5	5		50	Myrtaceae	2	3
6	Anacardiaceae	2	4		51	Nyctaginaceae	1	1
7	Anthericaceae	1	1		52	Oleaceae	2	4
8	Apiaceae	6	6		53	Oliniaceae	1	1
9	Apocynaceae	2	2		54	Onagraceae	1	1
10	Asclepiadaceae	5	7		55	Orobanchaceae	1	1
11	Asparagaceae	2	2		56	Oxalidaceae	1	1
12	Asphodelaceae	1	1		57	Papaveraceae	1	1
13	Aspleniaceae	1	2		58	Phytolaccaceae	1	1
14	Asteraceae	27	38		59	Pinaceae	1	1
15	Balanitaceae	1	1		60	Piperaceae	1	1
16	Berberidaceae	1	1		61	Pittosporaceae	1	1
17	Boraginaceae	5	5		62	Plumbaginaceae	1	1
18	Brassicaceae	1	1		63	Poaceae	20	31
19	Cactaceae	1	1		64	Podocarpaceae	1	1
20	Campanulaceae	1	1		65	Polygalaceae	1	2
21	Capparidiaceae	2	2		66	Polygonaceae	2	4
22	Caryophyllaceae	2	4		67	Portulacaceae	1	2
23	Celastraceae	1	3		68	Pteridaceae	1	1
24	Chenopodiaceae	1	2		69	Ranunculaceae	1	2
25	Combretaceae	1	1		70	Resedaceae	1	1
26	Commelinaceae	1	1		71	Rhamnaceae	3	4
27	Convolvulaceae	2	2		72	Rhizophoraceae	1	1
28	Crassulaceae	2	3		73	Rosaceae	3	3
29	Cucurbitaceae	1	1		74	Rubiaceae	7	7
30	Cuppressaceae	2	2		75	Rutaceae	3	3
31	Cyperaceae	1	2		76	Santalaceae	1	1
32	Dipsacaceae	1	1		77	Sapindaceae	4	4
33	Ebenaceae	1	1		78	Scrophulariaceae	5	6
34	Equisetaceae	1	1		79	Selaginellaceae	1	1
35	Ericaceae	1	1		80	Sinopteridaceae	1	1
36	Euphorbiaceae	6	10		81	Solanaceae	7	13
37	Fabaceae	19	31		82	Sterculiaceae	1	1
38	Flacourtiaceae	1	2		83	Tiliaceae	3	5

39	Geraniaceae	2	3		84	Ulmaceae	1	1
40	Hypericaceae	1	2		85	Urticaceae	2	2
41	Lamiaceae	12	22		86	Verbenaceae	2	2
42	Loganiaceae	2	2		87	Violaceae	1	2
43	Loranthaceae	1	2		88	Vitaceae	3	3
44	Malvaceae	3	5		**Total**		**238**	**326**
45	Meliaceae	2	2					

Appendix 3 New vascular plants recorded from Welo (WU) floristic region in the FEE

No	Species	Family	Habit
1	*Canthium pseudosetiflorum* Bridson	Rubiaceae	Shrub
2	*Cassipourea malosana* (Baker) Alston	Rhizophoraceae	Tree
3	*Celosia anthelminthica* Asch. in Schwinf	Amaranthaceae	Herb
4	*Cynanchum altiscandens* K. Schum	Asclepiadaceae	Climber
5	*Echium plantagineum* L.	Boraginaceae	Herb
6	*Hybanthus enneaspermus* (L.) F. Muell	Violaceae	Herb
7	*Pappea capensis* Eckl. & Zeyh.	Sapindaceae	Tree
8	*Poecilostachys oplismenoides* (Hack.) W. D. Clayton	Poaceae	Herb
9	*Portulaca quadrifida* L.	Portulacaceae	Herb
10	*Schinus molle* L.	Anacardiaceae	Tree

Appendix 4 Physiographic data of the sampled stands

Quadrat No	**Latitude (X)**	**Longitude (Y)**	**Altitude (m)**	**Slope (%)**	**Aspect**	**Grazing impact**	**Human impact**	**pH**	**EC (microS/cm)**	**% Sand**	**% Clay**	**% Silt**	**Textural Class**	**Locality**
G1	555669	1378552	2551	37	2.5	3	3	7.69	40.4	74	12	14	Sandy Loam	Tsiliageworgis
G2	555705	1378561	2537	40	2.5	2	2	7.47	47.4	72	10	18	Sandy Loam	Tsiliageworgis
G3	555354	1378521	2666	30	2.5	2	2	6.67	40.8	61	13	26	Sandy Loam	Tsiliageworgis
G4	555400	1378544	2636	33	2.5	1	1	6.5	33.6	77	11	12	Sandy Loam	Tsiliageworgis
G5	557733	1376822	2200	29	1	1	1	8.1	74.3	73	9	18	Sandy Loam	Adigasho
G6	557697	1376738	2146	28	1	1	1	7.48	152.5	63	13	24	Sandy Loam	Adigasho
G7	558511	1375797	1929	25	1	2	2	7.37	32.6	79	7	14	Loamy Sand	Adigasho
G8	558489	1375739	1892	24	1	0	1	7.6	51.5	73	11	16	Sandy Loam	Adigasho
G9	558450	1375651	1855	22	1	0	0	8.14	40.5	79	7	14	Loamy Sand	Adigasho
G10	558703	1378534	2414	32	2	3	3	7.11	73.4	72	10	18	Sandy Loam	Mistayha
G11	558547	1378599	2356	18	2	3	3	6.92	52.6	62	18	20	Sandy Loam	Mistayha
G12	558684	1378963	2393	30	2	3	3	7.1	65.5	82	8	10	Loamy Sand	Mistayha
G13	558482	1378853	2346	22	2	3	3	7.35	105.3	70	12	18	Sandy Loam	Mistayha
G14	557258	1378611	2409	25	1	3	3	6.9	108.9	78	10	12	Sandy	Higum

								8					Loam	
G15	557174	1378692	2381	30	1	2	2	7.67	73	64	18	18	Sandy Loam	Higum
G16	557236	1377575	2422	33	1	2	2	7.1	64.6	62	16	22	Sandy Loam	Higum
G17	557317	1377874	2329	33	1	3	3	8.1	93.4	59	17	24	Sandy Loam	Higum
G18	557374	1377979	2292	30	1	2	2	7.6	69.6	82	7	11	Loamy Sand	Higum
G19	555079	1379386	2688	33	1.3	1	1	7.45	40.5	82	9	9	Loamy Sand	Adibeshae
G20	555173	1379414	2625	34	3.3	0	1	8.2	49	78	13	9	Sandy Loam	Adibeshae
G21	555264	1379474	2556	35	1.3	0	0	7.35	45.4	80	11	9	Loamy Sand	Adibeshae
G22	555092	1378894	2719	30	2.5	0	1	6.5	37.1	82	9	9	Loamy Sand	Adibeshae
G23	555173	1378930	2666	33	1.3	0	0	6.91	31.8	84	5	11	Loamy Sand	Adibeshae
G24	555369	1377841	2702	31	3.3	1	1	6.8	45.7	62	17	21	Sandy Loam	Balago
G25	555483	1377886	2640	33	2.5	1	1	7.5	97.3	60	21	19	Sandy Clay Loam	Balago
G26	555496	1377199	2805	34	2.5	0	1	6.85	66.1	78	9	13	Sandy Loam	Balago
G27	555686	1377342	2640	33	2.5	2	2	7.2	99.7	62	17	21	Sandy Loam	Balago
G28	555789	1377423	2552	26	2.5	0	0	6.98	52.6	80	11	9	Loamy Sand	Balago
G29	556609	1376820	2459	27	3.3	2	2	7.3	47.5	53	19	28	Loam	Hawla
G30	556793	1376896	2322	32	3.3	1	1	8.4	70.3	72	13	15	Sandy Loam	Hawla

G31	556936	1376973	2229	33	3.3	0	1	7.2	59.5	72	11	17	Sandy Loam	Hawla
G32	556998	1376445	2359	26	3.3	1	1	7.3	39.4	86	7	7	Loamy Sand	Hawla
G33	557159	1376500	2214	30	3.3	0	1	7.2	73.5	76	7	17	Loamy Sand	Hawla
G34	558646	1375094	1878	30	3.3	2	2	6.52	32.2	86	9	5	Loamy Sand	Qaushira
G35	558785	1375174	1814	31	3.3	0	1	6.35	47.3	88	5	7	Sand	Qaushira
G36	558658	1374736	1934	32	3.3	1	1	7.13	45.4	82	9	9	Loamy Sand	Qaushira
G37	558893	1374719	1884	31	2.5	1	1	7.1	89	79	17	4	Sandy Loam	Qaushira
H38	558842	1396000	2413	30	1.3	0	1	8	57.3	52	15	33	Loam	Adishafi
H39	558953	1396178	2344	28	1.3	0	1	8.12	64.3	62	18	20	Sandy Loam	Adishafi
H40	559226	1395863	2417	30	1.3	0	1	7.3	71.7	73	11	16	Sandy Loam	Adishafi
H41	559316	1395962	2338	28	1.3	0	1	7.7	101.9	73	9	18	Sandy Loam	Adishafi
H42	559281	1396631	2220	26	1.3	0	0	7.63	47.7	62	18	20	Sandy Loam	Sifrahibey
H43	559460	1396590	2187	35	3.3	0	0	7.4	53.6	72	22	6	Sandy Clay Loam	Sifrahibey
H44	559867	1396720	2223	31	1.3	0	0	7.88	46.6	76	7	17	Loamy Sand	Sifrahibey
H45	559749	1396718	2174	30	1.3	0	0	7.96	36.7	90	5	5	Sand	Sifrahibey
H46	559232	1397534	2280	35	1.3	0	1	7.3	66.3	83	9	8	Loamy Sand	Sewhiendodo
H47	559358	1397508	2248	30	1.3	0	0	7.5	102.2	75	9	16	Sandy	Sewhiendo

													Loam	do
H48	559787	1397489	2280	29	1	1	1	8.1	57.1	82	7	11	Loamy Sand	Sewhiendo do
H49	559883	1397398	2247	31	0	0	0	6.7	66.6	75	17	8	Sandy Loam	Sewhiendo do
H50	560130	1395744	2177	36	1.3	0	1	7.1	155	78	9	13	Loamy Sand	Adidargo
H51	560202	1395824	2100	37	1.3	0	1	7.2	118.3	72	12	16	Sandy Loam	Adidargo
H52	559755	1395890	2245	42	3.3	1	1	7.9	42.3	80	7	13	Loamy Sand	Adidargo
H53	559839	1396039	2145	31	1.3	0	1	6.1	140	54	13	33	Loam	Adidargo
H54	560592	1396750	2376	31	1	1	1	7.3	45	76	12	12	Sandy Loam	Bandira
H55	560479	1396660	2332	35	1	0	1	8.1	129.1	64	20	16	Sandy Clay Loam	Bandira
H56	560137	1396990	2358	27	1	0	0	7.1	94.7	54	17	29	Sandy Loam	Bandira
H57	560701	1394755	2418	35	1.3	1	1	7.5	33.6	75	7	18	Loamy Sand	Mayamrak ut
H58	560804	1394884	2320	31	1.3	0	1	7	76.5	68	13	19	Sandy Loam	Mayamrak ut
H59	560870	1395005	2219	32	1.3	1	1	6.9	62.2	76	13	11	Sandy Loam	Mayamrak ut
H60	560943	1395111	2137	33	2.5	0	0	6.6	77.3	44	21	35	Loam	Mayamrak ut
H61	560184	1395217	2356	28	1.3	0	1	7.2	106	72	15	13	Sandy Loam	Chelaku
H62	560182	1395327	2287	32	0	0	0	7.2	52.4	71	17	12	Sandy Loam	Chelaku
H63	559631	1395503	2399	33	0	1	1	6.8 5	98	82	9	9	Loamy Sand	Chelaku

H64	559652	1395585	2355	32	1.3	1	1	6.96	74.1	74	9	17	Sandy Loam	Chelaku
H65	558667	1398171	2495	33	1.3	1	1	7.3	74.2	72	11	17	Sandy Loam	Degol
H66	558810	1398020	2444	37	1.3	0	1	7.5	62.5	62	18	20	Sandy Loam	Degol
H67	560025	1398608	2384	29	1	1	1	7.1	70.1	58	19	23	Sandy Loam	Degol
H68	560023	1398304	2360	31	1	0	1	7.3	44.6	76	9	15	Loamy Sand	Degol
H69	558049	1395006	2505	41	1.3	1	1	6.65	49.8	78	13	9	Sandy Loam	Adishila
H70	558097	1395010	2476	43	1.3	0	1	6.98	86.9	66	29	5	Sandy Clay Loam	Adishila
H71	558155	1395016	2455	44	1.3	0	0	7.3	40	74	9	17	Sandy Loam	Adishila
H72	558985	1395507	2502	43	1.3	1	1	7.23	40.6	73	11	16	Sandy Loam	Gerebhangero
H73	558983	1395452	2482	44	2	0	1	7.35	52	76	17	7	Sandy Loam	Gerebhangero
H74	558997	1395375	2450	43	0	0	1	7.2	44	50	9	41	Loam	Gerebhangero
H75	558859	1394416	2302	45	3.3	1	1	7.2	56.5	72	11	17	Sandy Loam	Girajale
H76	558865	1394496	2281	43	3.3	0	1	6.8	34.9	80	13	7	Sandy Loam	Girajale
H77	558874	1394579	2275	46	3.3	0	0	7.31	37.7	82	9	9	Loamy Sand	Girajale

Appendix 5 Frequency distribution of plant species in the edge, intermediate and interior habitats of the forest

No	Species	Frequency (%) at edge	Frequency (%) at intermediate	Frequency (%) at inner	Relative frequency (%)
1	*Juniperus procera*	94.12	84.85	81.25	86.74
2	*Hypoestes forskaolii*	88.24	81.82	87.5	85.85
3	*Carissa spinarum*	73.53	81.82	87.5	80.95
4	*Olea europaea* ssp. *cuspidata*	70.59	81.82	81.25	77.89
5	*Nuxia congesta*	61.76	63.64	68.75	64.72
6	*Maytenus undata*	70.59	72.73	50	64.44
7	*Andropogon abyssinicus*	58.82	54.55	68.75	60.71
8	*Acacia abyssinica*	58.82	51.52	68.75	59.70
9	*Cassipourea malosana*	55.88	57.58	62.5	58.65
10	*Rhus glutinosa*	61.76	60.61	50	57.46
11	*Dodonaea angustifolia*	61.76	45.45	62.5	56.57
12	*Cynodon dactylon*	55.88	57.58	43.75	52.40
13	*Olinia rochetiana*	50.00	42.42	50	47.47
14	*Dichanthium foveolatus*	50.00	42.42	43.75	45.39
15	*Cadia purpurea*	44.12	51.52	37.5	44.38
16	*Myrsine africana*	41.18	36.36	43.75	40.43
17	*Psydrax schimperiana*	26.47	39.39	43.75	36.54
18	*Dovyalis abyssinica*	44.12	33.33	31.25	36.23
19	*Bersama abyssinica*	23.53	39.39	43.75	35.56
20	*Osyris quadripartita*	35.29	33.33	37.5	35.38
21	*Cymbopogon pospischilii*	32.35	27.27	37.5	32.38
22	*Pittosporum viridiflorum*	20.59	36.36	37.5	31.48
23	*Themeda triandra*	29.41	30.30	31.25	30.32
24	*Calpurnia aurea*	23.53	33.33	31.25	29.37
25	*Dovyalis verrucosa*	29.41	24.24	31.25	28.30
26	*Pterolobium stellatum*	23.53	21.21	37.5	27.41
27	*Clutia abyssinica*	29.41	21.21	31.25	27.29
28	*Rhoicissus tridentata*	20.59	27.27	31.25	26.37
29	*Teclea simplicifolia*	26.47	21.21	31.25	26.31
30	*Tagetes minuta*	38.24	27.27	12.5	26.00
31	*Rhus natalensis*	17.65	27.27	31.25	25.39
32	*Rosa abyssinica*	23.53	15.15	31.25	23.31
33	*Asparagus racemosus*	29.41	21.21	18.75	23.12
34	*Acacia tortilis*	17.65	24.24	25	22.30
35	*Cynanchum abyssinicum*	17.65	24.24	25	22.30
36	*Cupressus lusitanica*	20.59	21.21	25	22.27

37	*Clematis hirsuta*	20.59	18.18	18.75	19.17
38	*Acacia etbaica*	14.71	15.15	25	18.29
39	*Astragalus atropilosulus*	14.71	12.12	25	17.28
40	*Euclea schimperi*	20.59	12.12	18.75	17.15
41	*Erica arborea*	23.53	9.09	18.75	17.12
42	*Verbascum sinaiticum*	11.76	12.12	25	16.30
43	*Aloe debrana*	5.88	24.24	18.75	16.29
44	*Ocimum forskolei*	17.65	18.18	12.5	16.11
45	*Dichrostachys cinerea*	23.53	12.12	12.5	16.05
46	*Eucalyptus globulus*	11.76	15.15	18.75	15.22
47	*Discopodium penninervium*	17.65	15.15	12.5	15.10
48	*Celtis africana*	5.88	18.18	18.75	14.27
49	*Ekebergia capensis*	14.71	9.09	18.75	14.18
50	*Sageretia thea*	11.76	18.18	12.5	14.15
51	*Senecio hadiensis*	14.71	15.15	12.5	14.12
52	*Vermifrux abyssinica*	5.88	15.15	18.75	13.26
53	*Debregeasia bicolar*	5.88	12.12	18.75	12.25
54	*Vigna membranacea*	8.82	15.15	12.5	12.16
55	*Dombeya torrida*	14.71	9.09	12.5	12.10
56	*Rhus retinorrhoea*	14.71	9.09	12.5	12.10
57	*Ficus palmata*	5.88	9.09	18.75	11.24
58	*Balanites aegyptiaca*	5.88	15.15	12.5	11.18
59	*Vernonia bipontini*	11.76	6.06	12.5	10.11
60	*Vernonia amygdalina*	11.76	6.06	12.5	10.11
61	*Achyranthes aspera*	11.76	12.12	6.25	10.05
62	*Clematis simensis*	2.94	12.12	12.5	9.19
63	*Pelargonium multibracteatum*	5.88	9.09	12.5	9.16
64	*Cynoglossum lanceolatum*	5.88	9.09	12.5	9.16
65	*Silene macrosolen*	5.88	9.09	12.5	9.16
66	*Solanum marginatum*	5.88	9.09	12.5	9.16
67	*Jasminum floribundum*	5.88	9.09	12.5	9.16
68	*Vernonia rueppellii*	8.82	6.06	12.5	9.13
69	*Solanum incanum*	11.76	9.09	6.25	9.04
70	*Solanum campylacanthum*	14.71	6.06	6.25	9.01
71	*Eucalyptus camaldulensis*	2.94	9.09	12.5	8.18
72	*Rhynchosia malacophylla*	2.94	9.09	12.5	8.18
73	*Hagenia abyssinica*	5.88	6.06	12.5	8.15
74	*Equisetum ramosissimum*	5.88	6.06	12.5	8.15
75	*Solanum nigrum*	5.88	6.06	12.5	8.15
76	*Hypericum revolutum*	5.88	12.12	6.25	8.08

77	*Barleria orbicularis*	5.88	12.12	6.25	8.08
78	*Barleria eranthemoides*	5.88	12.12	6.25	8.08
79	*Arthraxon prionodes*	5.88	12.12	6.25	8.08
80	*Datura stramonium*	5.88	12.12	6.25	8.08
81	*Grewia mollis*	8.82	9.09	6.25	8.05
82	*Verbesina encelioides*	2.94	6.06	12.5	7.17
83	*Helinus mystacinus*	5.88	9.09	6.25	7.07
84	*Cordia africana*	5.88	9.09	6.25	7.07
85	*Podocarpus falcatus*	5.88	9.09	6.25	7.07
86	*Clerodendrum myricoides*	8.82	6.06	6.25	7.04
87	*Acacia seyal*	8.82	6.06	6.25	7.04
88	*Crassocephalum rubens*	2.94	3.03	12.5	6.16
89	*Nepeta azurea*	2.94	3.03	12.5	6.16
90	*Myrica salicifolia*	5.88	6.06	6.25	6.06
91	*Allophylus abyssinicus*	5.88	6.06	6.25	6.06
92	*Ficus sur*	5.88	6.06	6.25	6.06
93	*Commicarpus peduncul*	5.88	6.06	6.25	6.06
94	*Cyanotis barbata*	5.88	6.06	6.25	6.06
95	*Maesa lanceolata*	8.82	3.03	6.25	6.03
96	*Echinops macrochaetus*	8.82	3.03	6.25	6.03
97	*Azadirachta indica*	2.94	6.06	6.25	5.08
98	*Grewia tembensis*	5.88	3.03	6.25	5.05
99	*Euphorbia schimperiana*	5.88	3.03	6.25	5.05
100	*Leonotis ocymifolia*	5.88	3.03	6.25	5.05
101	*Eragrostis tenuifolia*	5.88	3.03	6.25	5.05
102	*Adiantum incisum*	5.88	3.03	6.25	5.05
103	*Poa leptoclade*	5.88	3.03	6.25	5.05
104	*Laggera crispata*	5.88	3.03	6.25	5.05
105	*Sarcostemma viminale*	2.94	3.03	6.25	4.07
106	*Colutea abyssinica*	2.94	3.03	6.25	4.07
107	*Epilobium hirsutum*	2.94	3.03	6.25	4.07
108	*Crassocephalum macropappum*	2.94	3.03	6.25	4.07
109	*Polygala sphenoptera*	2.94	3.03	6.25	4.07
110	*Bidens pilosa*	2.94	3.03	6.25	4.07
111	*Cissus quadrangularis*	2.94	3.03	6.25	4.07
112	*Geranium arabicum*	2.94	3.03	6.25	4.07
113	*Cynanchum altiscandeus*	2.94	3.03	6.25	4.07
114	*Sanicula elata*	8.82	9.09	0	5.97
115	*Euphorbia monacantha*	2.94	12.12	0	5.02
116	*Solanum schimperianum*	5.88	9.09	0	4.99

117	*Celsia scrophularefolia*	5.88	9.09	0	4.99
118	*Kalanchoe marmorata*	5.88	9.09	0	4.99
119	*Portulaca quadrifida*	5.88	9.09	0	4.99
120	*Acokanthera schimperi*	2.94	9.09	0	4.01
121	*Pinus patula*	5.88	6.06	0	3.98
122	*Gomphocarpus fruticosa*	5.88	6.06	0	3.98
123	*Lippia adoensis*	5.88	6.06	0	3.98
124	*Scabiosa columbaria*	5.88	6.06	0	3.98
125	*Solanum adoense*	5.88	6.06	0	3.98
126	*Acacia asak*	8.82	3.03	0	3.95
127	*Pavetta oliveriana*	8.82	3.03	0	3.95
128	*Rubus steudneri*	2.94	6.06	0	3.00
129	*Acacia mearnsii*	2.94	6.06	0	3.00
130	*Gomphocarpus purpura*	2.94	6.06	0	3.00
131	*Spermacoce sphaeros*	2.94	6.06	0	3.00
132	*Nicandra physaloides*	2.94	6.06	0	3.00
133	*Oncocalyx kelleri*	2.94	6.06	0	3.00
134	*Toddalia asiatica*	5.88	3.03	0	2.97
135	*Sparmannia ricinocarpa*	5.88	3.03	0	2.97
136	*Monechma debile*	5.88	3.03	0	2.97
137	*Indigofera dendroides*	5.88	3.03	0	2.97
138	*Leucas abyssinica*	5.88	3.03	0	2.97
139	*Selaginella abyssinica*	5.88	3.03	0	2.97
140	*Phagnalon abyssinicum*	5.88	3.03	0	2.97
141	*Tragia cinerea*	5.88	3.03	0	2.97
142	*Meriandra dianthera*	5.88	3.03	0	2.97
143	*Euphorbia ampliphylla*	5.88	3.03	0	2.97
144	*Senna sinqueana*	2.94	3.03	0	1.99
145	*Laggera tomentosa*	2.94	3.03	0	1.99
146	*Meriandra bengalensis*	2.94	3.03	0	1.99
147	*Pennisetum polystachion*	2.94	3.03	0	1.99
148	*Snowdenia polystachya*	2.94	3.03	0	1.99
149	*Ocimum urticifolium*	2.94	3.03	0	1.99
150	*Oxalis corniculata*	2.94	3.03	0	1.99
151	*Parthenium hysterophorus*	2.94	3.03	0	1.99
152	*Portulaca oleracea*	2.94	3.03	0	1.99
153	*Mentha longifolia*	2.94	3.03	0	1.99
154	*Galinsoga quadriradiata*	2.94	3.03	0	1.99
155	*Silene burchellii*	2.94	3.03	0	1.99
156	*Celosia anthelminthica*	2.94	3.03	0	1.99
157	*Sporobolus pyramidalis*	2.94	3.03	0	1.99

158	*Ocimum suave*	2.94	3.03	0	1.99
159	*Hyparrhenia hirta*	2.94	3.03	0	1.99
160	*Veronica anagallis-aqua*	2.94	3.03	0	1.99
161	*Pennisetum setaceum*	2.94	3.03	0	1.99
162	*Cadaba farinosa*	2.94	3.03	0	1.99
163	*Ficus sycomorus*	2.94	3.03	0	1.99
164	*Buddleja polystachya*	0.00	9.09	18.75	9.28
165	*Ziziphus spina-christi*	0.00	6.06	12.5	6.19
166	*Cyperus fischerianus*	0.00	6.06	12.5	6.19
167	*Phytolacca dodecandra*	0.00	9.09	6.25	5.11
168	*Haplocarpha schimperi*	0.00	9.09	6.25	5.11
169	*Polypogon viridis*	0.00	9.09	6.25	5.11
170	*Heteromorpha trifoliata*	0.00	6.06	6.25	4.10
171	*Maytenus arbutifolia*	0.00	6.06	6.25	4.10
172	*Berberis holstii*	0.00	6.06	6.25	4.10
173	*Opuntia ficus-indica*	0.00	6.06	6.25	4.10
174	*Malva verticillata*	0.00	6.06	6.25	4.10
175	*Rumex abyssinicus*	0.00	6.06	6.25	4.10
176	*Medicago polymorpha*	0.00	6.06	6.25	4.10
177	*Panicum subalbidum*	0.00	6.06	6.25	4.10
178	*Crotalaria pycnostachya*	0.00	6.06	6.25	4.10
179	*Hypericum quartinianum*	0.00	6.06	6.25	4.10
180	*Euphorbia hirta*	0.00	6.06	6.25	4.10
181	*Ziziphus mucronata*	0.00	3.03	6.25	3.09
182	*Becium grandiflorum*	0.00	3.03	6.25	3.09
183	*Solanum benderianum*	0.00	3.03	6.25	3.09
184	*Echium plantagineum*	0.00	3.03	6.25	3.09
185	*Satureja punctata*	0.00	3.03	6.25	3.09
186	*Crotalaria lachnophora*	0.00	3.03	6.25	3.09
187	*Hyperthelia dissoluta*	0.00	3.03	6.25	3.09
188	*Pappea capensis*	0.00	3.03	6.25	3.09
189	*Aerva lanata*	0.00	3.03	6.25	3.09
190	*Anarrhinum forskaohlii*	0.00	3.03	6.25	3.09
191	*Pelargonium alchemilloides*	0.00	3.03	6.25	3.09
192	*Euphorbia petitiana*	0.00	3.03	6.25	3.09
193	*Crassocephalum crepidio*	0.00	3.03	6.25	3.09
194	*Trachydium abyssinicum*	0.00	3.03	6.25	3.09
195	*Conyza hypoleuca*	5.88	0.00	6.25	4.04
196	*Senecio lyratus*	5.88	0.00	6.25	4.04
197	*Kalanchoe lanceolata*	5.88	0.00	6.25	4.04
198	*Cardiospermum corindum*	5.88	0.00	6.25	4.04

199	*Ehretia cymosa*	2.94	0.00	6.25	3.06
200	*Pseudognaphalium oligandrum*	2.94	0.00	6.25	3.06
201	*Psidium quajava*	2.94	0.00	6.25	3.06
202	*Pennisetum unisetum*	2.94	0.00	6.25	3.06
203	*Justicia caerulea*	2.94	0.00	6.25	3.06
204	*Arundo donax*	2.94	0.00	6.25	3.06
205	*Combretum collinum*	2.94	0.00	6.25	3.06
206	*Ocimum lamiifolium*	8.82	0.00	0	2.94
207	*Carduus nyassanus*	8.82	0.00	0	2.94
208	*Otostegia fruticosa*	5.88	0.00	0	1.96
209	*Grewia ferruginea*	5.88	0.00	0	1.96
210	*Canthium pseudosetiflorum*	5.88	0.00	0	1.96
211	*Barleria ventricosa*	5.88	0.00	0	1.96
212	*Otostegia integrrifolia*	5.88	0.00	0	1.96
213	*Orobanche ramosa*	5.88	0.00	0	1.96
214	*Anthemis tigreensis*	5.88	0.00	0	1.96
215	*Lichen species*	5.88	0.00	0	1.96
216	*Chrysopogon aucheri*	5.88	0.00	0	1.96
217	*Snowdenia petitiana*	5.88	0.00	0	1.96
218	*Anthospermum herbacem*	5.88	0.00	0	1.96
219	*Cyathula uncinulata*	5.88	0.00	0	1.96
220	*Rumex nervosus*	2.94	0.00	0	0.98
221	*Maerua angolensis*	2.94	0.00	0	0.98
222	*Jasminum grandiflorum*	2.94	0.00	0	0.98
223	*Laggera elatior*	2.94	0.00	0	0.98
224	*Psiadia punctulata*	2.94	0.00	0	0.98
225	*Guizotia scabra*	2.94	0.00	0	0.98
226	*Cotula abyssinica*	2.94	0.00	0	0.98
227	*Helichrysum nudifolium*	2.94	0.00	0	0.98
228	*Dicrocephala chrysanthemifolia*	2.94	0.00	0	0.98
229	*Caylusea abyssinica*	2.94	0.00	0	0.98
230	*Torilis arvensis*	2.94	0.00	0	0.98
231	*Trifolium ruepellianum*	2.94	0.00	0	0.98
232	*Panicum hochstetteri*	2.94	0.00	0	0.98
233	*Pennisetum macrourum*	2.94	0.00	0	0.98
234	*Pennisetum thunbergii*	2.94	0.00	0	0.98
235	*Withania somnifera*	2.94	0.00	0	0.98
236	*Triumfetta pilosa*	2.94	0.00	0	0.98
237	*Erythrococca trichogyne*	2.94	0.00	0	0.98
238	*Dyschoriste radicans*	2.94	0.00	0	0.98

239	*Gerbera piloselloides*	2.94	0.00	0	0.98
240	*Chenopodium murale*	2.94	0.00	0	0.98
241	*Cyperus alternitolius*	2.94	0.00	0	0.98
242	*Verbascum pedunculosa*	2.94	0.00	0	0.98
243	*Bidens biternata*	2.94	0.00	0	0.98
244	*Schinus molle*	2.94	0.00	0	0.98
245	*Crepis rueppellii*	2.94	0.00	0	0.98
246	*Kniphofia pumila*	0.00	9.09	0	3.03
247	*Argemone mexicana*	0.00	9.09	0	3.03
248	*Flueggea virosa*	0.00	6.06	0	2.02
249	*Hebenstretia angolensis*	0.00	6.06	0	2.02
250	*Chloris gayana*	0.00	6.06	0	2.02
251	*Abutilon hirtum*	0.00	3.03	0	1.01
252	*Argyrolobium arabicum*	0.00	3.03	0	1.01
253	*Vernonia schimperi*	0.00	3.03	0	1.01
254	*Conyza aegyptiaca*	0.00	3.03	0	1.01
255	*Satureja biflora*	0.00	3.03	0	1.01
256	*Campanula edulis*	0.00	3.03	0	1.01
257	*Tephrosia uniflora*	0.00	3.03	0	1.01
258	*Ajuga integrifolia*	0.00	3.03	0	1.01
259	*Ceratostigma abyssinicum*	0.00	3.03	0	1.01
260	*Panicum hygrocharis*	0.00	3.03	0	1.01
261	*Lantana viburnoides*	0.00	3.03	0	1.01
262	*Panicum maximum*	0.00	3.03	0	1.01
263	*Hybanthus puberulus*	0.00	3.03	0	1.01
264	*Calotropis procera*	0.00	3.03	0	1.01
265	*Asplenium abyssinicum*	0.00	3.03	0	1.01
266	*Leucas martinicensis*	0.00	3.03	0	1.01
267	*Indigofera costata*	0.00	3.03	0	1.01
268	*Senna occidentalis*	0.00	3.03	0	1.01
269	*Urtica simensis*	0.00	3.03	0	1.01
270	*Lycopersicon esculentum*	0.00	3.03	0	1.01
271	*Hybanthus enneaspermus*	0.00	3.03	0	1.01
272	*Dicrocephala integrifolia*	0.00	3.03	0	1.01
273	*Conyza hochstetteri*	0.00	3.03	0	1.01
274	*Silybum marianum*	0.00	3.03	0	1.01
275	*Peperomia abyssinica*	0.00	3.03	0	1.01
276	*Abutilon longicuspe*	0.00	0.00	12.5	4.17
277	*Hibiscus panduriformis*	0.00	0.00	12.5	4.17
278	*Heliotropium cinerascens*	0.00	0.00	6.25	2.08
279	*Heracleum abyssinicum*	0.00	0.00	6.25	2.08

280	*Rhynchosia elegans*	0.00	0.00	6.25	2.08
281	*Thymus schimperi*	0.00	0.00	6.25	2.08
282	*Crassula schimperi*	0.00	0.00	6.25	2.08
283	*Crotalaria incana*	0.00	0.00	6.25	2.08
284	*Pennisetum sphacelatum*	0.00	0.00	6.25	2.08
285	*Convolvulus kilimandschari*	0.00	0.00	6.25	2.08
286	*Ocimum americanum*	0.00	0.00	6.25	2.08
287	*Cyphostemma cyphopetalum*	0.00	0.00	6.25	2.08
288	*Rumex nepalensis*	0.00	0.00	6.25	2.08

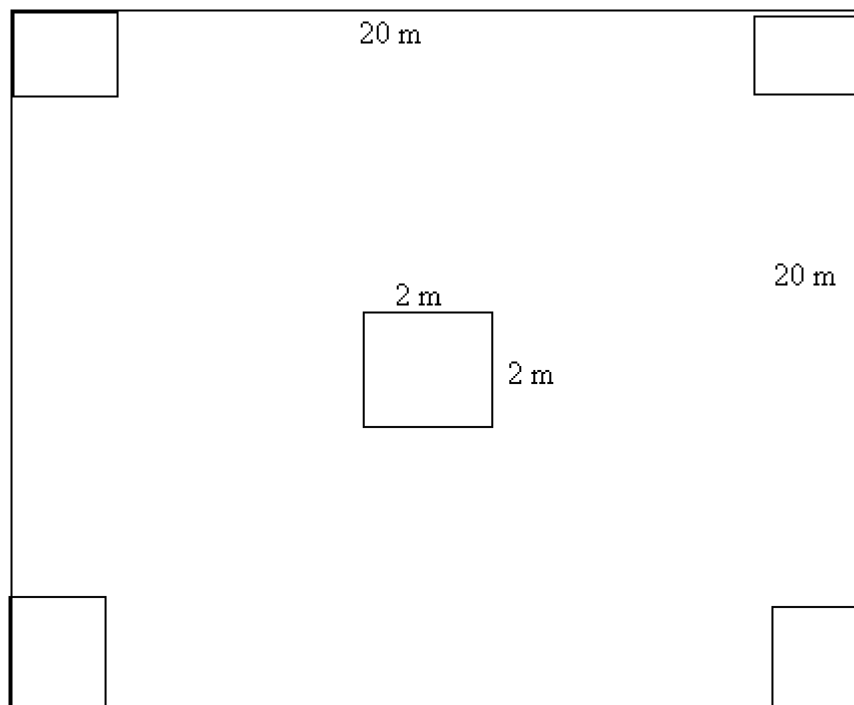
Appendix 6 Data sheet used for collection of floristic and environmental variables

Quadrat No __________ Date ____________
Locality-name___________Location __________________
Slope___________________Aspect________________

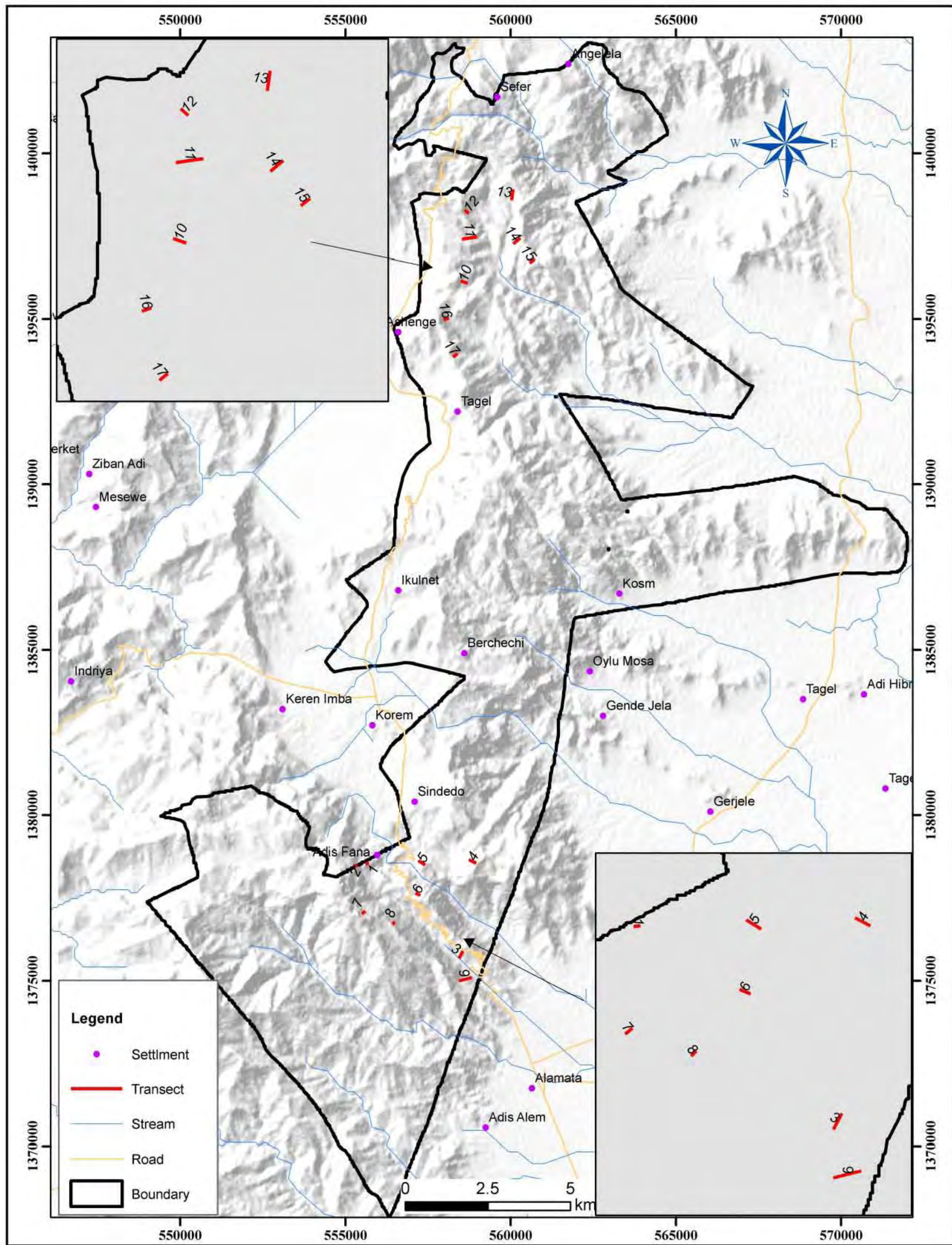
Lat/Lon/………….. ....Altitude___________

Species Code	Local name /Scientific name	Number/density	DBH	Height	Cover-abundance	Remark
Note:						

Appendix 7 Diagrammatic representation of the design of the study sample plot with subplots



Appendix 8 Location map of the study area showing sampling transects of edge effect study



Appendix 9 Photograph illustrating the forest overview

Appendix 10 Photograph illustrating dead trees (snags) and fallen boles (logs) in the forest

Appendix 11 Photograph illustrating some aspects of disturbance mainly at the edge of the forest

Appendix 12 Photograph illustrating soil testing in progress at the laboratory of AAU

DECLARATION

I, the undersigned declare that this dissertation is my original work and it or part of it has not been presented in any other University, college or institution for a degree or other purpose. All sources of materials used for the dissertation have been duly acknowledged.

Name: Leul Kidane Woldemichael

Signature: ______________________________________

Date of submission: ______________________________

We confirm that the work reported in this thesis was carried out by the candidate under our supervision.

	Signature	Date
1. Prof. Sileshi Nemomissa	________________________	______________________
2. Prof. Zerihun Woldu	__________________________	_____________________