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Floristic Composition, Structural Analysis and Land Use/Land Cover Change in Bore-Anferara-Wadera Forest, Southern Ethiopia

Mesfin Woldearegay

Addis Ababa University
Addis Ababa, Ethiopia
June 2017

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Floristic Composition, Structural Analysis and Land Use/Land Cover Change in Bore-Anferara-Wadera Forest, Southern Ethiopia

Mesfin Woldearegay Ahmed

A Dissertation Submitted to
The Department of Plant Biology and Biodiversity Management
Presented in Partial Fulfillment of the Requirements for the Degree of
Doctor of Philosophy (Biology: Botanical Sciences)

Addis Ababa University
Addis Ababa, Ethiopia
June 2017

ADDIS ABABA UNIVERSITY
GRADUATE PROGRAMMES

This is to certify that the Dissertation prepared by Mesfin Woldearegay Ahmed, entitled*:* *Floristic Composition, Structural Analysis*, *and Land Use/Land Cover Change in Bore-Anferara-Wadera Forest, Southern Ethiopia*, and submitted in partial fulfillment of the Requirements for the Degree of Doctor of Philosophy (Biology: Botanical Sciences) complies with the regulations of the University and meets the accepted standards with respect to originality and quality.

Signed by Research Supervisors:

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Abstract

Floristic Composition, Structural Analysis and Land use/Land Cover Change in Bore-Anferara-Wadera Forest, Southern Ethiopia

Mesfin Woldearegay Ahmed, Ph.D. Dissertation
Addis Ababa University, 2017

This study was conducted in Bore-Anferara-Wadera forest, southern Ethiopia, to investigate the floristic composition, vegetation structure, regeneration status and land use/land cover change. Vegetation data were collected from 112, 30 m x 30 m sample plots laid for trees at every 400 m distance along line transects and 5 m x 5 m and five 1 m x 1 m subplots for saplings and herbs, respectively. The regeneration status of woody species was assessed by employing total count of all seedlings within the main sample plot. Environmental variables such as altitude, slope, and exposure were measured in each sample plot. Soil samples were taken from two layers (0-25 and 25-50 cm) at five points in each sample plot and soil sample from these five points were mixed to form a composite sample. In each sample plot, woody species ≥ 3 m were counted and cover abundance values estimated as well as height and diameter at breast height were measured. Hierarchical cluster analysis was used to identify plant communities and synoptic values for identification of the dominant species for naming plant communities. Density, frequency, basal area and importance value index (IVI) of woody species were also computed. Shannon-Wiener diversity index was used to assess species richness and evenness. Sorensen's similarity coefficient was used to measure similarities among communities and between Bore-Anferara-Wadera and eight Afromontane forests in Ethiopia. Canonical correspondence analysis (CCA) was used to assess the relationship between plant community types and environmental variables. Moreover, three periods land sat images (1986 TM, 2000 ETM+ and 2014 OLI/TIRS) were acquired and analyzed by using remote sensing and GIS technologies to generate information on the temporal changes in land use and land cover types. A total of 136 vascular plant species belonging to 119 genera and 63 families were recorded. About 4.4% of the species were endemic to Ethiopia and 11.8 % of the species were new records for the Sidamo floristic region of the flora area. The overall Shannon-Wiener diversity and evenness values of Bore-Anferara-Wadera forest were 3.84 and 0.78, respectively. Size class distribution of woody species across different DBH and height classes indicated the relatively high proportion of individuals at lower classes, indicating impacts of past disturbance. Analysis of population structure and regeneration status of the forest revealed various patterns of population dynamics where some species were represented by few mature plants only suggesting that they are on the verge of local extinction and thus immediate conservation measures should be taken. Community classification using the free statistical software R version 3.1.1 resulted in four, namely Acanthus eminens - Dracaena afromontana, Syzygium guineense subsp. afromontanum - Ocotea kenyensis, Pouteria adolfi-friederici - Psychotria orophila and Scolopia theifolia - Teclea nobilis community types. Canonical correspondence analysis (CCA) result showed that altitude and slope were among the main environmental variables in determining patterns of species distribution and plant community formation. The results of land sat image *analysis revealed that agricultural land and built up area are expanding rapidly at the expense of other land use and land cover types. Forest and shrub land areas have declined drastically over the last 28 years. Population pressure, deforestation, land tenure system, and forest fire were the main driving forces responsible for the change in land use and land cover types in the study area. Therefore, a joint management and conservation measures should be taken by the government, local people and other stakeholders in order to reduce and/or stop the fast rate of vegetation cover declining and sustainable utilization of the forest resources in the study area.*

Keywords: Anferara, Biodiversity, Conservation, Land use/land cover, Plant community

DEDICATION

I would like to dedicate this dissertation to my wife Mekdes Gerawork and my children Samuel and Hemen Mesfin for their unreserved support, encouragement and patience over the years, and to my parents who did not enjoy formal education but strongly committed to teach their children hoping that their future will be better off.

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LIST OF ACRONYMS

AAU	Addis Ababa University
BA	Basal Area
BLI	Bird Life International
CBD	Convention on Biological Diversity
CCA	Canonical Correspondence Analysis
CEC	Cation Exchange Capacity
CI	Conservation International
CSA	Central Statistical Authority
DBH	Diameter at Breast Height
DCA	Detrended Correspondence Analysis
EBI	Ethiopian Biodiversity Institute
EFAP	Ethiopian Forestry Action Program
EMA	Ethiopian Mapping Authority
EMSA	Ethiopian Meteorological Services Agency
ETH	National Herbarium
EWNHS	Ethiopian Wildlife and Natural History Society
FAO	Food and Agricultural Organization of the United Nations
FDREPCC	Federal Democratic Republic of Ethiopia Census Commission
FEE	Flora of Ethiopia and Eritrea
GPS	Global Positioning System
IBC	Institute of Biodiversity Conservation
IPCC	Intergovernmental Panel on Climate Change

ISRIC	International Soil Reference and Information Center
IUFRO	International Union for Forestry Research Organization
IVI	Importance Value Index
LULC	Land Use and Land Cover
MEA	Millennium Ecosystem Assessment
NMSA	National Meteorological Services Agency
NRGOBFED	National Regional Government of Oromia Bureau of Finance and Economic Development
NRSOBFED	National Regional State of Oromia Bureau of Finance and Economic Development
OC	Organic Carbon
SCBD	Secretariat of the Convention on Biological Diversity
SNNP	Southern Nations, Nationalities, and Peoples
WCMC	World Conservation Monitoring Center

CHAPTER ONE

1. INTRODUCTION

1.1 Background

Biological diversity (biodiversity) is defined as “the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems” (CBD, 1992). It covers the variety of life in different scales. Thus, it is not only the variety of plant and animal species but also the variety of genes within those species and the variety of ecosystems in which the species inhabit. Biodiversity is important for the functioning of the ecosystem on which humans depend for a variety of goods and services such as food, fiber, medicine, water, recreation and protection from natural disasters. However, the current trends of biodiversity loss are thought to be severely affecting these goods and services, such that continued loss may result in a point where the capacity of ecosystems to provide these vital services will be greatly reduced (Mace *et al.*, 2012).

Anthropogenic interferences in the natural environment are the factors most responsible for the loss of biodiversity (MEA, 2005; Barnosky *et al.*, 2011). The major human activities driving biodiversity loss are the degradation, fragmentation, and destruction of habitats, overexploitation of biological resources, pollution, invasive species and unsustainable practices in agriculture, aquaculture and forestry (WCMC, 1992; MEA, 2005; Butchart *et al.*, 2010; SCBD, 2014). In many tropical areas of the world, for example, deforestation is still increasing, and habitats of all types, including forests, grasslands, wetlands and river systems, continue to be fragmented and degraded (SCBD, 2014). According to Hooper *et al.* (2012), loss of biodiversity could rival the problems of carbon dioxide increase as one of the major drivers of ecosystem change in the 21st century. With the loss of species, we lose the wild relatives of domesticated crop species and the genes we use to improve agricultural resilience and the production of a wide range of ecosystem services that support humans and all life on Earth (McNeely *et al.*, 2009; SCBD, 2014).

Ethiopia, which is found in the northeastern highlands of tropical Africa, has unique ecological settings with considerably varied edaphic (soil), climatic and biological resources. Ethiopia also shares more than 50% of the Afromontane regions, land area above 1500 m, of Africa (Yalden, 1983; Tamrat Bekele, 1994) and has endowed with diversified fauna and flora that make it an important regional center of biological diversity and endemism (Sayer *et al.*, 1992; Zerihun Woldu, 1999). Many of the genetic resources of the country are still unexplored. However, these large biodiversity resources are under continuous threats of destruction mainly due to habitat loss and fragmentation, unsustainable utilization of biological resources, invasive species, climate change and pollution (Mulugeta Limeneh & Demel Teketay, 2004; EBI, 2014).

The rate of deforestation accelerated towards the beginning of twenty century and about 16% of the land area was estimated to have been covered by high forests in the early

1950s which declined to 3.6% in the early 1980s and further down to 2.7% in 1989 (EFAP, 1994). A recent analysis on the rate of deforestation shows that the country's forest resource is rapidly declining at the rate of 141,000 ha per year (FAO, 2010). This extensive amount of habitat loss or destruction may have resulted in a rapid dwindling of the genetic resources of the country (EBI, 2014). According to Ensermu Kelbessa *et al.* (1992), 120 endemic plant species of Ethiopia has been reported to be threatened by forest destruction. This underscores the need for sustainable utilization and management of the remnant natural forests to preserve the rich biodiversity resources of the country from complete disappearance (Mulugeta Limeneh & Demel Teketay, 2004).

Han *et al*. (2011) have shown that understanding diversity, distribution and extent of use of plants in a country is a basis for designing and implementing a sound resource management and utilization system in a sustainable manner. Although different floristic inventories were carried out in different parts of Ethiopia such as those by Sebsebe Demissew (1988), Zerihun Woldu *et al*. (1989), Tamrat Bekele (1993), Demel Teketay and Tamrat Bekele (1995), Kumlachew Yeshitela and Tamrat Bekele (2002), Feyera Senbeta and Demel Teketay (2003), Abate Ayalew *et al*. (2006), Ermias Lulekal *et al*. (2008), Haile Yineger *et al*. (2008), Abreham Assefa *et al*. (2013), Birhanu Kebede *et al*. (2014), etc., the country's overall floristic composition study is yet to be complete. The Bore-Anferara-Wadera Forest in southern Ethiopia has never been scientifically studied for its plant diversity, structural analysis, regeneration status and land use and land cover change except one ecological study, which described the variation in vegetation and their relationship with some environmental variables, by Hailu Sharew in 1982 and thus it was targeted for investigation in this research.

Recently, biodiversity conservation at the genetic, species and ecosystem level has become a major environmental and natural resource management issue of local, national and global importance (Lovett *et al.,* 2000). The Convention on biological Diversity (CBD) considers protected areas as cornerstones of biodiversity conservation (CBD, 2009) and hence well-governed and effectively managed protected areas are proven method for conserving both habitats and populations of species and for delivering important ecosystem services (Sharrock *et al.,* 2014). In Ethiopia, protected areas cover 14% of the total area of the country and contribute a significant role in conservation, ecotourism, recreation and employment (EBI, 2014). Protected areas include biodiversity hotspots.

Biodiversity hotspots are biogeographic regions that obtain the highest priority for conservation activities (Myers *et al.,* 2000) due to the highest vulnerability of habitats and high irreplaceability of species found within these regions. That is, these areas and the species present within them are both under high levels of threat and of significant global value based on their uniqueness (CI, 2014). Currently, 34 biodiversity hotspots have been identified throughout the world. Most of these biodiversity hotspots are found in the tropical forests since they are rich in their species richness and concentration of endemic species (Mittermeier *et al.*, 1998; Brooks *et al.,* 2006). They contain around 50% of endemic plant species and 42% of all terrestrial vertebrates of the world (CI, 2014).

These hotspots retain 14.9% of their total area as natural intact vegetation and most of them have much less natural intact vegetation than previously estimated (Solan *et al.,* 2014), which underscore the need to focus on conservation of these biologically critical regions.

Most areas of Ethiopia fall within two of these biodiversity hotspot regions, namely, Horn of Africa and The Eastern Afromontane. The present study area is part of Key Biodiversity Area (KBA) in the Eastern Afromontane hotspot region which contains vulnerable species such as *Leptopelis ragazzii* (amphibian), *Serinus xantholaemus and Tauraco ruspolii* (birds) and *Ocotea kenyensis* (tree) in the highlands of southern Ethiopia (BLI, 2012). Despite high conservation priority area, basic information necessary to develop and implement appropriate conservation and management strategies is lacking. Therefore, the outcomes of this study fill this gap by providing concrete information useful to develop an efficient management plan for biodiversity conservation and sustainable utilization of the natural resources of the study area.

1.2 Research questions and objectives

1.2.1 Research questions

The following were the major research questions identified for deep investigation in this study:

- What is the floristic composition and diversity of the vegetation in Bore-Anferara-Wadera forest?

- What are the plant community types and environmental variables determining the patterns of species distribution and community formation?
- What is the natural regeneration status of woody species in Bore-Anferara-Wadera forest?
- What is the vegetation structure of woody species in Bore-Anferara-wadera forest?
- How is the rate and extent of land use/land cover change in Bore-Anferara-Wadera forest between 1986 and 2014?
- What are the major factors responsible for land use/land cover change in the study area?

1.2.2 Research objectives

1.2.2.1 General objective

The general objective of this research was to investigate the floristic composition, vegetation structure, regeneration status and land use and land cover change in Bore-Anferara-Wadera forest.

1.2.2.2 Specific objectives

The specific objectives of this research were to:

- Identify plant species composition and vegetation structure of Bore-Anferara-Wadera forest;
- Assess the composition, structure, and density of regenerating woody species in Bore-Anferara-Wadera forest;

- Identify the plant community types and analyze the relationship between environmental factors and patterns of community formation;
- Assess and compare plant species diversity among plant community types in Bore-Anferara-Wadera forest;
- Generate tangible information on land use and land cover change in the study area;
- Identify the major factors responsible for land use and land cover change in the study area; and
- Suggest appropriate methods of biodiversity conservation and management strategies for sustainable utilization of the natural resources in the study area

CHAPTER TWO

2. LITERATURE REVIEW

2.1 Plant diversity

Biodiversity is a term used to describe the total number, variety and variability of living organisms as well as the diversity of the ecosystem they are living in (Krebs, 1999; CBD, 2009). It plays a key role in ecosystem functioning and has been widely used as an indicator of ecosystem health (FAO, 2005). Even though the concept of biodiversity is very broad, diversity of species remains the most frequently and widely used measure of biodiversity (Heywood *et al.,* 1995). Plant diversity is one component of biodiversity that refers to the variety (in both genetic and species level) of plants that exist in all terrestrial and aquatic regions of the earth with the exception of ice-covered regions. Plant species diversity represents millions of years of evolution and provides an important visible expression of biodiversity. Healthy ecosystems based on plant diversity provide the conditions and processes that sustain life and are essential to the well-being and livelihoods of mankind (CBD, 2009; Sharrock *et al.,* 2014). Specific ecosystem services provided by plants include the production of oxygen, assimilation, and sequestration of atmospheric carbon dioxide, soil conservation, watershed protection, slowing run-off rate of precipitation and promoting water infiltration and purification. Reduction in plant diversity, however, affects these ecosystem services. Ecosystem sustainability depends on a large extent on the buffering capacity provided by having a rich and healthy plant diversity of genes, species, and habitats. Losing plant diversity is like losing the life-support systems of the earth on which human beings and other species depend upon.

Despite the importance of plants, the total number of species in existence is not yet known accurately. Plant scientists estimated around 400,000 species with an average addition of 2,000 new species every year (Sharrock *et al.,* 2014). These plant species are unevenly distributed across the globe, with the majority of plants being found in the tropics. Many of them are restricted in range with a significant number being endemic to single country. Islands contain high numbers of endemic plants and are home to 35% of the world's plants. However, it is predicted that as many as two-thirds of the world's plant species are in danger of extinction during the 21st century. Extinction and declines in plant diversity are due to a range of factors including human population growth, high rates of habitat loss and fragmentation, deforestation, over-exploitation for timber, the impact of invasive alien species, pollution and climate change (Sharrock *et al.,* 2014).

Based on species richness, endemicity, and level of threats, 35 biodiversity hotspots have been identified for biodiversity conservation. Of these globally identified biodiversity hotspots, eight of them are found in Africa and Ethiopia shared two biodiversity hotspots (The Eastern Afromontane and Horn of Africa hotspots) (CI, 2014). Bore-Anferara-Wadera forest is a key biodiversity area in the Eastern Afromontane hotspot region, which contains vulnerable plant and animal species.

2.1 The vegetation types of Ethiopia

Ethiopia is found in the Horn of Africa, stretching 3^0 to 15^0 N latitude and 33^0 to 48^0 E longitude, with a total area of 1, 127, 127 km^2 (EBI, 2014). It is a country of great geographical diversity with rugged mountains and plateaus, deep gorges, river valleys and plains. The altitude ranges from the highest peak at Mount Ras Dashen, 4,620 m above sea level, down to Danakil depression, -116 m below sea level (EFAP, 1994; Demel Teketay, 2001; EBI, 2014). The Great African Rift valley divides the western and south-eastern highlands that host most of the Afromontane vegetation of the country (Friis *et al.,* 2011). Extensive highland plateaus, with an altitude of over 2,500 m above sea level, covers 40% of the country (Zerihun Woldu, 1999). Topographic and altitudinal variation in the Ethiopian landscapes has resulted in the existence of varied vegetation types, floristic diversity, and soil (edaphic) factors.

The Ethiopian flora is very heterogeneous and has a rich endemic element. It is estimated to contain around 6,000 species of higher plants, of which about 10% are endemic (Ensermu Kelbessa & Sebsebe Demissew, 2014). Many researchers have studied and described the floristic diversity and vegetation types of Ethiopia. Earlier descriptions of vegetation types include the works of Pichi-Sermolli (1957), von Breitenbach (1963), White (1983), Friis (1986; 1992), Sebsebe Demissew (1996), and Friis & Sebsebe Demissew (2001). All these attempts have had considerable contributions towards the understanding of the vegetation of the country. The oldest and most significant overall vegetation survey of Ethiopia made by Pichi-Sermolli (1957), recognized 24 vegetation types (in East Africa, 22 of which occur in Ethiopia), laid the basis for systematic studies of the vegetation and environmental factors in Ethiopia (Zerihun Woldu, 1999). Later on, White (1983) broadly divided the vegetation of East Africa into three major types based on ‘centers of endemism’. They are Sudanian Regional Centre of Endemism, Somalia-Massai Regional Centre of Endemism, and Afromontane Archipelago-like Regional Centre of Endemism. Recent studies made by Friis *et al.* (1982), Hailu Sharew (1982), Sebsebe Demissew (1988), Zerihun Woldu *et al*. (1989), Zerihum Woldu and Mesfin Tadesse (1990), Tamrat Bekele (1993), Kumilachew Yeshitila and Tamrat Bekele (2002), Teshome Soromessa *et al.* (2004), Tesfaye Awas (2007), Haile Yineger *et al.* (2008), Fekadu Gurmessa *et al.* (2012), Abrham Assefa *et al*. (2013), Mamo Kebede *et al.* (2013) represent some of the vegetation surveys made in different parts of the country in order to describe community types and their relationship with some anthropogenic and environmental factors.

Previous attempts of classification of the Ethiopian vegetation have been unsatisfactory due to the complexity of the vegetation (Zerihun Woldu, 1999). The complexity arises from the great variations in altitude implying equally great spatial differences in moisture regimes as well as temperatures within very short horizontal distances. Recently, Friis *et al.* (2011) described twelve major potential vegetation types of Ethiopia. These are: (1) Desert and semi-desert scrubland; (2) *Acacia-Commiphora* woodland and bushland; (3) Wooded grassland of the western Gambella Region; (4) *Combretum–Terminalia* woodland and wooded grassland; (5) Dry evergreen Afromontane forest and grassland complex; (6) Moist evergreen Afromontane forest; (7) Transitional rainforest; (8) Ericaceous belt; (9) Afroalpine vegetation; (10) Riverine vegetation; (11) Freshwater lakes (including lake shores, marshes, swamps and floodplain vegetation) and (12) Salt water lakes (including lake shores, salt marshes and pan vegetation). Of these vegetation types, the moist evergreen Afromontane forest is the vegetation type of the present study area. This type of vegetation, in most cases, is characterized by one or more closed strata of evergreen trees that may reach a height of 30 to 40 m. Sometimes only the lower stratum remains owing to the removal of the canopy. Large areas covered by this vegetation type have now changed to farmland, secondary montane grassland, secondary montane woodland and secondary evergreen bushland (Hedberg *et al.,* 2009).

2.2 Threats to plant diversity in Ethiopia

Owing to its geographical and climatic diversity, Ethiopia is one of the regional centers of biological diversity and genetic resources in the world (ZerihunWoldu, 1999; IBC, 2009). However, the rapid population growth and the demand for natural resources have put great pressure on the biodiversity of the country. The present trends of population growth and deforestation need to be abated, using through family planning and appropriate land use practice respectively to reverse the deterioration of natural resources and to prevent the decline or loss of biodiversity (Teshome Soromessa *et al.,* 2004).

The direct causes of biodiversity loss in Ethiopia are deforestation and land degradation, overexploitation, overgrazing, invasive species, pollution and climate change, while the proximate causes are poverty, population growth, lack of alternative livelihoods, inadequate policy support, and inappropriate investment (IBC, 2009). Forest degradation in Sub-Saharan Africa has widely taken place because people gain immediate economic benefits from the forest-related economic activities (Mogaka *et al.,* 2001). Similarly, clearing of forests for crop cultivation and fuelwood, to satisfy the food and energy requirements of the increasing population, causes accelerated deforestation and habitat fragmentation, which are now major environmental concerns in Ethiopia (Tadesse Woldemariam and Demel Teketay, 2001).

There is no reliable figure on the extent of past forest cover in Ethiopia. Owing to lack of a single definition for the forest, different authors are using different definitions and hence resulted in large variations in estimating the past forest cover of the country (Eshetu Yirdaw, 2002). However, the numerous isolated mature forest trees and patches of forest or woodland of approximately the same species composition as in the remaining areas with closed forest indicated that most of the highland areas were once covered by Afromontane forests (Friis, 1986; Eshetu Yirdaw, 2002).

According to FAO (2001), a forest is defined as “land with a tree crown cover (or equivalent stocking level) of more than 10% and an area of more than 0.5 hectares; the trees should be able to reach a minimum height of 5 m at maturity *in situ*”. Following this definition, the vegetations of Ethiopia that may qualify as ‘forests’ are natural high forests, woodlands, plantations 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 among countries with a forest cover of 10-30%. Based on this report Ethiopia’s forest cover is 12.2 million ha (11% of the land area). On the other hand, the forest cover of the country declined from 15.1 million ha in 1990 to 12.2 million ha in 2010, with an estimated 141,000 ha annual rate of deforestation between 1990 and 2010. About 2.65% of the forest cover was deforested during this period (FAO, 2010).

Historically, settlement in Ethiopia was influenced by avoiding the risk of tropical diseases such as malaria and trypanosomiasis. As a result, humans and livestock settlements have concentrated in the highland areas, especially in the 2300 – 3200 m above sea level range. This population pressure on the highlands coupled with sedentary rainfed agriculture and extensive cattle herding activities has resulted in heavy deforestation, habitat fragmentation and over-exploitation of species (Eshetu Yirdaw, 2002). The remaining natural forests are located primarily in southern, southeastern and southwestern parts of the country. High forests in these areas have been identified and efforts are being made to manage, protect and conserve these resources on a sustainable basis. At present, however, accessible high forest areas are exposed to various development project pressures, including inadequate consideration to biodiversity conservation in the development plan, conversion of natural forest areas into large scale tea and coffee plantations, human resettlement, illegal charcoal production, overexploitation and pollution of Rift valley lakes and illegal timber extraction from forests with threatened indigenous tree species (Badege Bishaw, 2001; IBC, 2009), among others.

The degradation of natural resources in general and deforestation, in particular, have resulted in land degradation and soil erosion. These coupled with other factors have led to aggravating the impacts of recurrent drought, erratic and insufficient rainfall, leading to desertification, now recognized as the result of climate change (increase in temperature, shortening of rainy season). If the major threats mentioned above are not adequately addressed and reversed, the biodiversity of the country will deteriorate with huge adverse implications for the environment, in general, and on human well-being, in particular. Therefore, efficient and sustainable management plan for conserving the various aspects of biodiversity should be developed based on research findings to prevent the remnant natural forests from the verge of extinction.

2.3 Plant community theories

The concept of a plant community is very basic in the field of vegetation science and hence many ecologists attempted to give botanical definitions. Kent and Coker (1992) defined plant community as “the association of plant species growing together in a particular location that shows a definite association or affinity with each other”. The idea of association is very important and implies that certain species are found growing together in certain locations and environments more frequently than would be expected by chance. A plant community denotes associations of plants occurring in particular locality and dominated by one or more prominent species in a given time and space (Begon *et al*., 1996; Ricklefts, 1997). Most environments of the world support certain associated species which can, therefore, be characterized as a plant community. A plant community can also be understood as a combination of plant species that are dependent on their environment and influence one another and modify their own environment (Muller-Dombois & Ellenberg, 1974).

The reason behind certain species grow together in a particular environment may usually be because, first, they have similar requirements for existence in terms of environmental factors such as light, temperature, soil moisture, drainage and soil nutrients (Kent & Coker, 1992). Second, biotic factors like the competitive ability of species and facilitative aspects such as shade tolerating vs. light phase species (Walter, 1985; Chapman and Reiss, 1992) influence the distribution of species and transition of communities. It is the spatial change of the environmental factors and the subsequent response of the individual populations of a species to these changes that influence the distribution of species and therefore, the transition of communities (Chapman and Reiss, 1992). In addition to the environmental and biotic factors, community organization may be influenced by temporal factors. These include the dispersal mechanisms and subsequent accumulation of species in a site (Jacquemyn *et al.,* 2001) and the developmental stage of vegetation or succession. Plant community level study is a useful approach to conservation planning. The concept of plant community, for instance, provides useful information on the underlying environmental drivers of species distribution, as plant species that live together have similar environmental requirements for their existence.

The debate regarding the nature of community structure has been going on for long time. Basically, two theories have been proposed to explain the structure of plant community. These are: the community-unit theory of Clements (1936) and the continuum theory of Whittaker (1951; 1953) and Curtis (1959), the latter based on Gleason's individualistic distribution of species (Gleason, 1926). Community-unit theory states that communities are highly structured repeatable and identifiable associations of species controlled by environmental gradients. Conversely, continuum theory states that plant communities change gradually along complex environmental gradients, such that no distinct associations of species can be identified (Whittaker, 1953). Although differences still exist among ecologists on the concepts of plant communities, plant ecologists who favor vegetation classification follow the approach of Clements and group species into communities. Those ecologists who do not accept classification follow the continuum theory and arrange species along environmental gradients as continua, using ordination methods (Kent and Coker 1992).

2.3.1 The community-unit theory (The discrete community concept)

The community unit theory implies the existence of distinct communities (Walter, 1971) and it explains plant communities as clearly recognizable and definable entities which repeated themselves with great regularity over a given region of the earth's surface. Therefore, the distinctive vegetation of each area represents a distinct community, which is separated by the sharp vegetational transition from other communities (Ricklefts, 1997). This theory viewed communities as holistic and interdependent and predicts that groups of species or communities would replace one another along certain gradients. Within each grouping, most species have similar distributions and the end of one group coincides with the beginning of another (Shipley and Keddy, 1987). In this case species with wider distribution ranges are avoided and distinction 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; Kent and Cooker, 1992). Discontinuities over the continuous environment pattern are an important feature of discrete communities and the existence of such communities could be attributed to the competition and exclusion of other less competitive ones by the few dominant species (Robert, 1987) and subsequent modification of the environment by the vegetation.

The plant community is the basic unit in community unit theory and may be represented by a group of relatively homogenous samples that could be classified based on floristic similarity into a hierarchical order (Palmer and Van Staden, 1992). Moreover, this view regards communities as having a degree of internal organizations, which jointly modifies the environment with sharp delimitation from other environments. Species belonging to a community are closely associated with one another implying, the ecological limits of each species will coincide with the distribution of the community as a whole (Barbour *et al*., 1987).

2.3.2 The continuum theory (The individualistic concept)

The continuum theory contemplates that plant species respond individually to variation in environmental factors and those factors vary continuously in both space and time. Thus, the combination of plant species found at any given point on the earth’s surface was unique (Kent & Coker, 1992). 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). Each species is distributed in its own way, according to its own genetic, physiological, and life-cycle characteristics and its way of relating to both physical environment and interactions with other species and hence no two species are alike in distribution (Whittaker, 1975). Species distributions and abundances are based on the ranges in their tolerance to various abiotic factors and resource requirements.

Empirical evidences suggest that the continuum in its current form does not fully describe the observed patterns of vegetation along environmental variables (Collins *et al*., 1993). In addition, both community unit and continuum concepts give greater emphasis to environmental factors and very little attention to the modification effect of the vegetation and their subsequent influence on the pattern of the community (Robert, 1987). Thus, there is a need for a hypothesis that integrates the two different concepts to resolve this issue. Accordingly, Collins *et al.* (1993) have proposed a third alternative, the hierarchical continuum concept of plant community organization. The hierarchical continuum concept incorporates the dynamic nature of vegetation (Roberts, 1987) and the variation in environmental gradients. With respect to the dynamic nature of vegetation, species will change their distribution and abundance patterns along the gradient in response to environmental fluctuations (Collins *et al.*, 1993). Environmental resources are not uniformly distributed. As a result, plant species are not uniformly distributed. Based on the above points, the hierarchical continuum concept assumes that some species will have a wider distribution, while others are localized and still some others will have a much-restricted distribution across the sample area. Thus, the distribution pattern and abundance of species assume a hierarchical structure where species with wider, intermediate and restricted distribution ranges show some kind of hierarchies than a continuum or discrete alone. Both species with wider and restricted range of distributions were not used by association analysis for community classification in the community unit theory. A plant community with a hierarchical continuum concept can be understood as a combination of plants that are dependent on their environment, influence one another, and modify their environment (Mueller-Dombois and Ellenberg, 1974). The hierarchical continuum concept of Collins *et al*. (1993) can, therefore, be viewed as a modern synthesis that recognizes the validity of both views and their complementarities in their application for different aspects of community analysis.

2.4 Community diversity, evenness and richness

2.4.1 Community diversity

The two fundamental parameters when considering the basic structure of biological communities or ecosystems are the number of species and the number of individuals within each of these species (Hamilton, 2005). Although ecologists (e.g. Hurlbert, 1971; May, 1975; Sugihara, 1980) have studied the inter-relationships between them (number of individuals within species and number of species) over many decades, there is lack of clear distinctions between these two different concepts (Hamilton, 2005) and hence in most cases they are used interchangeably.

Community 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 or species equitability) (Pielou, 1966; Spellerberg, 1991). One approach to measure community diversity is using indices commonly known as diversity indices. Thus, the description of plant communities involves the analysis of community diversity, richness and evenness indices. Diversity and equitability (evenness) 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 (Kent and Coker, 1992).

2.4.2 Species richness

According to Hamilton (2005), species richness can refer to the number of species present in a given area without considering the number of individuals in each species. It is one of the oldest and most fundamental concepts (Peet, 1974). Species richness is the simplest way to describe community and regional diversity (Magurran, 1988), and thus it forms the basis of many ecological models of community structure (Connel, 1978; Stevens, 1989). It is a measure of the number of different species in a given site and can be expressed in a mathematical index to compare diversity between sites (Zerihun Woldu, 1985).

Measuring species richness is important for basic comparisons among communities. Maximizing species richness is often the goal of conservation studies (May, 1988), and current rates of species extinction are calibrated against the patterns of species richness (Simberloff, 1986). Measures of community diversity and species richness have been broadly used as indicators of ecosystem status, and play a critical role in studies dealing with the assessment of human impact on ecological systems (Leitner and Turner, 2001). However, since the biodiversity of any ecosystem is very complex to be comprehensively quantified, suitable indicators of biodiversity are needed. Conceptually, species richness is the most straightforward parameter to measure biodiversity. Nonetheless, for several reasons, determining the true species richness of a community is not an easy task (Magurran, 2004).

2.5 Measures of community diversity

Species are the most widely used level of biological organization in the study of biodiversity since they are easily detectable and quantifiable in nature. The number of species that can be found on a particular site or region is a variable that can be measured without any notable technical or conceptual difficulties. Spatial scale is very important in the evaluation of community diversity since the processes that influence biodiversity vary with scale (Gering and Crist, 2002). So, at the local or community level, ecological processes like niche structure, biological interactions, environmental variables, etc exert the greatest influence while at the regional level, evolutionary and biogeographical aspects (dispersal, extinction, speciation, etc.) are the most important. On the landscape scale, both sets of processes affect the number and quality of species (Ricklefs & Schluter, 1993).

Community diversity patterns are the results of historical, evolutionary and ecological processes that vary across geographical regions, and temporally within each region (Ricklefs and Schluter, 1993; Rosenzweig, 1995). Therefore, understanding the processes that are responsible for diversity patterns at different geographical scales has been a vital issue in vegetation ecology (Cody, 1993). Measuring diversity at different scales: alpha, beta and gamma diversity help in understanding the causes shaping patterns of diversity (Whittaker, 1972). Alpha diversity (α) refers to the diversity of species within a particular habitat or community. Beta diversity (β) is a measure of the rate and extent of change in species composition along an environmental gradient from one habitat to another. It is sometimes called habitat diversity since it represents differences in species composition between very different areas or environments and the rapidity of change of habitats (Kent & Coker, 1992). It calculates the number of species that are not the same in two different communities. Beta diversity has gained considerable value as a conservation tool by representing either species turnover in space or time, or ecological connectivity

(Whittaker, 1975). Thus, β-diversity can help in defining regional-scale diversity and assessing changes across environmental and biogeographic gradients through characterizing the rate of species accumulation from place to place. A high β-diversity index indicates a low level of similarity in species composition between the two areas or habitats and vice versa. Gamma diversity (γ) is a measure of the overall diversity for the different ecosystems within a region (Whittaker, 1972). It is the diversity of species in comparable habitats along geographical gradients (Kent and Cooker, 1992). Gamma diversity of a landscape, or geographic area, is a product of the alpha diversity of its communities and the degree of beta differentiation among them (Whittaker, 1972).

Various types of diversity indices have been used to measure community diversity. Magurran (2004) provided an in-depth review of concepts and measurements of diversity. Species richness, the measure of the number of different species in a given site or area, is often taken as community diversity. However, most methods used in measuring community diversity consist of two components (Magurran, 2004). These are the species richness and the relative abundance (evenness or equitability) of species within a sample or community. Whittaker (1972) considers species richness as a strong measure of community diversity. However, using species richness alone as a measure of diversity is criticized, because species richness is just one component of community diversity (Hurlbert, 1971; Sanjit and Bhatt, 2005). The Shannon-Wiener and Simpson's indices of diversity, which combine species richness with relative abundance, are widely used to measure community diversity (Kent and Coker 1992). The Shannon index expresses the relative evenness or equitability of species, while Simpson's index gives weight to dominant species (Whittaker 1972). Therefore, in this study, Shannon-Wiener diversity index is computed to evaluate the community diversity and evenness (equitability) of the study area. The values of Shannon-Wiener diversity index naturally varies between 1.5 and 3.5 and rarely exceeds 4.5 while equitability assumes a value between 0 and 1 with 1 being complete evenness (Kent and Coker, 1992).

2.6 Multivariate data analysis

Vegetation classification has been a major activity in vegetation science for a long time (Kent & Ballard, 1988). Owing to its high relevance for vegetation mapping; a topic that has received considerable interest in light of global climate and land-use change (Xie *et al.,* 2008) and it is also highly relevant for nature conservation.

Plant community data are multivariate in nature and their analysis is closely related to the development of multivariate techniques, i.e. methods of multivariate analysis to study the joint relationship of variables (Stephane *et al.*, 2008). Multivariate techniques are employed to study the complex nature of plant communities with the general objectives of summarizing large complex data sets obtained from community samples, aiding in the interpretation of the data and the generation of hypothesis about community structure and variation (Gauch and Whittaker, 1972; Gauch, 1982). Recently, a wide variety of multivariate techniques are available to study the complex nature of the plant communities. Among the multivariate techniques for studying the complex nature of plant communities; classification and ordination are the two main and basic techniques (Mueller-Dombois and Ellenberg, 1974; Whittaker, 1975). Both classification and ordination techniques continue to contribute considerably to the explanation of the complexities within communities. Therefore, the choice of the method to be used depends on the type of question to be answered (Gauch and Whittaker, 1972; Whittaker, 1975).

2.6.1 Classification

Putting items into groups is a natural mental activity. This is possibly one reason for the existence of various classification methods. Classification is the placement of species and/or sample units into groups based on their similarity and differences. It aims at grouping individual stands into categories. The members of each category posses common attribute which serves to set them apart from members of another category (Anderson, 1965). Those stands, which are closely similar with one another normally, form one class. Vegetation classification attempts to identify discrete, repeatable classes of relatively homogenous vegetation communities or association about which reliable statements can be made. The groups derived from a set of individual sample units (quadrats) through classification on the basis of their floristic content are usually taken as the plant communities of the study area (Kent & Coker, 1992).

Classification methods can be classified into hierarchical or non-hierarchical. A non-hierarchical method of classification, where groups have no joint structure, may be chosen when there is no particular advantage in the groups of species or sites being arranged in a hierarchy (Digby & Kempton, 1987). The aim of such method is producing the most efficient groups regardless of the route by which they are derived (Greig-Smith, 1983). If homogeneity of a group is of a prime importance in the application process, the non-hierarchical methods are best fit. In hierarchical methods, samples having the same properties are arranged into classes and the classes at any level are subclasses of classes at a higher level. Therefore, the groups themselves are arranged into a hierarchy (Digby and Kempton, 1987).

Hierarchical classification methods can be further divided into agglomerative and divisive methods. Agglomerative hierarchical methods consider each unit as being a separate group and proceed by repeatedly combining the two closest groups until only a single group remains while divisive methods start with all the units as one group and proceed by repeatedly dividing groups into two until all the units are separate (Digby & Kempton, 1987). Divisive methods thus concentrate essentially on differences, while agglomerative methods seek similarities. Divisive methods start from maximal information obtained over the whole population, while agglomerative techniques start from single units of minimal information. Divisive methods can be terminated at any convenient levels, while agglomerative methods require the whole analysis to be completed before the large-scale divisions at the top of the hierarchy can be obtained. In general, agglomerative hierarchical methods are becoming popular methods (Lambert and Dale, 1964; Greig-Smith, 1983; Digby and Kempton, 1987).

2.6.2 Ordination

Ordination means arranging items along a single or multiple axes and is often used to seek and describe patterns. It involves the arrangement of vegetation samples in relation to each other in terms of their similarity of species composition or associated environmental variables. Ordination methods can help in the following ways: 1) summarizing plant community data and providing an indication of the true nature of variation within the vegetation of the study area, 2) enabling the distribution of individual species within different communities to be examined and compared, and 3) providing summaries of variation within sets of vegetation samples which can then be correlated with environmental controls to define environmental gradients (McCune and Grace, 2002; Kent and Coker, 1992). The result of ordination in two dimensions (two axes) is a diagram in which sites are represented by points in two-dimensional space. The aim of ordination is to arrange the points (species or samples) such that points that are close together correspond to sites that are similar in species composition, and points that are far apart correspond to site that are dissimilar in species composition (ter Braak, 1995). Similarly, if the ordination is on species space, points close together represent that those species have similar occurrences in their distribution.

Ordination methods are also part of gradient analysis. In gradient analysis, variation in species composition is related to variation in associated environmental factors, which can usually be represented by environmental gradients. Thus, ordination methods emphasize on individual samples or species and their degrees of similarity to each other and on determining how the order of the individuals is correlated with underlying environmental controls. Correlation analyses between environmental factors and ordination axes scores can be used to identify and describe environmental factors that influence species distribution patterns (Kent and Coker, 1992; McCune and Grace, 2002).

Both gradient analysis and ordination methods are essentially descriptive and enable researchers to formulate ideas about plant community structure as well as possible causal relationships between variation in vegetation and its environment. Ordination or gradient analysis can be direct or indirect. Direct gradient analysis or ordination is used to display the variation of vegetation in relation to environmental factors by using environmental data to order the vegetation samples. It is used to study how species are distributed along specific environmental gradients of interest (Kent and Coker, 1992; Austin *et al.*, 1994; Oksanen and Minchin, 1997). Direct gradient analysis can be univariate where sample units are ordinate along a single environmental factor or multivariate. Species distribution along an altitudinal gradient is an example of a univariate gradient. Indirect gradient analysis or ordination ordinates sample units by examining the variation within the vegetation independently of the environmental data. Once the major sources of variation in the vegetation data have been described and summarized, the environmental data will be compared and correlated with the summarized vegetation data in order to detect possible environmental gradients. Indirect ordination methods can be used in situations where the underlying environmental gradients are unknown, although they are equally applicable where the environmental gradients are known. Indirect ordination methods are more widely used than direct ordination methods due to the fact that species data are usually easier to collect than environmental data (Whittaker, 1967; Kent and Coker 1992). In addition, plant ecologists are more often concerned with community structure in relation to environmental factors rather than with the response of individual species which are often very unpredictable (Kent & Coker, 1992).

Indirect ordination methods are performed in a number of different techniques like Bray and Curtis polar ordination (PO), reciprocal averaging/correspondence analysis (RA/CA) and detrended correspondence analysis (DCA) (Kent & Coker, 1992; McCune and Grace, 2002). Each of the above indirect ordination technique has its own advantage and disadvantage, and the choice of ordination method depends on the type of the data, the sampling effort, and objectives of the study (Southwood and Handerson, 2000).

Ordination methods, which use environmental variables in the construction of the ordination, are called constrained ordination or direct ordination since the positions of the samples in the ordination are constrained by the environmental variables (1995). Direct ordination is often performed using Canonical Correspondence Analysis (CCA) although Redundancy Analysis (RDA) and principal component analysis (PCA) may also be available for the same purpose.

CCA is a direct gradient analysis method and currently one of the most popular ordination techniques in community ecology. CCA differs from other ordination methods in that it incorporates the correlation and regression between floristic and environmental data within the ordination analysis itself (ter Braak and Prentice, 1988).

2.7 Natural regeneration of woody plant species

FAO (1993) has shown that 2.5% of higher plant species were lost due to deforestation in the montane regions of tropical Africa between 1981 and 1990. Extensive deforestation has led to severe depletion of natural forests, loss of biodiversity and land degradation in Ethiopia (EFAP, 1994; IBC, 2009). In fact, deforestation has eroded the biological diversity to such an extent that some plants are facing local extinction. Owing to its importance to forest management, the dynamics of regeneration after exploitation has received particular attention (Bazzaz, 1991).

Regeneration is a key ecological process and a central component of forest ecosystem dynamics and restoration. Demel Teketay (2005) indicated that a good understanding of natural regeneration in any forest ecosystem requires information on the presence and absence of persistent soil seeds or seedling banks, quantity and quality of seed rain, losses of seeds to predation and sources of regrowth after disturbances. Variation in patterns of regeneration both through differences in their constituent species and the environmental variables in which they grow was observed in tropical forests (Whitmore, 1996; Demel Teketay, 1997). According to Garwood (1989), tropical forest plants regenerate in different pathways, i.e. (1) *seed rain*: seeds dispersed recently; (2) *soil seed bank*: seeds that are in the soil; (3) *seedling bank*: seedlings found in the understorey; and (4) *coppice*: root or shoot sprouts of damaged individuals. The success of woody plant regeneration in a forest is determined by successful completion of several events in the plant life cycle such as coppicing (sprouting) ability (Garwood, 1989; Evans, 1992; Bellingham and Sparrow, 2000), seed production, dispersal to favorable sites, germination and seedling emergence, seedling establishment and onward growth (Barik *et al.*, 1996). However, all these events can be affected negatively or positively by ecological factors. Therefore, understanding the patterns of regeneration enables to undertake proper forest management plan that helps to conserve the biodiversity and utilize the forest ecosystem wisely and sustainably.

2.7.1 Regeneration pattern and population structure of woody plants

Analysis of population structures, using the size-class distribution of diameter at breast height (DBH) and height of trees and density of naturally regenerated woody species, can provide an insight into their regeneration status (Silvertown, 1982; Silvertown & Doust, 1993). Plant population structure shows whether or not the population has a stable distribution that allows continuous regeneration to take place. Plant population structure may change due to changes in recruitment of individuals at low diameter size classes or exploitation of individual at high size classes or throughout the class size structure. The population structural change is the function of regeneration pattern of individuals within the community. If regeneration was taking place continuously, then, the distribution of species would show reverse J shape structure, which is an indicator of healthy/good regeneration (Silvertown, 1982; Demel Teketay, 1997), which is common in natural forests where external disturbances are minimum (Feyera Senbeta, 2006; Getachew Tesfaye *et al.*, 2002; 2010). Any population structural pattern differed from inverted J-shape (e.g. U-shape, J-shape and bell shape) have been attributed to disturbed forests (Poorter *et al.,* 1996).

The variation in relative abundances of size classes and population structure of species were the result of past and present disturbance as well as management history of the forest. Based on the intensity of disturbances woody species show variation in population structure pattern reflected through differences in the abundance of different size classes (Tamrat Bekele, 1994; Demel Teketay and Granstrom, 1997). Ecologically, there is co-existence of individuals both with small and large size classes among successional associations unless disturbance has occurred. Thus, size class distribution of woody species is a good indication of the impact of disturbance and the forest successional trends (Mligo & Rocky, 2012). Such information is valuable for the understanding of conservation needs of forest ecosystems. Several studies in Ethiopia have shown variations in population structure and regeneration pattern in forest tree species that has been attributed to different levels of disturbance of forest species (Demel Teketay and Granstrom, 1997; Feyera Senbeta, 2006).

Successfully regenerating plant species are characterized by the presence of a sufficient number of individuals at both lower (seedling and sapling) and higher size classes including the reproductive individual trees that are capable of contributing a significant number of viable seeds (Mligo & Rocky, 2012). Lack of individuals at low size classes and mature reproductive individuals may be related to their regeneration failure in combination with over-exploitation for poles, firewood, charcoal burning and timber extraction. Selective cutting of matured trees leaving out the few stunted and genetically poor individuals causes population decline (Lyaruu *et al*., 2000). Active regeneration could have been influenced by the presence of relatively abundant reproductive mature trees scattered in the forest. However, poor seedling densities might be caused by intensive exploitation of mature reproductive individuals (Mligo & Rocky, 2012). Seedling densities in forest understoreys are dynamic and rates may vary among species and in gap and shade environments (Bazzaz, 1991). The rates also vary due to mortality, drought and biotic factors such as herbivory, disease or competition (Augspurger, 1984). Understanding tree seedling ecology can provide alternatives for forest development by improving the recruitment, establishment and growth of the desired species (Swaine, 1996). Therefore, regeneration studies have major roles in the management, conservation and restoration of degraded natural forests.

2.7.2 Factors affecting regeneration of woody plants in a forest ecosystem

The regeneration potential of a given forest in general and a species, in particular, can be affected by several ecological factors. For instance, disturbances such as tree fall gaps, livestock grazing, and logging affect the abundance and composition of the seedlings in the forest understorey (Benitez-Malvido, 1998; Alemayehu Wassie *et al.*, 2009) and therefore, play an important role in woody plants regeneration processes. Natural or anthropogenic disturbances have much influence on community composition, tree population structure and regeneration ability of forest ecosystems. The process of natural regeneration of forest stands begins with the supply of seeds from the soil. The regularity of seed production varies greatly among species. Some species produce good seed crops annually, others show one or two good crops every four to six years (Pardos *et al.*, 2005), while in most of the tree species seed production is irregular (Demel Teketay, 2005). The availability of seeds (both in quality and quantity) and favorable environmental conditions affect seedling establishment, an ecological process that encompasses seed germination and emergence of seedling (Alemayehu Wassie, 2007). Because seed limitation is strong for many plant species (Demel Teketay & Granstorm, 1995; Svenning and Wright, 2005), this factor mainly affects the number of seedlings to establish. Seeds are available for germination from seed rain (recently dispersed seeds) and soil seed bank (dormant seeds in the soil) (Garwood, 1989; Demel Teketay, 1997; Getachew Tesfaye *et al.*, 2002; Alemayehu Wassie, 2007). The amount of seed rain may be reduced due to changes in tree phenology as a result of increased mortality of reproductive individuals that might lead to a reduction of flowering and fruiting and increased pre-dispersal seed predation (Alemayehu Wassie, 2007). Even if there is enough quality seed, unless it is coupled with appropriate dispersal mechanism still there could be a shortage for seedling establishment. Seed dispersal is thought to enable seeds to escape competition (with their parents, with other seedlings and/or with surrounding vegetation) and to colonize favorable sites. According to Barot *et al.* (1999), in tropical forests, as a seed is far from its parent tree, the more likely it is to germinate and to recruit later as a seedling. The presence of viable seed in the soil seed bank depends on the inherent properties of seeds such as viability, dormancy and germination as well as the environmental conditions where seeds land and the presence of seed predators and pathogens. Actually, some species lose their viability immediately after they are released from the mother plants while others may persist to have their viability for longer periods (Demel Teketay and Granstrom, 1995; Demel Teketay, 2005). 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.

Besides the availability of seeds, the presence of favorable environmental condition plays an important role for successful seedling establishment. If seeds do not obtain the optimum environmental condition i.e., moisture, light and temperature, they will not germinate and stay dormant waiting for favorable condition to come (Demel Teketay, 2005). The germination and dormancy mechanisms are of great adaptive importance to plants in ensuring that seedling emergence occurs at the most advantageous time and place (Bewley and Black, 1994). Owing to its own characteristic requirement of each species, a site, which may be favorable for one species, may not be favorable for another. Even though several species share many characteristic features such as life form, phenology and habitat range, they might have different seedling requirements (Grubb, 1977). The heterogeneity in edaphic, biotic and abiotic factors in an ecosystem can, therefore, play a vital role in both species as well as genetic diversity through their impact on seed germination. Because of this heterogeneity, the same site may not be favorable for different species or individual of the same species. For instance, the emergence of tree species exhibited great variation in response to different layer of litter (Facelli & Pickett, 1991). The litter layer constitutes a physical barrier for seeds, seedlings, and shoots because it may impede or retard seeds reaching the soil, and inhibit the emergence of seedlings or sprouts. Seeds retained in the litter may either have delayed or unsuccessful germination. Seedlings that germinated within the litter layer died because their roots were unable to reach the soil (Fowler, 1986). Seedlings and sprouts emerging from beneath a litter layer have to devote energy and time to penetrate it. Seedlings produced by small seeds may be unable to emerge, because of energy shortage. Even seedlings originating from large seeds can be confined within the shady and wet environment of the litter layer (Sydes & Grime, 198l), increasing the risk of fungal infection or herbivore attack. Establishment of many species is negatively affected by litter because of shading, mechanical impedance and reduced thermal amplitude in the soil or biochemical effects (Facelli & Pickett, 1991).

Another factor affecting the process of natural regeneration is the availability of optimum level of moisture and light to sustain subsequent seedling development and growth after seed germination. Different species show different requirements of moisture and light. For instance, Demel Teketay (2005) reported that seedlings of *Podocarpus falcatus* and *Ekebergia capensis* survived better under the shade while seedlings of *Bersama abyssinica* and *Juniperus procera*, survival did not differ under shade and in the open in dry Afromontane forests of Ethiopia. In addition to the light requirement, physical damages and destruction due to trampling and grazing (Demel Teketay, 2005; Alemayehu Wassie, 2007), competition by weeds and herbs raise seedling mortality. Therefore, in this study the type and effect of disturbance (grazing, fire, tree cutting and firewood collection) on natural regeneration of woody species in Bore-Anferara-Wadera forest is investigated.

2.8 Significance of land use/land cover change studies

Ellis & Pontius (2007) have shown that the rates, extents and intensities of human pressure on land are by far greater than before, affecting the status, properties, and functions of ecosystems, which in turn affect the provision of ecosystem services and hence human well-being. This has been demonstrated by Global Forest Resources Assessment (FAO, 2010) that estimated the net global change in forest area between 2000 and 2010 at around -5.2 million hectares per year and counted the net loss of forests for the 2000s as a whole as 52 million hectares. According to this assessment, the overall natural forest loss is high in the tropics. These changes cause great environmental concerns as they contribute to local and regional climate change, biodiversity loss, soil degradation and the pollution of water, soils and air (Chase *et al.*, 1999; Ellis & Pontius, 2007).

Land use/land cover is a fundamental variable that impacts and links many parts of the human and physical environments. According to de Sherbinin (2002), land use is the term used to describe human uses of land, or a series of operations on land, carried out by humans, in order to obtain products or benefits through using land resources. On the other hand, land cover refers to the natural vegetative cover types that characterize a particular land area. Land cover is the visible features of the Earth's surface included in the vegetation cover, natural and as modified by humans (Campbell and Wynne, 2011). In short, land cover represents the visible evidence of land use. A piece of land can have only one land cover (e.g., forests), but can have more than one land use e.g., recreational, educational, and conservational (Giri, 2012).

Land cover change is regarded as the single most important variable of global change affecting ecological systems (Vitousek, 1994). Both natural and anthropogenic forces are responsible for the change. Natural forces such as continental drift, glaciation, flooding, and tsunamis and anthropogenic forces such as conversion of forest to agriculture, urban expansion and forest plantations have changed the dynamics of land use/land cover types throughout the world (Giri, 2012). In recent decades, anthropogenic land-use/land-cover change has been proceeding much faster than natural change. This extraordinary rate of change has become a major environmental concern worldwide. As a result, almost all ecosystems of the world have been significantly altered or are being altered by humans, undermining the capacity of the planet's ecosystems to provide goods and services. Two main forces responsible for anthropogenic changes are technological development and the growing human population (Lambin and Meyfroidt, 2011). Owing to the rapid and extraordinary land-use/land-cover change in recent years, negative consequences such as soil erosion, loss of biodiversity, water pollution, and air pollution have increased. The benefits and economic gains provided by ecosystems have started eroding because these benefits are derived at the expense of degradation of the ecosystem (Giri, 2012).

Land use and land cover changes affect ecosystems in two major ways. First, the change on aquatic and terrestrial ecosystems and the second is climate change by contributing to carbon emission. Therefore, understanding the distribution and dynamics of land cover is crucial to better understanding of the fundamental ecosystem characteristics and processes, including productivity of the land, the biological diversity, and the biogeochemical and hydrological cycles. Information on land use and land cover change is needed to manage natural resources and monitor global environmental changes and their consequences (de Sherbinin, 2002).

Since the 1970s the focus on land use and land cover change emerged in the research agenda on global environmental change (Lambin *et al.*, 2003). The main focus of researchers and policymakers around the world has become both observing and mitigating the negative consequences of land use and land cover change and sustaining the production of essential resources (Ellis & Pontius, 2007). Although land degradation is a serious problem in Africa, it is most severe in the densely populated highlands of

East Africa (Pender *et al*., 2006). The Ethiopian highlands are among the most densely populated agricultural areas in Africa (Eshetu Yirdaw, 2002). Ethiopia has experienced recorded anthropogenic interference on ecosystems through land use change for four to five decades (Girma Hailu, 2000). Environmental degradation is one of the major constraints impeding sustainable development in the country, which can be expressed in terms of land and water resources degradation as well as loss of biodiversity (Demel Teketay, 2001). Most researchers are increasingly realizing the contribution of land degradation towards declining agricultural productivity and the continuing food insecurity (Girma Taddese, 2001). Therefore, knowledge of land use and land cover change, both on local and wider scale is essential in decision-making in relation to a wide range of issues, such as reversing land degradation, deforestation, and climate change. Improving the understanding of land use and land cover dynamics can lead to forecast the future land use and land cover change and to apply more appropriate policy interventions for achieving better land management (Lambin *et al.*, 2001).

2.8.1 Application of remote sensing (RS) and geographic information systems (GIS) in land use and land cover dynamics (LULCD)

Remote sensing is broadly defined as the science and art of obtaining information about an object, area or phenomenon through analysis of the data acquired by a device which is not in contact with the object, area or phenomenon under investigation (Reddy, 2008). It is the process of acquiring information about any object without physically contacting it in anyway. RS can also be defined as the practice of obtaining information about the Earth's land and water surfaces using images taken from an overhead perspective, using electromagnetic radiations reflected or emitted from the Earth's surface (Campbell and

Wynne, 2011), while GIS is a computer-based system that can deal with virtually any type of information about features that can be referenced by geographical location; and capable of handling both location data and attribute data about such features (Lillesand *et al*., 2004). GIS is a computerized database system for capture, storage, retrieval, analysis, and display of spatial data (Reddy, 2008). It offers the possibility of integration of geospatial data sets from different sources such as surveys, remote sensing, and recycled paper maps. Remote sensing simply measures the reflective response of the earth's surface, and so it can be used to directly observe the land cover for a given pixel. Land use must be deduced by relating the measured land cover with supplementary information such as socioeconomic data, field data, or regional and local knowledge that links a given land cover in a region with a given land use. Remote sensing excels at detecting the surface cover type and condition and provides a number of landscape attributes that can be used by LULC models (Campbell & Wynne, 2011; Giri, 2012).

Remote Sensing is a powerful technique for surveying, mapping and monitoring earth resources. This technology combined with GIS provides efficient methods for analysis of land use issues and tools for land use planning and modeling (Netsanet Deneke, 2007; Daniel Ayalew, 2008). GIS databases are used to improve the extraction of relevant information from remote sensing imagery, whereas RS data provide periodic pictures of geometric and thematic characteristics of terrain objects and improving our ability to detect changes (Janssen, 1994). The potential of remote sensing and GIS in the field of forestry become established over many years through the use of aerial photos and satellite image interpretations in forest cover change detection analysis, for the generation of the cover map and inventory analysis (Meseret Mideksa, 2009).

Remote sensing brings together a multitude of tools to better analyze the scope and rate of deforestation and provides multi-temporal data for change detection analysis. Images of earlier years are compared to recent pictures, to actually measure the differences in the sizes and extents of forest cover change. RS data can also be combined with other data to address a specific practical problem, such as LULC change, land-use planning, mineral exploration, or water quality mapping (Netsanet Deneke, 2007; Meseret Mideksa, 2009). In recent years, remote sensing techniques have been proved to be of immense value for preparing accurate LULC maps and monitoring changes at regular intervals. This technique is perhaps the best method of obtaining the required data with a reasonable cost and time in regions that are not easily accessible.

CHAPTER THREE

3 MATERIALS AND METHODS

3.1 Description of the study area

3.1.1 Location and Topography

This study was conducted in Bore-Anferara-Wadera forest located in Guji Administrative Zone of Oromia National Regional State, in Southern Ethiopia (Figure 1 & 2). Guji is bordered on the south by Borena zone, on the west by the Southern Nations, Nationalities, and Peoples Region, on the north by Bale zone and on the east by the Somali Region. The study area is situated 470 km south of Addis Ababa at 5^0 40’ N to 6^0 26’ N latitudes and 39^0 27’ E to 38^0 26’ E longitudes and it lies within three districts, namely, Bore, Adola, and Wadera. Three forest patches, Kilenso (1,944 ha) in Bore, Anferara (20,440 ha) in Adola and Danissa (1,578 ha) in Wadera were studied (NRSOBFED, 2012). Anferara forests are one of the Key Biodiversity Area (KBA) in the Eastern Afromontane Biodiversity Hotspot region recognized for biodiversity conservation (BLI, 2012). These forests contain vulnerable species such as *Leptopelis ragazzii* (Amphibian), *Serinus xantholaemus* and *Tauraco ruspolii* (Aves) and the severely logged tree *Ocotea kenyensis* (BLI, 2012). The topography of the study area is generally rugged with an altitude ranging between 1828 and 2304 m above sea level. Rolling plateaus, dissected hills, valleys as well as plains characterize the topography of the study area. The forest is also surrounded by two big river systems, Genale to the East and Awata (a major tributary of the Dawa River) on the West (EWNHS, 2001).

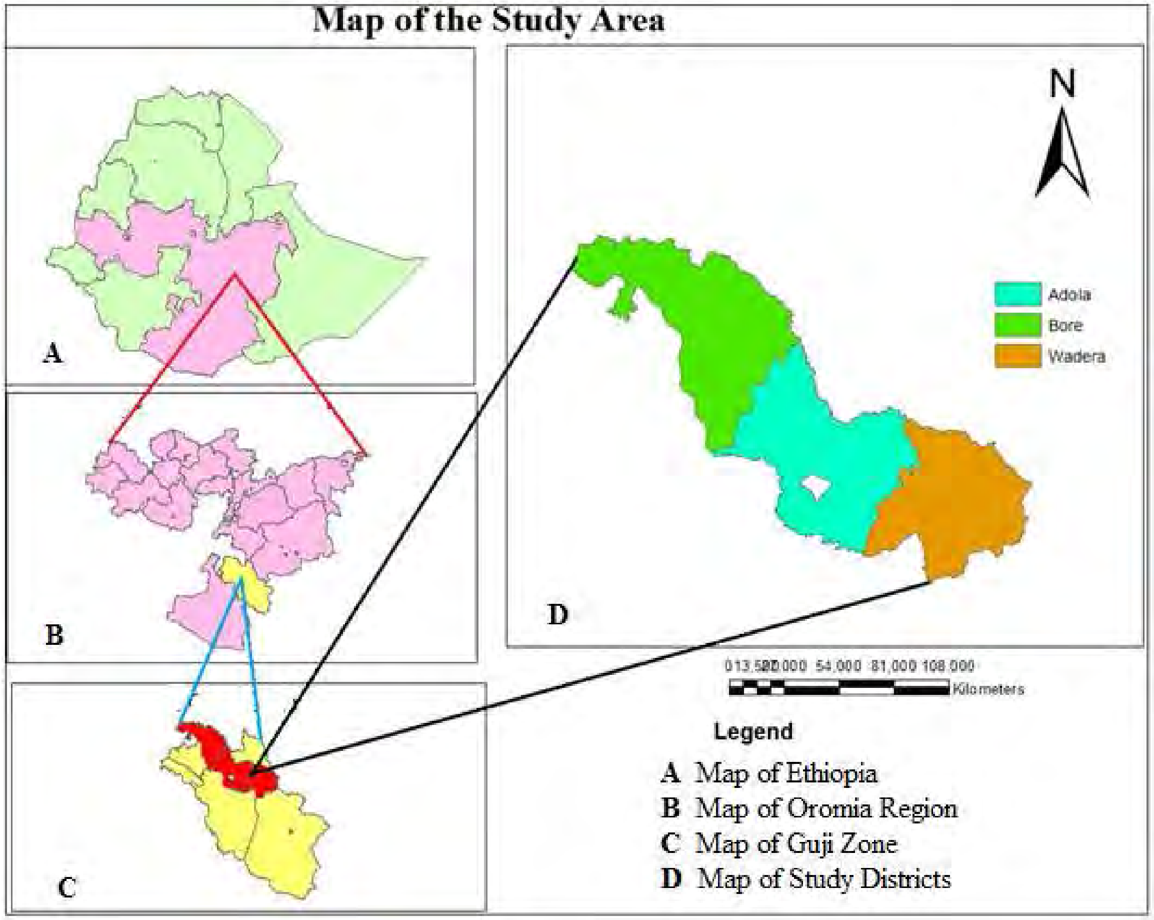


Figure 1 Map of Ethiopia showing the study area

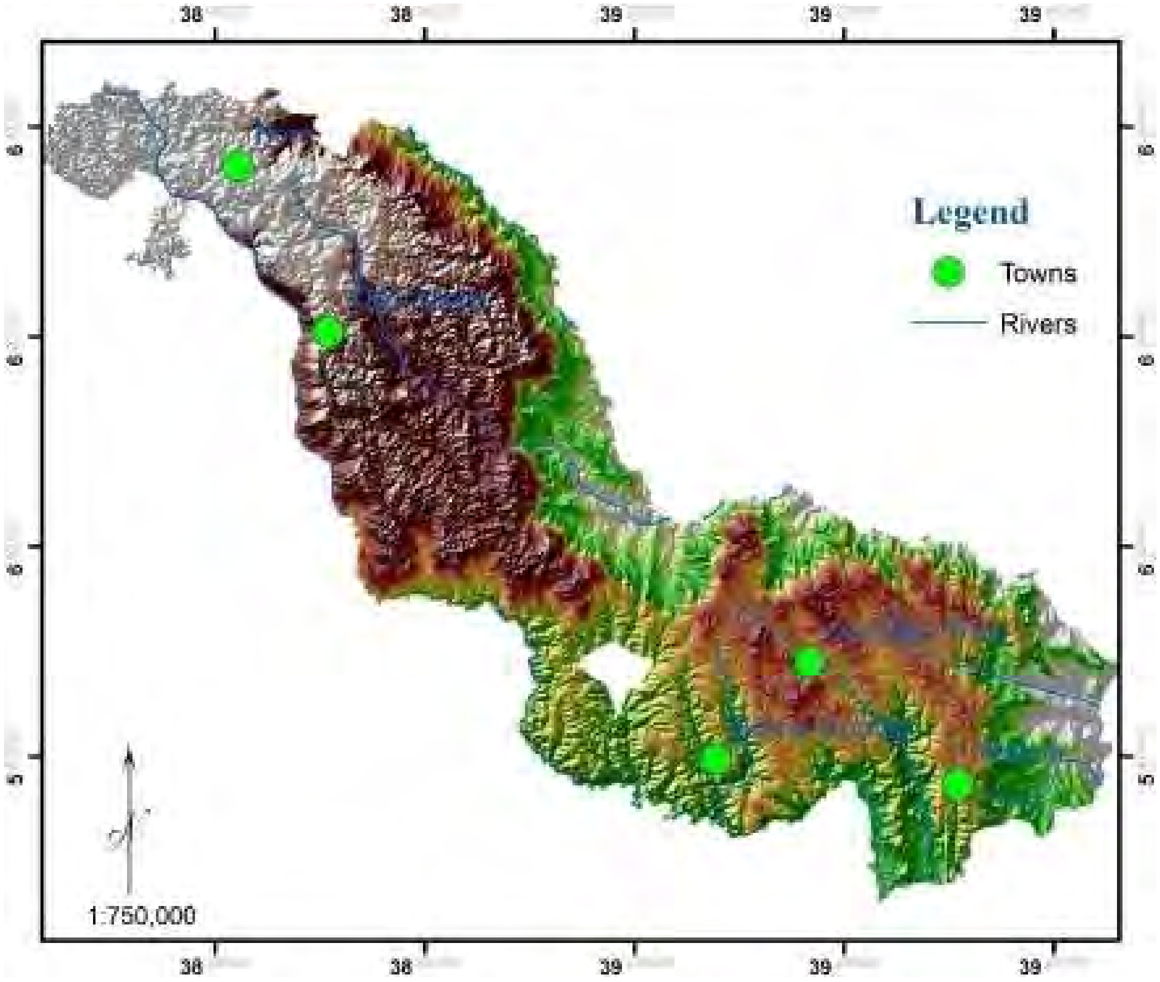


Figure 2 Digital Terrain Model of the Study area and the location of major towns and rivers in the study area

3.1.2 Geology and soil

According to Mohr (1983), tectonic activities (faulting and folding), glaciations and uplifting occurred in Ethiopia from the late Mesozoic to the early Tertiary Era. After these events, vast quantities of basaltic lava was pushed out and reached the surface. As thick series of lava flows (Trappean basalt layers) which often reach several thousand meters in thickness, they covered a large portion of the underlining Mesozoic rock. Mesozoic rocks mainly made up of sandstones and limestones, which are overlain by tertiary lava, cover the Precambrian rocks (Mohr, 1971). Precambrian rocks that underlie the whole of Ethiopia consist of a complex of metamorphic and igneous rocks of many different grades and types. The main rock types in the study area include limestone, shale, chlorite, talc, schists, amphibolites, muscovite schists, andalusite, and quartz (Mohr, 1971). The majority of soil types in the study area are dominated by fertile soils of volcanic origin. Nitosols (red basaltic soils), Luvisols, Cambisols, and Fluvisols are broadly found in highland areas and flat to sloppy terrain, and their utilizations are good under the natural vegetation (NRGOBFED, 2009a).

3.1.3 Climate

According to the climate data (2000 - 2014) obtained from the National Meteorological Services Agency (NMSA), the mean annual rainfall of the study area amounts to 814 mm (NMSA, 2015). The study area has a bimodal rainfall with a long rainy period occurring between late February to June and in September and October. High concentration of rainfall occurs in April (Daniel Gamachu, 1977). The dry period extends from December to February and in July. The mean annual temperature of the study area is 16.1 ^{0}C ranging from the mean annual minimum of 6.5 ^{0}C to the mean annual maximum of 25.3 ^{0}C (Figure 3).

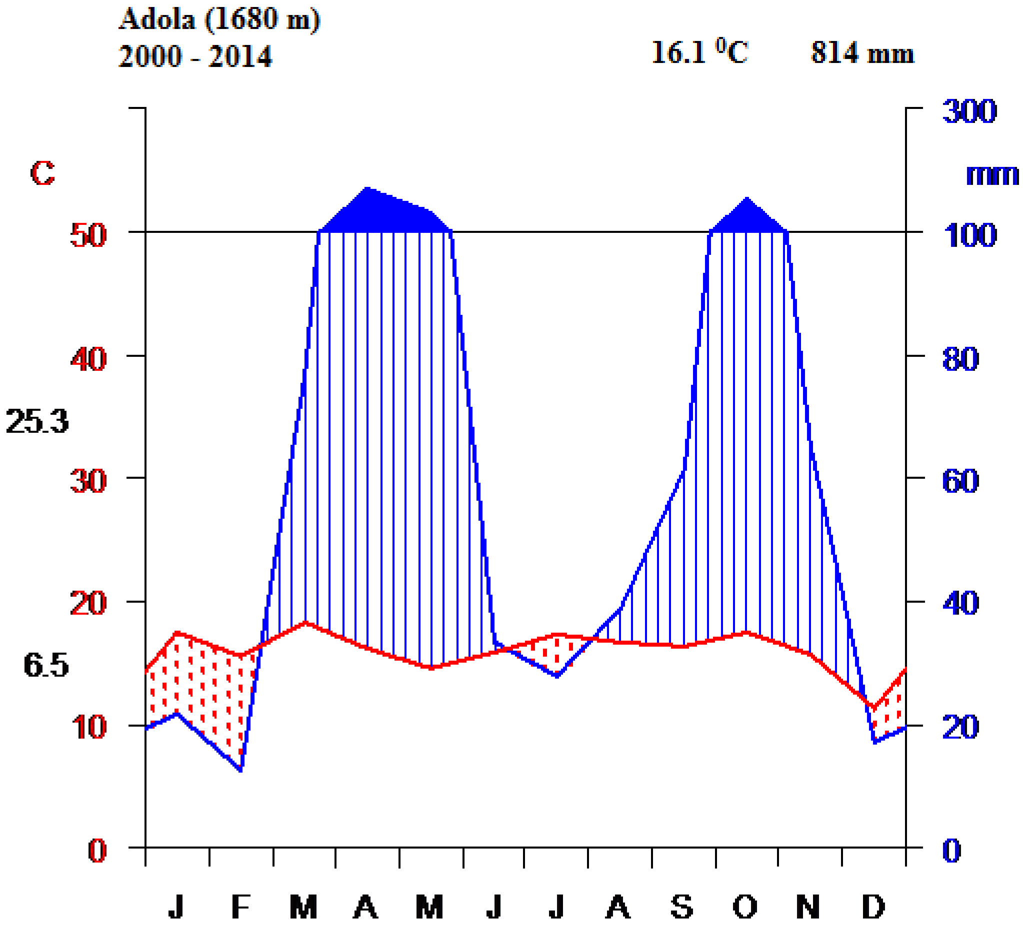


Figure 3 Climate diagram of Adola (Data Source: NMSA, 2015)

3.1.4 Vegetation

According to the current classification of Ethiopian vegetation (Friis *et al*., 2011), the natural vegetation of the study area falls in the categories of the moist evergreen Afromontane forest. This type of forest is characterized by one or more closed strata of evergreen trees that may reach a height of 30 to 40 m. Sometimes only the lower stratum remains owing to the removal of the canopy (Hedberg *et al.,* 2009). Large areas covered by this vegetation type have now changed to farmland, secondary montane grassland, secondary montane woodland and secondary evergreen bushland (Hedberg *et al.,* 2009).

The tree canopies typically contain a mixture of *Podocarpus falcatus* and broad-leaved species. *Podocarpus falcatus* is predominant in the southeast and becomes gradually more infrequent towards the southwest in Kefa and Illubabour as the rainfall increases, while *Pouteria adolfi-friederici* becomes more prominent in the same direction. *Pouteria adolfi-friederici* is the only emergent species in the study area in the 20-30 m high canopy layer. Other major canopy trees of 10-30 m height include *Albizia gummifera*, *Celtis africana, Croton macrostachyus, Ekebergia capensis, Ficus sur, F. thonningii, Macaranga capensis, Schefflera abyssinica*, *Ilex mitis*, *Prunus africana*, *Ocotea kenyensis*, *Polyscias fulva*, *Syzygium guineense* Subsp. *afromontanum* and *Olea capensis* Subsp. *hochstetteri*. The forest floor is dark, moist and usually dominated by the grass *Poecilostachys oplismenoides* and *Acanthus eminens*, *Isoglossa somalensis*, *Achyranthes aspera, Hypoestes forskaolii*, and different species of ferns (Friis, 1986; 1992; Hedberg *et al.,* 2009). *Ocotea kenyensis* is a vulnerable tree species in Anferara forests due to overexploitation of this species for different purposes by the local people (Bird Life International (BLI), 2012).

3.1.5 Fauna

The study area is also home to diverse variety of mammals and bird species. According to NRGOBFED (2009b), the most common wild animals found in Bore-Anferara-Wadera forests include spotted hyaena (*Crocuta crocuta*), vervet monkey (*Chlorocebus aethiops*), leopard (*Panthera pardus*), warthog (*Phacochoerus africanus*), colobus monkey (*Colobus guereza*), red fox (*Vulpes vulpes*), civet (*Civettictis civetta*), hare (*Lepus europaeus)*, bushbuck *(Tragelaphus scriptus)*, lesser kudu *(Tragelaphus imberbis),* and duikers (*Sylvicapra grimmia)*. The study area also contains different bird species including vulnerable bird species of *Serinus xantholaemus* and *Tauraco ruspolii* (BLI, 2012).

3.1.6 Population size, characteristics and land use type

As the study area lies in three districts, the total population of Bore, Adola and Wadera districts is 210, 078 (105, 671 males and 104, 407 females), 113, 735 (57, 821 males and 55, 914 females) and 50, 075 (25, 413 males and 24. 662 females), respectively. Of whom 10, 256 (4.88%) in Bore, 22, 937 (20.17%) in Adola and 4, 727 (9.44%) in Wadera are urban inhabitants (FDREPCC, 2008). The population density of Bore district is 162 People/km^2 while that of Adola and Wadera districts is 93 and 43 people/km^2, respectively, which is greater than the zone average of 34 People/km^2 (NRGOBFED, 2009b). The major ethnic groups living in these districts include Oromo (86.53%), Amhara (5.46%), Sidama (3.42%), Gedeo (3.03%), Soddo Gurage (0.91%) and Silt'e (0.1%). Oromiffa is the first language spoken by 84.59% of the people followed by Amharic (9.4%) and Gedeo (2.82%). About 37.45% of the population in Adola district are Protestant, 23.7% are Muslims, 20.44% are Ethiopian Orthodox Christians, 9.63% practiced traditional beliefs and 3.8% are Catholic whereas in Bore district 60.22% are Protestants, 17.62% practiced traditional beliefs, 8.47% are Ethiopian Orthodox Christians, 4.29% are Catholic and 1.33% are Muslims. In wadera district, the majority of the inhabitants are Muslims (64.18%) followed by Protestants (26.62%), Ethiopian Orthodox Christians (4.78%), people who practiced traditional beliefs (2.62%) and Catholic (1.11%) (CSA, 1996; 2007). With regard to land use type, about 33% of the total land area of Adola district is arable land (only 24% under cultivation), 30% pasture, 20% covered with forest (high forests, woodlands, bushlands, and plantations) and the remaining 17% is considered swampy and degraded area. In Bore district, 29% of the total land area is arable land (20.9% under cultivation), 33% pasture, 30% forest (including high forest, woodland, bushland, and plantations) and the remaining 8% is considered the swampy and degraded area. In Wadera district, 25.69% of the total land area is cultivated land, 28.63% pasture, 30.68% forest (including high forest, woodland, bushland, and plantations) and the remaining 15% is considered swampy and degraded area (NRGOBFED, 2009b).

3.2 Research Methods

3.2.1 Site selection and establishment of sampling plots

A reconnaissance survey was made from 2 to 25 January 2013 to get a general impression about the physiognomy of Bore-Anferara-Wadera forest and select sampling sites. Vegetation and environmental data were collected from 4 April to 15 June 2013, 28 November 2013 to 4 March 2014 and 2 December 2014 to 12 January 2015. As one of the objectives of this study was to compare community diversity and species richness along altitudinal gradients, systematic random sampling technique was employed to select study plots from Anferara, Kilenso and Danisa forests, which were selected based on low rate of disturbance, accessibility and concern for biodiversity conservation, for collecting vegetation and environmental data. Once the first sampling plot was established randomly, subsequent independent sampling plots were laid down along line transects at every 400 m interval between each sampling plots and 750 m apart between each line transect. An attempt was made to select representative sampling plots based on homogeneity of the vegetation and visually checking for uniformity in floristic composition. The dimension and total number of sampling plots were based on relevant plant ecological literature and represents a compromise between the recommended practice, accuracy and practical considerations of time and effort. Following Muller-Dombois and Ellenberg (1974) and Kent & Coker (1992), nested sampling plots of 30 x 30 m^2 for measuring trees and 5 x 5 m^2 and five 1 x 1 m^2, one at the center and the other four at each corner of the main sampling plot, subplots inside to measure saplings and herbs, respectively were used throughout the study area (Figure 4). The regeneration status of woody species was assessed by employing total count of all seedlings within the main sample plot. In total, 112 sampling plots (900 m^2 each): 50 plots from Anferara forest (Adola District), 40 plots from Kilenso forest (Bore District) and the remaining 22 plots from Danisa forest (Wadera District) were laid.

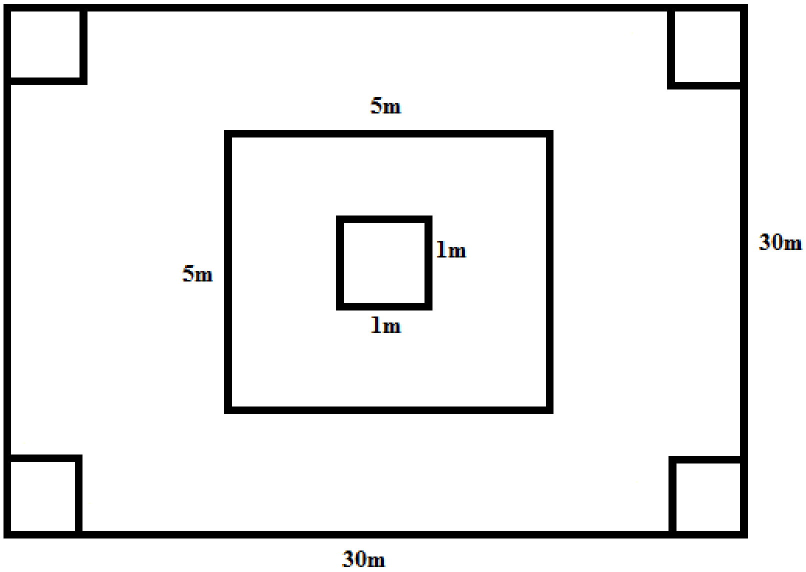


Figure 4 Layout of the sample plot

3.2.2 Vegetation data collection

In this study, plants with a height of ≥ 3 m will be considered as shrub or tree; plants between 1.5 m and < 3 m long will be considered as a sapling, and plants below 1.5 m long will be taken as seedlings, following Getachew Tesfaye and Demel Teketay (2005). In each sample plot and sub-plot, all woody plants were identified, counted, and height and diameter at breast height (DBH) was measured and recorded. Tree/shrub that branches at breast height or below, the diameter of each branch was measured separately and the average was taken. The height of woody plants was measured using Suunto Clinometer and calibrated bamboo stick, where topography and crown structure made it difficult to measure using Clinometer, and the circumferences of woody plants at breast height (about 1.3 m) were also measured and converted to the diameter at breast height (DBH). Cover/abundance values for all woody species were visually estimated in each sample plot and converted to Braun-Blanquet scale as modified by van der Maarel (1979). Growth forms of plants were listed and voucher specimens were collected. Specimens were dried, numbered, pressed, identified and deposited at the National Herbarium (ETH) in Addis Ababa University (AAU). Identification of specimens was performed both in the field and later at ETH using taxonomic keys in the Flora of Ethiopia and Eritrea and also by comparison with authenticated herbarium specimens. The nomenclature of plant specimens followed the Flora of Ethiopia and Eritrea.

3.2.3 Environmental data collection

Terrain variables such as altitude, aspect, slope and geographical coordinates were taken and recorded for each sample plot. Altitude and geographical coordinates were measured using Garmin 60 GPS; slope and aspect of each plot were measured using Suunto Optical Reading Clinometer and compass, respectively. Values for aspect were codified based on Zerihun Woldu *et al*. (1989), where N = 0, NE = 1, E = 2, SE = 3, S = 4, SW = 3.25, W = 2.5, NW = 1.25 before analysis.

Type and extent of disturbance were assessed for each sample plot. The magnitude of disturbance was rated from 0 to 3 following Feyera Senbeta *et al*. (2007) based on visible signs of disturbance such as livestock grazing, fire, tree stumps (tree cutting) and collection of firewood. Values were fixed as follows: 0: no disturbance; 1: if any one of the above-mentioned factors is present; 2: if any two of the above-mentioned factors are present; and 3: if all of the above-mentioned factors are present.

3.2.4 Land use/Land cover change data acquisition

Past and present information were collected in order to identify historical and recent land use/land cover dynamics of the study area. The basic data for the study were time series Landsat images acquired from the Global Land Cover Facility (GLCF; www.glcf.org) online imagery portal. Satellite image from Landsat TM (Thematic Mapper) for 1986, ETM$^+$ (Enhanced Thematic Mapper Plus) for 2000 and OLI/TIRS (Operational Land Image and Thermal Infrared Sensor) for 2014 were used (Table 1). Topographic map of the study area with a scale of 1:50,000 from EMA (Ethiopian Mapping Authority), Google Earth and focus group discussions with elder farmers and experts from Borena-Guji Forest and Wildlife Enterprise were employed to quantify the land use/land cover change in the study area. For ground truth verification, 80 agricultural lands, 76 forest land, 61 built up area, 67 shrublands, 64 grazing lands, and 69 wetland sample training sites have been collected.

Table 1 Landsat data used in land use/land cover classification

Landsat Type	Date of acquisition	Path and Row	Resolution
L1-5TM	February 1986	P168/r56	28.5*28.5
L7-ETM$^+$	February 2000	P168/r56	30*30
L8-OLI/TIRS	February 2014	P168/r56	30*30

Field observations were conducted to obtain Ground Control Points (GCPs) to support visual interpretation of the images and select reference areas [consisting of training areas (for supervised classification) and test areas (for accuracy assessment)]. All reference areas were localized by GPS measurements.

3.3 Data analysis

3.3.1 Cluster analysis

Cluster analysis is a multivariate technique which is widely used to group together a set of observations (plots or vegetation samples) based on their attributes or floristic composition (Kent and Coker, 1992; McCune and Grace, 2002). In ecology, cluster analysis is used to classify sites, species or variables into groups based on their similarities (van Tongeren, 1995). In this study, hierarchical (agglomerative) cluster analysis was performed using the free statistical software R version 3.1.1 (R Development Core Team, 2014) to classify the vegetation into plant community types. The abundance data of the species was used in this analysis. Dissimilarity ratio was used as a resemblance index (van Tongeren, 1995; McCune and Grace, 2002) and Ward's linkage method of grouping was applied:

$$1-\left[\frac{\sum\left(x_{k,i}*x_{k,j}\right)}{\left(\sum x^{2}{}_{k,i}+\sum x^{2}{}_{k,j}\right)-\sum\left(x_{k,i}*x_{k,j}\right)}\right]$$

The community types identified from the cluster analysis were further refined in a synoptic table where species occurrences were summarized as synoptic cover-abundance values (van der Maarel *et al*., 1987). Synoptic value is the average cover abundance values of a species. Dominant species of each community type were identified based on their synoptic values and community types were named after one or more dominant and/or characteristic species.

3.3.1.1 Diversity analysis

Species richness is an appropriate measure of alpha (α) diversity and is usually expressed as a number of species per sample unit (Whittaker, 1972). However, Magurran (2004) stated that community diversity consists of two components. These are the species richness and the relative abundance (evenness or equitability) of species within a sample or community. The Shannon-Wiener diversity index is the most widely used measure of community diversity because it combines species richness with species evenness (relative abundance) (Magurran, 1988; Kent and Coker, 1992). The Shannon diversity index (H') was calculated using the following equation:

$$\mathbf{H' = -\sum_{i=1}^{s} Pi\ ln\ Pi}$$

Where, **H’ =** Shannon diversity index, **s =** the number of species, **Pi** = the proportion of individuals of the i^{th} species expressed as a proportion of total cover in the sample, and **ln** = the natural logarithm. The values of the Shannon diversity index usually found to fall between 1.5 and 3.5 and only rarely exceed 4.5 (Kent and Coker, 1992).

The Shannon evenness index (J) was also calculated using the following equation:

$$\mathbf{J = \frac{H'}{H'max} = \frac{H'}{ln\ s}}$$

Where J = Shannon equitability or evenness index, H’ = Shannon-Wiener diversity index, H’ max = the maximum level of diversity possible within a given population, which equals In s, and s = the number of species, In = the natural logarithm. The values of J ranges normally between 0 and 1, where 1 representing a situation in which all species are equally abundant (Magurran, 1988).

3.3.1.2 Comparison of species composition among community types

Sorensen similarity index was used to assess the degree of floristic similarity between plant communities and Bore-Anferara-Wadera forest with other eight Afromontane forests in Ethiopia. It is preferred because it gives weight to species that are common to the quadrats or sample plots rather than to those that only occur in either sample plots (Kent & Coker, 1992). The following equation was used to calculate Sorensen similarity index:

$$Ss = \frac{2a}{2a+b+c}$$

Where Ss = Sorensen similarity coefficient, a = the number of species common to both sites, b = the number of species present in one of the sites to be compared and c = the number of species present on the other site. Often the coefficient is multiplied by 100 to give a percentage similarity figure.

Species composition analysis was complemented by calculating the floristic dissimilarity between all pairs of sites using beta (β) diversity. Beta diversity measures the change in the diversity of species among a set of habitats. In simpler terms, it calculates the number of species that are not the same in two different habitats. Beta diversity is, therefore, the rate of change of community along an ecological gradient. There are different ways to calculate beta diversity where all methods determine species turnover (replace one another) between different sites or along environmental gradients (Perlman and Adelson, 1997). Noticing the problem associated with the increasing number of species with increasing sample size, Whittaker (1972) suggested that beta diversity could better be calculated from pair-wise comparison of sites. In the present study, pair-wise comparison of beta diversity between community types was computed using the formula (Whittaker, 1972),

$$\beta = \frac{b+c}{2a+b+c}$$

Where a, is the number of shared species in two sites, and b and c are the numbers of species unique to each site. A higher value of beta diversity index indicates a low level of similarity, while a lower value of beta diversity index shows a high level of similarity.

3.3.1.3 Species accumulation curve

The species accumulation curve, which is, the plot of the expected number of detected species as a function of sampling effort, arises as a graphical representation of the sampling process (Palmer, 1990). Species accumulation is the accumulation of species when the number of site increases. The species accumulation curve in adequately sampled study area levels off before the total number of sampling plots is reached (Zerihun Woldu, 2016). It may also be used to estimate the expected number of new species that may be encountered for given additional sampling effort (Zerihun Woldu, 2016). In this study, species accumulation curve was plotted to examine the degree of species collection (sampling effort).

3.3.2 Soil analysis

Soil samples were taken in all the study plots for soil analysis. Two layers of soil samples were taken from each sample plot, first layer 0-25 cm and second layer 25-50 cm. These soil samples were taken from five points measuring 15 x 15 cm^2, one at the center and the other four at each corner, of each sample plot (Eyob Tenkir, 2006; Dereje Denu, 2006;

Iwara *et al*., 2011). Similar layers from these five points were mixed to form a composite sample in order to reduce variability within the sample plots. One kg of a soil sample from each layer was taken for organic carbon, soil texture (clay, silt and sand particles), soil pH and Cation Exchange Capacity (CEC) analysis. The samples were air-dried and soil analyses were made at Holeta and Debre Birhan agricultural research center soil laboratories following the standard procedures outlined in ISRIC (2002). Thus, the pH was measured using a pH meter at 1:2.5 soil- distilled water suspension. The texture was determined on the basis of Hydrometer method with the categories sand, silt, and clay (expressed as % weight). Cation Exchange Capacity (CEC) and organic carbon were determined by Ammonium acetate (NH_4OAc) and Walkley and Black methods, respectively.

3.3.3 Ordination

In the present study, Canonical Correspondence Analysis (CCA) is applied as a direct ordination method using the vegetation data and environmental data collected. CCA (ter Braak, 1995) was chosen as the most appropriate method for assessing the relationship between vegetation and environmental data.

CCA is best defined as a method of direct ordination with the resulting ordination being a product of the variability of both the environmental and species data. The resulting ordination diagram expresses not only patterns of variation in floristic composition but also demonstrates the principal relationships between the species and each of the environmental variables (Kent, 2012). Before the application of CCA, environmental variables which were relatively more important in explaining the species data were isolated by computing variance inflation factor (vif), which can help to eliminate those environmental variables that are collinear (variables significantly correlated with each other and add little in explaining the variability of the species data) and leave only the variables that contain unique information. Variables having vif values higher than 5 are collinear and are thus candidates for elimination (Zerihun Woldu, 2016). Variance inflation factor (vif) analysis was conducted using the free statistical software R version 3.1.1 (R Development Core Team, 2014). In this analysis, 112 sample plots, 136 species and 10 environmental variables (Altitude, Slope, Aspect, Disturbance, pH, Organic carbon (OC), Cation exchange capacity (CEC), Clay, Silt and Sand particles) were included.

3.3.4 Vegetation structure analysis

Analysis of population structure is an extremely useful tool for planning management activities and assessing the impact of resource extraction (Peters, 1996). Vegetation structure analysis of the four community types was based on frequency, density, DBH, height, and basal area per hectare, following Mueller-Dombois & Ellenberg (1974). The frequency of a species was computed as the proportion of sample plots within which a species is found. Density was then computed by converting the count from all sample plots into a hectare basis as indicated in Kent & Coker (1992). Diameter at breast height (DBH) was calculated from circumference of each adult woody species using the formula

$$\mathbf{D = \frac{C}{\pi}}$$

Where D = diameter at breast height, C = circumference and π = a constant value of 3.14. Both DBH and height were classified into various DBH and height classes, and the density distribution of woody species in each class was computed (Kent and Coker, 1992). The ratio of the density of individuals with DBH > 10 cm and DBH > 20 cm was computed to measure the size class distribution of species in the community types, following Grubb *et al*. (1963). The relative density of species in different DBH classes was used to get representative patterns of species population structures, following Popma *et al*. (1988). Basal area (BA) (in m^2 per hectare) of trees was computed using the following formula to measure dominance, where the term dominance refers to the degree of coverage of a species as an expression of the space it occupies (Barbour *et al*., 1987).

$$\mathbf{BA} = \left(\frac{\mathbf{d}}{\mathbf{200}}\right)^{\mathbf{2}} \times \boldsymbol{\pi}$$

Where BA = Basal area in m^2 per hectare, d = diameter at breast height in cm and π = 3.14. Importance value index (IVI) of each tree species was computed as indicated in Mueller-Dombois and Ellenberg (1974) based on the formula: IVI = Relative density (RD) + Relative frequency (RF) + Relative dominance (RDO) where,

$$\textbf{Relative density (RD)} = \left(\frac{\textbf{Number of individuals of a tree species}}{\textbf{Total number of all tree species}}\right) \times \mathbf{100}$$

$$\textbf{Relative frequency (RF)} = \left(\frac{\textbf{Frequency of a tree species}}{\textbf{Total frequency of all tree species}}\right) \times \mathbf{100}$$

$$\textbf{Relative dominance (RDO)} = \left(\frac{\textbf{Dominance of a tree species}}{\textbf{Dominance of all tree species}}\right) \times \mathbf{100}$$

The individuals of a species with higher IVI value are dominant over individuals of species with relatively lower IVI values. The maximum IVI value of a species is 300.

Floristic similarity analysis between Bore-Anferara-Wadera forest and eight other Afromontane forests of Ethiopia were computed using; Sorensen's similarity coefficient

$$Ss = \frac{2a}{2a+b+c}$$

Where Ss = Sorensen similarity coefficient, a = the number of species common to both sites, b = the number of species present in one of the sites to be compared and c = the number of species present on the other site (Kent and Coker, 1992).

3.3.5 Land use/Land cover change data analysis

Image pre-processing and enhancement

A blend of steps and procedures were involved in analyzing, mapping, interpret and quantify the available data sets (Figure 5). The data were analyzed by using ArcGIS software version 9.3 and clipped with the boundary of the study area for further processing. Image enhancement was carried out to improve the visual interpretability of an image by increasing the apparent distinction between features in the scene (Lillesand and Kiefer, 2000). If an image is enhanced the distinct features are clearer so that image analysis, classification, and interpretation are better. In addition, image enhancement is used to increase the details of the image by assigning the image maximum and minimum brightness values to maximum and minimum display values, it is done on pixel values, and this makes visual interpretation easier and assists the human analyst. The original low dynamic range of the image is stretched to full dynamic range which is from 0 to 256 by using histogram equalization. Moreover, spatial enhancement of convolution of kernel 5 by 5 of high pass filtering has been done on the images of the respective years.

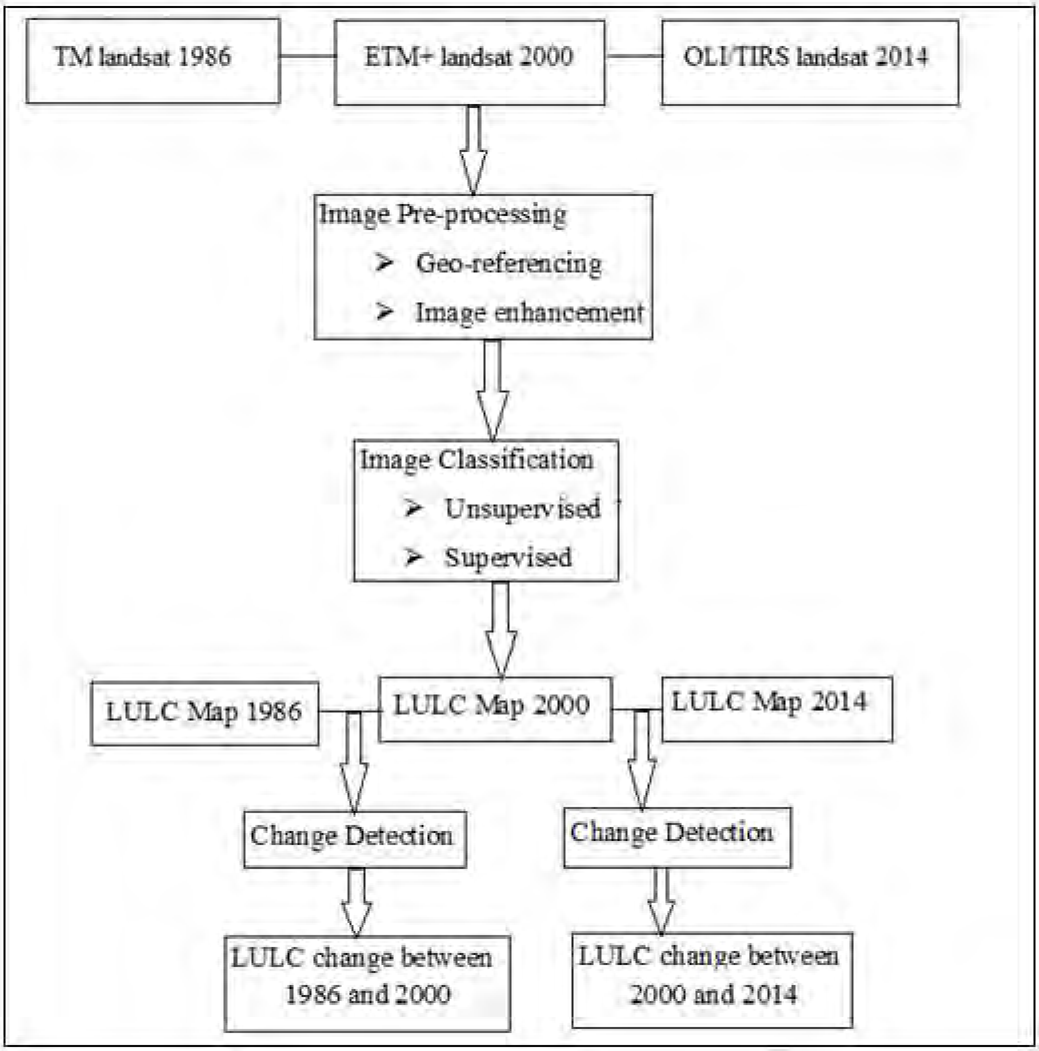


Figure 5 Flow chart showing the general methodology of land use/land cover assessment

Image classification

Digital image classification is the process of assigning the pixel to classes. By comparing pixel to one another and to pixels of known identity, it is possible to assemble groups of similar pixels into classes that match to the informational categories of interest to users of remotely sensed data (Walelign Salem, 2007). Lillesand and Kiefer (2000) stated that there are two ways of digital image classification, namely, supervised and unsupervised. Supervised classification is a method in which the analyst defines small training sites on the image, which were representative of each desired land cover category whereas unsupervised classification technique is performed when there was little or no knowledge of the geography of the region where classification was undertaken. In this study, both unsupervised and supervised image classification methods were employed. An unsupervised classification was first carried out for identification of the features in a pixel form, and then supervised classification was employed to categorize the images using ground truths (training areas). Based on the training areas, the rest of the study area is classified to the nearest to that training areas. Maximum likelihood classifier algorithm was used as a decision for supervised classification. The coordinates of training areas have been collected from the field with GPS and from a topographic map of the study area which has been digitized. Based on the training areas, the images of the study area have been classified into six general land use/land covers classes. These are Forest land (F), Agricultural land (AL), Grazing land (GL), Shrubland (SL), Wetland (WL) and Built up area (BUA) (Table 2). A combination of information collected from the field and the satellite image were effectively used in the preparation of the legend. Identification of some of the land use/land cover classes required a number of field visits and discussions with elder farmers, to have not only a clear understanding of the main land use and land cover types but also to establish what types of changes are expected over time. Categorization of LULC types resulted in the production of the LULC legend, the establishment of its characteristics, and identification and mapping of LULC types.

Table 2 Description of land use/land cover types identified in the study area

LULC classes	**General description**
Forest land	Land with a tree crown cover of more than 10% and an area of more than 0.5 hectares; the trees should be able to reach a minimum height of 5 m at maturity *in situ*.
Agricultural land	Areas allotted for rainfed and irrigated crop production (both annual and perennial) mostly of cereals in subsistence farming, including areas currently under crop, fallow land and land under preparation.
Grazing land	It consists of land areas where grasses are the predominant natural vegetation. It also includes grasses mixed with scattered trees and it is used for grazing and browsing.
Shrubland	It comprises of land area covered by small trees, bushes and shrubs, in some cases mixed with grasses; less dense than forests.
Wetland	It represents areas of marsh, bog, peatlands
Built up area	The land surface which is mainly covered by bare soil and exposed rocks at the surface of the earth and land covered by structures which include urban towns and rural villages.

Source: FAO, 2001; Mohammed Assen, 2011; Tegegne Sishaw and Aklilu Amsalu, 2012

Accuracy assessment

Land use and land cover maps derived from remote sensing always contain some sort of errors due to several factors, which range from classification technique to method of satellite data capture. For proper use of the land cover maps which are derived from remote sensing and the accompanying land resource statistics, the errors must be quantitatively evaluated in terms of classification accuracy and intended to produce information that describes reality. Therefore, an accuracy assessment was carried out to verify to what extent the produced classification is compatible with what actually exists on the ground (Congalton, 1991). The process of accuracy assessment involves the production of references (samples) that evaluate the produced classification. These references were identified from Google Earth and Global Positioning System (GPS) points from field work, which was independent of the ground truths used in the classification scheme. Thus, total accuracy and Kappa statistics were computed from this error matrix.

Rate of land use/land cover change

The change detection matrices were determined by overlaying two land use and land cover maps at a time in ERDAS IMAGINE 2010 image processing software. The areas which were converted from each of the classes to any of the other classes were computed (Gautam *et al*., 2003). The land cover map for the three period series of images was analyzed based on land use/ land cover types area comparison and land use/ land cover changes using tables and graphs. The changes over 28 years were analyzed and rate of change for each land use/ land cover type for the two periods from 1986 – 2000 and 2000 – 2014 was calculated using the following formula:

$$\mathbf{Rate\ of\ change\ (ha/year) = \ (A - B)/C}$$

Where, A = Recent area of the land use and land cover in ha, B = Previous area of the land use and land cover in ha and C = Time interval between A and B in years. It should be noted that the negative values indicate the magnitude of the decline in that particular land use and land cover type.

CHAPTER FOUR

4 RESULTS

4.1 Species accumulation curve

In ecological communities, the number of species increases with increasing area sampled. However, the rate of newly added species decreases with increasing sampling effort. This trend of species accumulation in the sampling effort could be represented graphically using species accumulation curve. The graph was plotted for the cumulative number of species recorded as a function of sampling effort. Species accumulation curve helps to illustrate the rate at which new species are included as the sampling effort proceed. One of its applications is to assess whether a study area has been sufficiently sampled or not. A curve labels off (or approximately reaching an asymptote) indicating that no or few species would be collected if sampling effort is further continued. Accordingly, as it is indicated in figure 6, the curve almost leveled off showing that a few species would be collected if sampling effort is further continued. This could prove that representative samples have been taken in the study area.

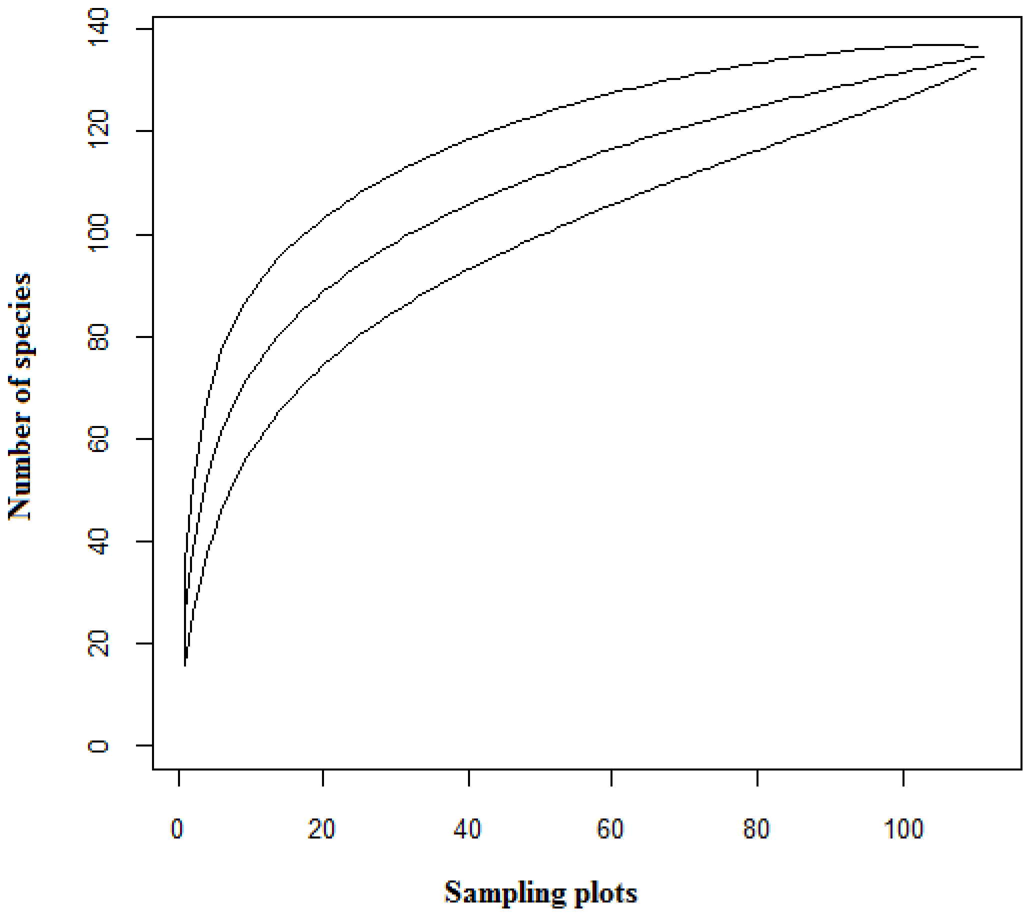


Figure 6 Species accumulation curve for Bore-Anferara-Wadera forest

4.2 Floristic composition

The current study yielded a total of 136 species of vascular plants representing 119 genera and 63 families in the study area (Appendix 1). All of these plant species were recorded from the sampling plots laid for the study. This total includes 5 pteridophyte families, 1 gymnosperm family and 57 angiosperm (6 monocotyledons and 51 dicotyledons) families. The total number of species for each group is indicated in Table 3.

Table 3 Total number of families and species for each group

	Pteridophyte	**Gymnosperm**	**Angiosperm**	
			Monocotyledons	Dicotyledons
Families	5	1	6	51
Species	5	1	8	122

Seventy-four percent of the families were represented by more than one species. The highest number of species was recorded for the families Fabaceae (11 species, 8.09%) and Euphorbiaceae (9 species, 6.62%) followed by Celastraceae, Oleaceae, Rubiaceae (7 species, 5.15% each), Rutaceae (6 species, 4.41%) and Asteraceae (5 species, 3.68%). Seven families, i.e., Acanthaceae, Apocynaceae, Flacourtiaceae, Myrsinaceae, Poaceae, Rhamnaceae and Rosaceae were represented by three species (2.21%) each. Fourteen (20.59%) of the families were represented by two species, whereas the remaining 35 (25.74%) families only had single-species representation. Seven species-rich families contributed to 38.25% of the total plant species in the study area (Figure 7). Species like *Lippia adoensis*, *Maytenus addat*, *Millettia ferruginea*, *Kotschya recurvifolia*, *Vepris dainellii* and *Vernonia rueppellii*, which are endemic to Ethiopia, were recorded in the study area. These endemic plant species accounted for 4.41% of the total species recorded. In terms of plant habit, shrubs were the dominant growth form represented by 56 species (41.18%) followed by trees (44 species, 32.35%), lianas (18 species, 13.24%) and herbs (18 species, 13.24%).

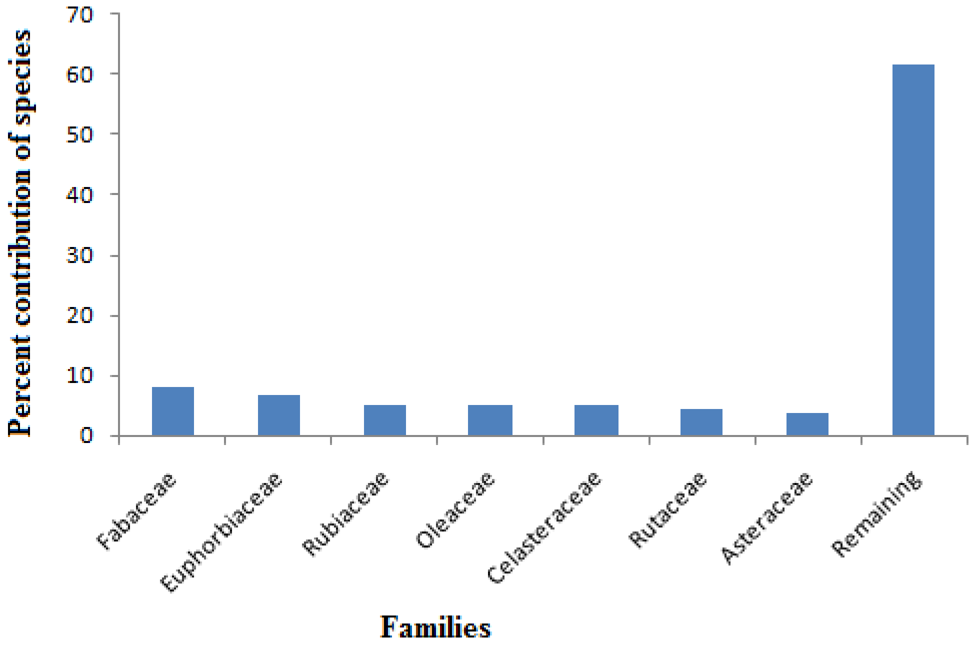


Figure 7 Percent species contribution of the families in decreasing order

In this study, sixteen (11.8%) new records were made for the Sidamo floristic region of the flora area. These were six trees, seven shrubs, one liana and three herbs (Table 4).

Table 4 New records for Sidamo floristic region in the FEE

No	**Species name**	**Family**	**Habit**
1	*Agarista salicifolia*	Ericaceae	Tree
2	*Albizia gummifera*	Fabaceae	Tree
3	*Celtis toka*	Ulmaceae	Tree
4	*Dracaena afromontana*	Dracaenaceae	Tree
5	*Elaeodendron buchananii*	Celastraceae	Tree
6	*Erythrococca trichogyne*	Euphorbiaceae	Shrub
7	*Halleria lucida*	Scrophulariaceae	Shrub
8	*Hippocratea pallens*	Celastraceae	Liana
9	*Ilex mitis*	Aquifoliaceae	Tree
10	*Poecilostachys oplismenoides*	Poaceae	Herb
11	*Rosa abyssinica*	Rosaceae	Shrub
12	*Solanum anguivi*	Solanaceae	Shrub
13	*Teclea nobilis*	Rutaceae	Shrub
14	*Thalictrum rhynchocarpum*	Ranunculaceae	Herb
15	*Vernonia unionis*	Asteraceae	Herb
16	*Zanthoxylum chalybeum*	Rutaceae	Shrub

4.3 Plant community types

Four plant community types were identified from the hierarchical cluster analysis (Figure 8). The analysis was based on the abundance data of the species on the study plots. The data matrix contained 112 plots and 136 vascular plant species. Two or more species with the highest synoptic values in the group were used to name the corresponding communities (Table 5). The relationship of the forests with the plant community types was also indicated in Figure 9.

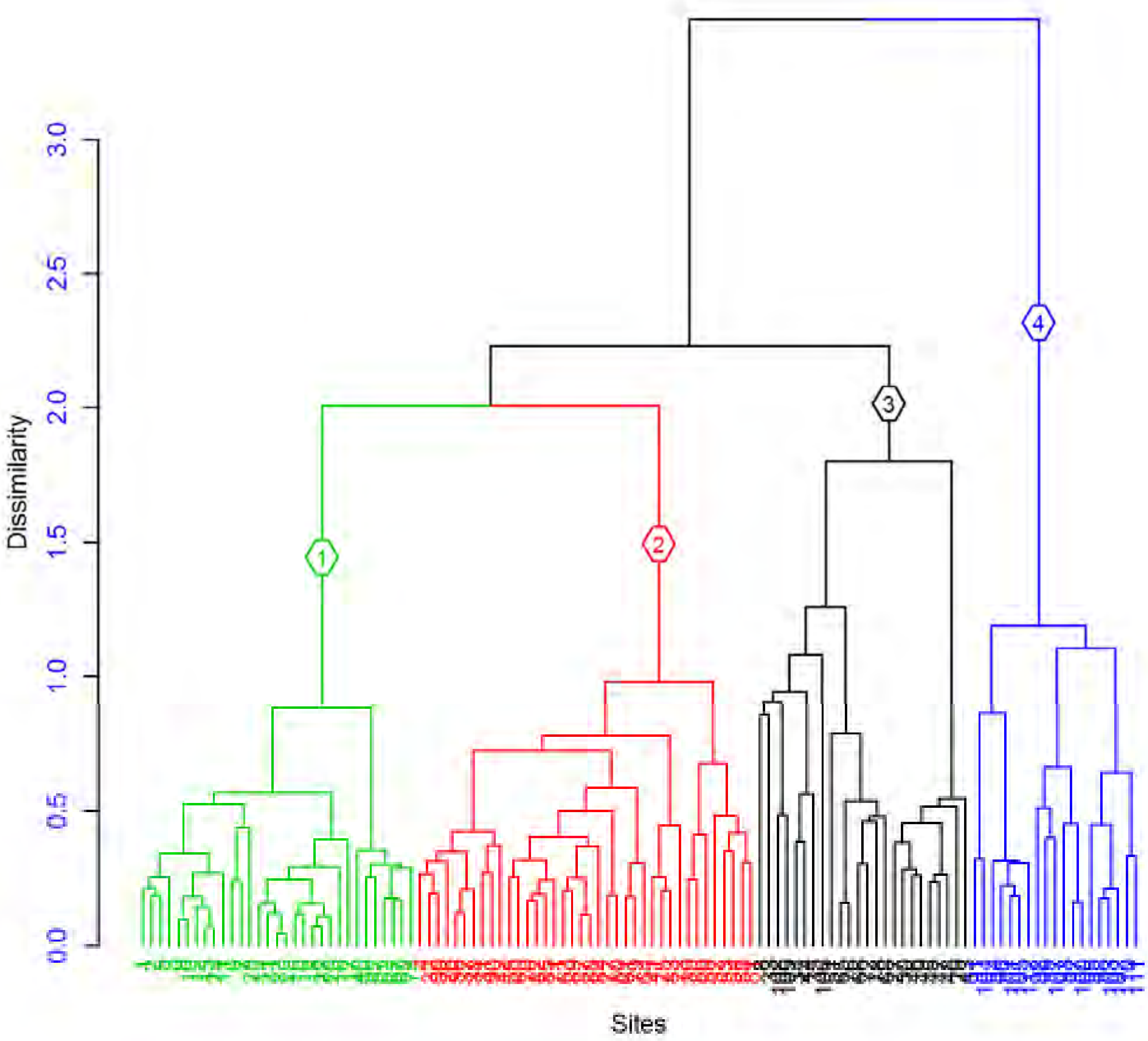


Figure 8 Dendrogram obtained from hierarchical cluster analysis of species abundance data of Bore-Anferara-Wadera Forest.

The grouping was at 2.0 dissimilarity level (1 = Community type 1; 2 = Community type 2; 3 = Community type 3 and 4 = Community type 4).

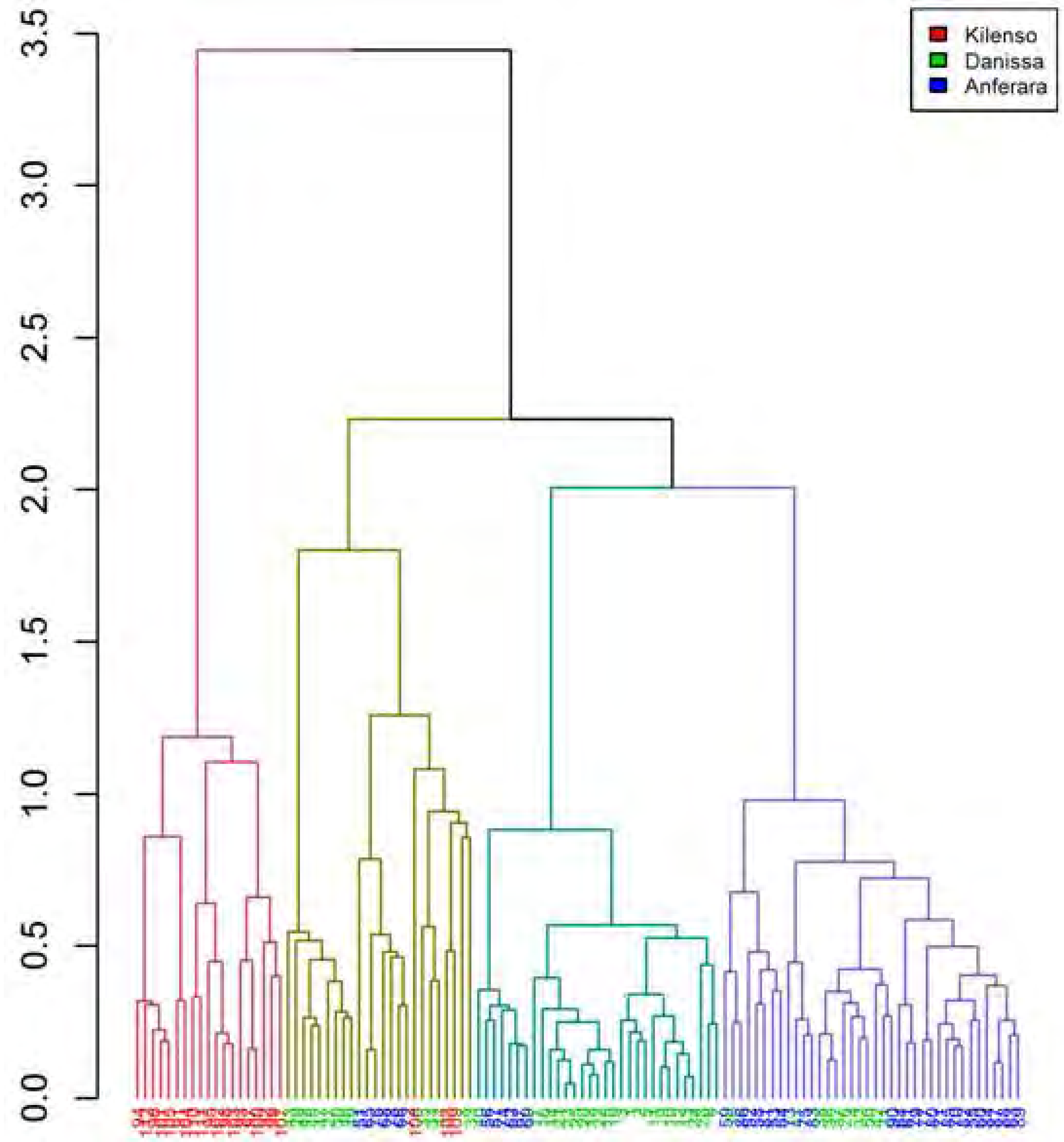


Figure 9 Dendrogram showing the relationship between the forests and the community types

Table 5 Synoptic cover abundance values of species reaching a value of ≥ 0.5 in at least one community type in Bore-Anferara-Wadera Forest. Values in bold refer to species used to name community types.

Community	**1**	**2**	**3**	**4**
Cluster size	**31**	**38**	**24**	**19**
Maytenus addat	0.5	0.1	0.0	0.0
Ficus sur	0.8	0.7	0.2	0.0
Acanthus eminens	**5.5**	3.1	1.8	0.0
Dracaena afromontana	**3.6**	2.2	2.1	0.0
Macaranga capensis	1.7	1.8	0.5	0.0
Isoglossa somalensis	2.4	0.1	0.8	2.2
Lepidotrichilia volkensii	2.0	1.4	1.2	0.0
Poecilostachys oplismenoides	2.2	2.3	2.1	1.6
Prunus africana	1.1	0.9	0.5	0.2
Chionanthus mildbraedii	0.9	0.4	0.9	0.1
Allophylus abyssinicus	1.0	0.8	0.6	0.1
Croton macrostachyus	0.7	0.7	1.2	0.6
Hippocratea goetzei	0.1	0.5	0.3	0.2
Syzygium guineense subsp. *afromontanum*	3.3	**3.2**	2.0	0.3
Ocotea kenyensis	3.0	**2.6**	1.3	0.0
Galiniera saxifraga	0.5	0.6	0.3	0.0
Fagaropsis angolensis	0.3	0.3	0.8	1.0
Ehretia cymosa	0.3	0.1	1.0	0.6
Nuxia congesta	0.5	0.2	0.7	0.1
Apodytes dimidiata	0.7	0.7	0.4	0.3
Polyscias fulva	0.6	0.6	0.8	0.4
Bersama abyssinica	0.9	1.1	0.8	0.5
Trichilia emetica	0.5	0.2	0.3	0.3

Embelia schimperi	0.2	0.5	0.2	0.0
Psydrax schimperiana	1.5	1.6	0.8	1.5
Ilex mitis	0.2	0.6	0.2	0.0
Maytenus arbutifolia	0.8	1.4	0.8	0.6
Desmodium repandum	0.2	0.7	0.4	0.0
Pouteria adolfi-friederici	2.4	1.6	**2.5**	0.4
Psychotria orophila	1.7	2.2	**2.3**	1.0
Olea capensis subsp. *macrocarpa*	1.2	1.6	1.0	1.6
Albizia gummifera	0.2	0.0	0.5	0.1
Elaeodendron buchananii	1.0	1.0	1.1	1.4
Vepris dainellii	0.2	1.2	1.6	0.4
Podocarpus falcatus	0.5	1.0	1.0	2.0
Rytigynia neglecta	0.2	0.1	0.4	1.1
Millettia ferruginea	0.1	0.0	0.9	0.2
Landolphia buchananii	0.0	0.3	0.3	0.8
Celtis africana	0.2	0.1	0.5	1.4
Ochna holstii	0.0	0.3	0.1	0.8
Achyranthes aspera	0.0	0.5	0.5	1.8
Erythrococca trichogyne	0.1	0.0	0.5	0.7
Hypoestes forskaolii	0.5	1.1	2.0	1.2
Carissa spinarum	0.0	0.0	0.0	0.6
Scolopia theifolia	0.0	0.0	0.0	**3.1**
Teclea nobilis	2.4	2.2	1.9	**2.7**
Hippocratea africana	0.0	0.1	0.2	0.7
Premna schimperi	0.0	0.0	0.1	0.5
Calpurnia aurea	0.0	0.0	0.2	1.2
Mimusops kummel	0.0	0.0	0.1	1.0
Acokanthera schimperi	0.0	0.0	0.0	1.1
Bridelia micrantha	0.0	0.0	0.0	0.8

The four community types were named after two dominant species. Thus, community type 1 was designated as *Acanthus eminens - Dracaena afromontana;* community type 2 as *Syzygium guineense* subsp. *afromontanum - Ocotea kenyensis;* community type 3 as *Pouteria adolfi-friederici - Psychotria orophila* and community type 4 as *Scolopia theifolia - Teclea nobilis.* The community types varied in size, ranging from 19 - 38 plots (Table 5). The major characteristic of each community type is summarized as follows:

Community 1: *Acanthus eminens - Dracaena afromontana* community type

Altitudinal distribution of this community ranges from 1840 to 2304 m a.s.l. The community was distributed on a moderate slope (average 14.9%) facing largely towards west (22.58%) and northwest (19.35%) direction with a low rate of disturbance. The community was composed of 87 plant species distributed in 31 plots, of which 25 plots were from Anferara forest and the remaining 11 plots were from Kilenso forest and hence this community type can be named as “Anferara forest” (hereafter Anferara forest). The dominant tree and shrub species in this community include *Acanthus eminens*, *Dracaena afromontana*, *Croton macrostachyus*, *Trichilia emetica*, *Maytenus arbutifolia*, *Elaeodendron buchananii*, *Syzygium guineense* subsp. *afromontanum*, *Nuxia congesta*, *Bersama abyssinica*, *Olea capensis* subsp. *macrocarpa*, *Psychotria orophila*, *Ocotea kenyensis*, *Polyscias fulva*, *Maytenus addat*, *Macaranga capensis*, *Ficus sur*, *Galiniera saxifraga*, *Apodytes dimidiata*, *Lepidotrichilia volkensii*, *Pouteria adolfi-friederici*, *Prunus africana*, *Chionanthus mildbraedii*, *Allophylus abyssinicus*, *Teclea nobilis* and *Psydrax schimperiana*. *Isoglossa somalensis*, *Poecilostachys oplismenoides*, and *Hypoestes forskaolii* were found to be common in the herbaceous layer of this community. In this community, there were no dominant liana species covering large area compared with other community types.

Community 2: *Syzygium guineense* subsp. *afromontanum* - *Ocotea kenyensis* community type

This community type was found between 1835 and 2285 m a.s.l. The community was found on a moderate slope (average 15%) facing towards east (34.21%), north (15.79%) and south (15.79%) directions and most of the plots were less disturbed. The community was comprised of 69 plant species distributed in 38 plots. Of the total plots, 27 plots were from Kilenso forest and 11 plots from Anferara forest and hence this community type can also be named as "Kilenso forest" (hereafter Kilenso forest). The tree and shrub layer of this community type was dominated by plant species such as *Syzygium guineense* subsp. *afromontanum*, *Ocotea kenyensis*, *Maytenus arbutifolia*, *Croton macrostachyus*, *Psychotria orophila*, *Macaranga capensis*, *Elaeodendron buchananii*, *Vepris dainellii*, *Bersama abyssinica*, *Galiniera saxifraga*, *Olea capensis* subsp. *macrocarpa*, *Pouteria adolfi-friederici*, *Acanthus eminens*, *Ficus sur*, *Podocarpus falcatus*, *Apodytes dimidiata*, *Prunus africana*, *Lepidotrichilia volkensii*, *Allophylus abyssinicus*, *Dracaena afromontana*, *Teclea nobilis*, *Psydrax schimperiana*, *Ilex mitis* and *Polyscias fulva*. The dominant liana species of the community were *Hippocratea goetzei* and *Embelia schimperi*. The herbaceous layer was mainly dominated by *Poecilostachys oplismenoides*, *Achyranthes aspera*, *Hypoestes forskaolii* and *Desmodium repandum*.

Community 3: *Pouteria adolfi-friederici - Psychotria orophila* community type

This community type was distributed in the altitudinal range between 1828 and 2264 m a.s.l. The community was dispersed in a strong slope (average 19%) facing largely towards west (36%) and east (20%) directions with the relatively low rate of disturbance. The community was represented by 24 plots. Of these, 14 plots were from Anferara forest, 7 plots were from Kilenso forest and 3 plots from Danissa forest and hence it is named as "Mixed forest" (hereafter Mixed forest). Ninety-eight plant species were associated with this community type. *Pouteria adolfi-friederici*, *Psychotria orophila*, *Acanthus eminens*, *Croton macrostachyus*, *Millettia ferruginea*, *Maytenus arbutifolia*, *Teclea nobilis*, *Elaeodendron buchananii*, *Nuxia congesta*, *Olea capensis* subsp. *macrocarpa*, *Polyscias fulva*, *Syzygium guineense* subsp. *afromontanum*, *Bersama abyssinica*, *Ocotea kenyensis*, *Vepris dainellii*, *Erythrococca trichogyne*, *Podocarpus falcatus*, *Dracaena afromontana*, *Fagaropsis angolensis*, *Ehretia cymosa*, *Lepidotrichilia volkensii*, *Chionanthus mildbraedii*, *Albizia gummifera* and *Celtis africana* were found to be the dominant tree and shrub species of the community. No dominant liana species were found in this community. The herb layer of this community was dominated by *Hypoestes forskaolii*, *Poecilostachys oplismenoides*, and *Achyranthes aspera*.

Community 4: *Scolopia theifolia - Teclea nobilis* community type

The *Scolopia theifolia - Teclea nobilis* community type was found to be distributed between 1869 and 1954 m a.s.l. The community was found on a strong slope (average 22%) facing largely towards east (50%) direction. Most of the plots were relatively less disturbed. The community consisted of 74 species scattered in 19 plots, which were all from Danissa forest and hence this community can be named as “Danissa forest” (hereafter Danissa forest). The dominant tree and shrub species of the community include *Scolopia theifolia*, *Teclea nobilis*, *Psychotria orophila*, *Premna schimperi*, *Erythrococca trichogyne*, *Rytigynia neglecta*, *Calpurnia aurea*, *Fagaropsis angolensis*, *Celtis africana*, *Ehretia cymosa*, *Acokanthera schimperi*, *Croton macrostachyus*, *Elaeodendron buchananii*, *Mimusops kummel*, *Olea capensis* subsp. *macrocarpa*, *Podocarpus falcatus*, *Ochna holstii*, *Psydrax schimperiana*, *Bridelia micrantha* and *Carissa spinarum*. *Landolphia buchananii* and *Hippocratea africana* were found to be the dominant liana species in this community. The herb layer was composed of species such as *Achyranthes aspera*, *Poecilostachys oplismenoides*, *Isoglossa somalensis* and *Hypoestes forskaolii*.

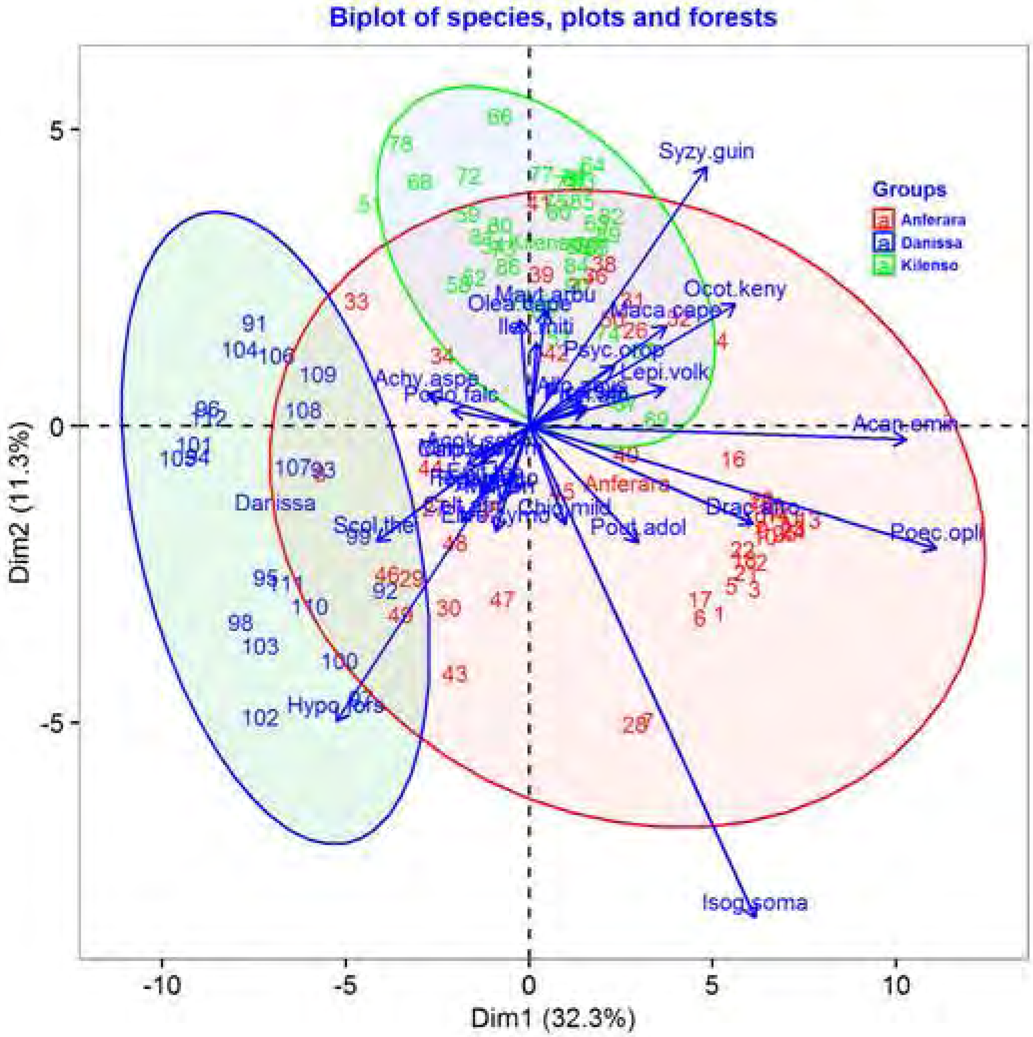


Figure 10 Biplot of the plots, species and the forests

The biplot complements the synoptic table in that those species that are most significantly related to the forest are also some of those species that are characteristic of community types (Figure 10).

4.3.1 Comparison of community diversity among plant community types

The overall Shannon – Wiener diversity and evenness values of Bore-Anferara-Wadera forest were 3.84 and 0.78, respectively (Table 6). The four plant community types showed variation in their species richness, diversity, and evenness. The mixed forest had the highest species richness, diversity, and evenness, while Anferara forest was found to have the least diversity and evenness but the second highest species richness. Danissa forest showed the highest evenness and the second highest diversity and the third highest species richness. On the other hand, Kilenso forest exhibited the least species richness but the second highest evenness and the third highest diversity.

Table 6 Species richness, diversity and evenness values of plant communities identified in Bore-Anferara-Wadera forest

Community	**Elevation**	**Species richness (S)**	**Shannon diversity (H')**	**Shannon evenness (J)**
Anferara	1840 - 2304	87	3.38	0.76
Kilenso	1835 - 2285	69	3.49	0.82
Mixed	1828 - 2264	98	3.93	0.86
Danissa	1869 - 1954	74	3.72	0.86
Overall	1828 - 2304	136	3.84	0.78

4.3.2 Comparison of species composition among community types

Sorensen's similarity coefficient and beta diversity indices were used to compare floristic composition similarity among the four forest patches in Bore-Anferara-Wadera forest (Table 7). Sorensen's similarity coefficient indicated the highest floristic similarity between Anferara and Kilenso forests (0.76) followed by Mixed and Kilenso forests (0.74) and the lowest between Anferara and Danissa forest (0.53) (Table 7). The least species similarity with any other forest was recorded for Danissa forest. Similarity coefficients of all forest patch range from 0.53 to 0.76. Forest patches with the highest species similarity (Anferara & Kilenso forests) shared 62.2% of the total species while those with least species similarity (Anferara & Danissa forests) shared 37%.

The magnitude of beta diversity indicates the change in species composition between adjacent forest patches along the environmental gradient. Similarity coefficients and beta diversity are inversely related. Therefore, forest patches with the highest similarity coefficients had the least beta diversity (0.23) (Anferara and Kilenso forests) and forest patches with the least similarity coefficients had the highest beta diversity (0.46) (Anferara and Danissa forests) (Table 7).

Table 7 Sorensen's similarity coefficient and beta diversity index in species composition between the four forest patches in Bore-Anferara-Wadera forest. Values in bold indicate Sorensen's coefficient while those in italics indicates beta diversity index.

Forest patches	**Anferara**	**Kilenso**	**Mixed**	**Danissa**
Anferara		*0.23*	*0.27*	*0.46*
Kilenso	**0.76**		*0.25*	*0.45*
Mixed	**0.72**	**0.74**		*0.34*
Danissa	**0.53**	**0.54**	**0.65**	

4.4 Ordination

Canonical correspondence analysis (CCA) was employed to examine the effect of environmental variables on the patterns of variation in floristic composition in Bore-Anferara-Wadera forest. The major objective was not only to display the patterns of variation in floristic composition but also to demonstrate the principal relationships between the species and each of the environmental variables. The environmental variables were selected by computing variance inflation factor (vif) for their significance (Table 8).

Table 8 Results of the variance inflation factor (vif) test of environmental variables (Environmental variables having vif values higher than 5 are less significant).

Variable	**Variance Inflation Factor (vif)**
Altitude	1.673475
Slope	1.486273
Aspect	1.124908
Disturbance	1.094957
pH2	2.201661
OC2	1.504446
CEC2	1.899940
Clay2	1.264821
Silt2	3.201915
Sand2	4.486958

The ordination of sample plots of the four community types with significant environmental variables produced four groups based on their variation in species composition (Figure 11). The CCA result tends to strengthen the cluster analysis result indicating that the two methods are complementary. These four groups were separated following the arrows of the environmental variables and corresponded to the four community types identified earlier. Plots representing community type 1 were located at the lower right side of the ordination diagram, plots representing community type 2 were found at the upper right side, plots representing community type 3 were grouped at the lower left side and plots representing community 4 were located at the upper left side of the ordination diagram (Figure 11).

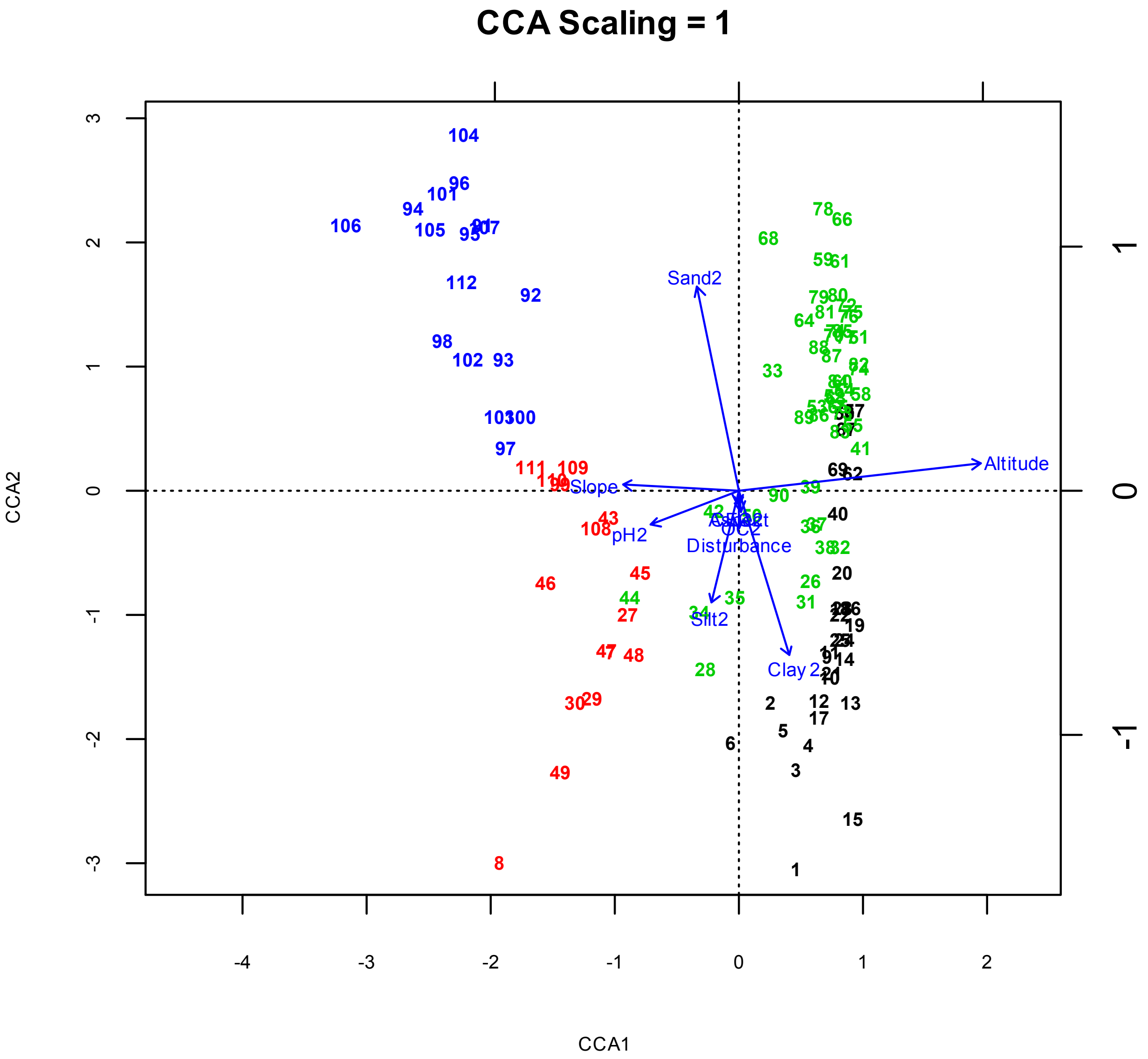


Figure 11 Canonical correspondence analysis (CCA) ordination diagram of the plot-environment biplot

The relationship of environmental variables to the four groups of the community types which were identified in the cluster analysis (Figure 8) is indicated by the length of arrows from the origin (center) of the scatterplot to the points representing the environmental variable. Altitude with the longest arrow was the most important variable in influencing the variation of patterns in species composition. The different environmental variables were correlated to each other differently depending on the angle between the arrows. Altitude and pH, for example, were negatively correlated since the arrows are pointing in opposite direction while altitude and clay soil with an intermediate angle between the arrows were positively correlated.

Table 9 Biplot scores for constraining variables and their correlation with the CCA axes, eigenvalues and proportion of variance explained

Variable	**CCA1**	**CCA2**	**CCA3**	**CCA4**
Altitude	0.988	0.11	0.02	0.05
Slope	-0.474	0.32	-0.23	-0.37
Aspect	0.000	-0.06	-0.32	-0.29
Disturbance	0.000	-0.16	-0.06	-0.02
pH2	-0.357	-0.14	-0.09	0.60
OC2	0.005	-0.08	-0.27	0.17
CEC2	-0.006	-0.05	-0.50	0.22
Clay2	0.206	-0.67	0.44	-0.17
Silt2	-0.110	-0.46	-0.26	0.06
Sand2	-0.169	0.83	-0.28	0.19
Eigenvalue	0.31	0.08	0.03	0.03
Proportion Explained	0.55	0.15	0.06	0.05
Cumulative Percentage	0.55	0.70	0.75	0.80

The environmental variable highly correlated with axis one was altitude followed by slope, pH, clay and sand soil to a lesser extent. Therefore, these variables, altitude, in particular, contributed significantly in explaining variations in the patterns of plant distribution and the formation of the plant community types. The first axis explained about 55% of the variation in the patterns of plant species distribution and community formation, followed by the second axis (15%) and the importance of higher axes in explaining the variation decreased successively.

4.5 Comparison of floristic similarity with other Afromontane forests in Ethiopia

Bore-Anferara-Wadera forest was compared with eight other Afromontane forests in Ethiopia (Table 10). The vegetation data from Bore-Anferara-Wadera was only collected to a maximum of 2304 m a.s.l. whereas other forests go beyond that. However, the comparison was carried out since all the forests compared are Afromontane forests. Bore-Anferara-Wadera forest was floristically more related to Belete (0.47) and Magada (0.41) forests whereas it showed the least similarity with Egdu (0.22), Dodola (0.23) and Hugumburda (0.25) forests.

Table 10 Comparison of floristic similarities between Bore-Anferara-Wadera and eight other Afromontane forests in Ethiopia

(N = Number of species included in comparison; a = number of species common to both forests compared; b = number of species available only in Bore-Anferara-Wadera forest; c = number of species available only in the other forest compared; Ss = Sorensen's similarity coefficient)

Forest	**Data source**	**Elevation (m a.s.l.)**	**N**	**a**	**b**	**c**	**Ss**
Belete	Kitessa Hundera & Tsegaye Gadissa (2008)	1850 - 2250	79	50	86	29	0.47
Egdu	Abiyou Tilahun *et al.* (2011)	2574 - 2948	217	40	96	177	0.22
Dodola	Kitessa Hundera *et al*. (2007)	2500 - 3500	113	29	107	84	0.23
Gurra Farda	Kitessa Hundera & Bishaw Deboch (2008)	800 - 1900	66	37	99	29	0.37
Magada	Genene Bekele & Reddy (2015)	1750 - 2100	197	69	67	128	0.41
Mana Angetu	Ermias Lulekal *et al*. (2008)	1533 - 2431	211	69	67	142	0.40
Masha	Abreham Assefa *et al*. (2013)	1700 - 3000	58	36	100	22	0.37
Hugumburda	Ermias Aynekulu (2011)	1860 - 2700	79	27	109	52	0.25
Present study		1828 - 2304	136	136	0	0	1

4.6 Vegetation structure

4.6.1 Density of trees and shrubs

The total density of trees and shrubs with DBH greater than 2 cm was 1,047 individuals ha^{-1} in Bore-Anferara-Wadera forest. *Dracaena afromontana* accounted for 13.70% of the total density of woody species > 2 cm DBH followed by *Teclea nobilis* (11.24%), *Psychotria orophila* (7.97%), *Ocotea kenyensis* (6.83%), *Syzygium guineense* subsp. *afromontanum* (6.05%) and *Pouteria adolfi-friederici* (5.43%). These six woody species accounted for 51.23% of the total density. On the other hand, the remaining 85 woody species altogether accounted for 48.77% of the total density in the forest (Table 11). The density of trees and shrubs with DBH > 10 cm was 500.30 individuals ha^{-1} whereas that of species with DBH > 20 cm was 246.63 individuals ha^{-1} (Appendix 2). The ratio of the density of trees and shrubs with DBH > 10 cm to DBH > 20 cm is taken as a measure of size class distribution. Accordingly, the ratio of individuals with DBH > 10 cm to DBH > 20 cm was 2.03.

Table 11 Density and percentage contribution of six woody species in Bore-Anferara-Wadera forest

Species name	**Density**	**%**
Dracaena afromontana	143.55	13.70
Teclea nobilis	117.76	11.24
Psychotria orophila	83.53	7.97
Ocotea kenyensis	71.53	6.83
Syzygium guineense subsp. *afromontanum*	63.39	6.05
Pouteria adolfi-friederici	56.85	5.43
Other 85 species	510.91	48.77
Total	1047.52	100

4.6.2 Diameter at Breast Height (DBH) class distribution

The size class distribution of trees and shrubs in Bore-Anferara-Wadrea forest across seven DBH classes indicated a relatively large number of individuals in the first DBH class (542.66 ha^{-1}, 51.86%) and second DBH class (257.94 ha^{-1}, 24.65%). As the DBH class size increases, the number of individuals gradually decreases towards the successive higher DBH classes (Figure 12). Though the distribution of individuals of species along the higher DBH classes decreased, distribution continued up to the highest DBH class (DBH class 7). *Pouteria adolfi-friederici*, *Syzygium guineense* subsp. *afromontanum*, *Ficus sur*, *Macaranga capensis*, *Ocotea kenyensis*, *Olea capensis* subsp. *macrocarpa*, *Prunus africana*, *Podocarpus falcatus*, *Polyscias fulva* and *Trichilia emetica* were found to be the dominant large-sized trees in Bore-Anferara-Wadrea forest, with DBH > 110 cm. The overall contribution of individuals of species to the total DBH was highly variable where *Dracaena afromontana* (13.71%) contributed the highest proportion followed by *Teclea nobilis* (11.25%) and *Psychotria orophila* (7.98%), whereas *Acacia abyssinica*, *Combretum molle* and *Euclea racemosa* (each 0.01%) contributed least to the total DBH (Appendix 3).

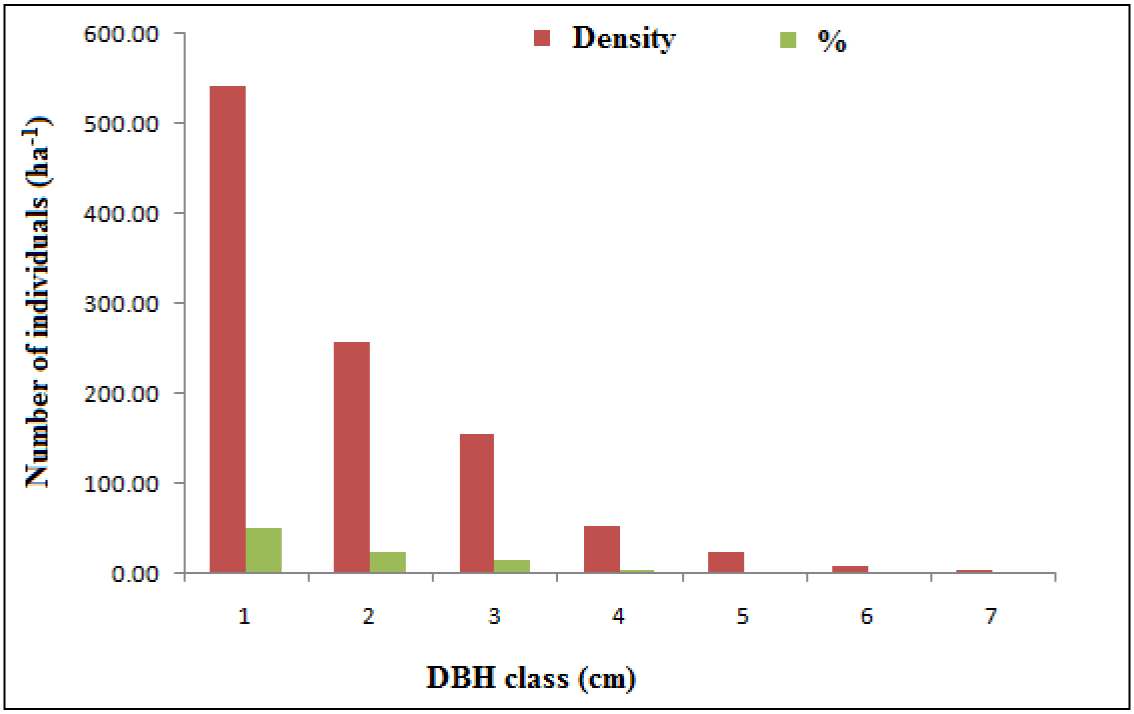


Figure 12 DBH class distributions of trees and shrubs in Bore-Anferara-Wadera forest

DBH class 1 = 2.1 – 10 cm; 2 = 10.1 – 20 cm; 3 = 20.1 – 50 cm; 4 = 50.1 – 80 cm; 5 = 80.1 – 110 cm; 6 = 110.1 – 140 cm; 7 = > 140 cm

4.6.3 Height class distribution

The distribution of trees and shrubs in different height classes in Bore-Anferara-Wadera forest is given in Figure 13. The patterns of height class distribution of trees and shrubs revealed a high proportion of individuals in the first class followed by a rapid decline across the successive higher classes. Large proportions of individuals were aggregated in the first three height classes. About 67.07% of trees and shrubs were less than 13 m tall (Height classes 1 and 2). Only a small proportion, about 17.3%, reached a height of 23 m and above indicating the predominance of shorter plants (Figure 13). Trees and shrubs that contributed higher proportions of individuals to the lowest height classes (Height classes 1-3) include *Dracaena afromontana* (13.59%), *Teclea nobilis* (11.27%) and *Psychotria orophila* (8.04%) (Appendix 4).

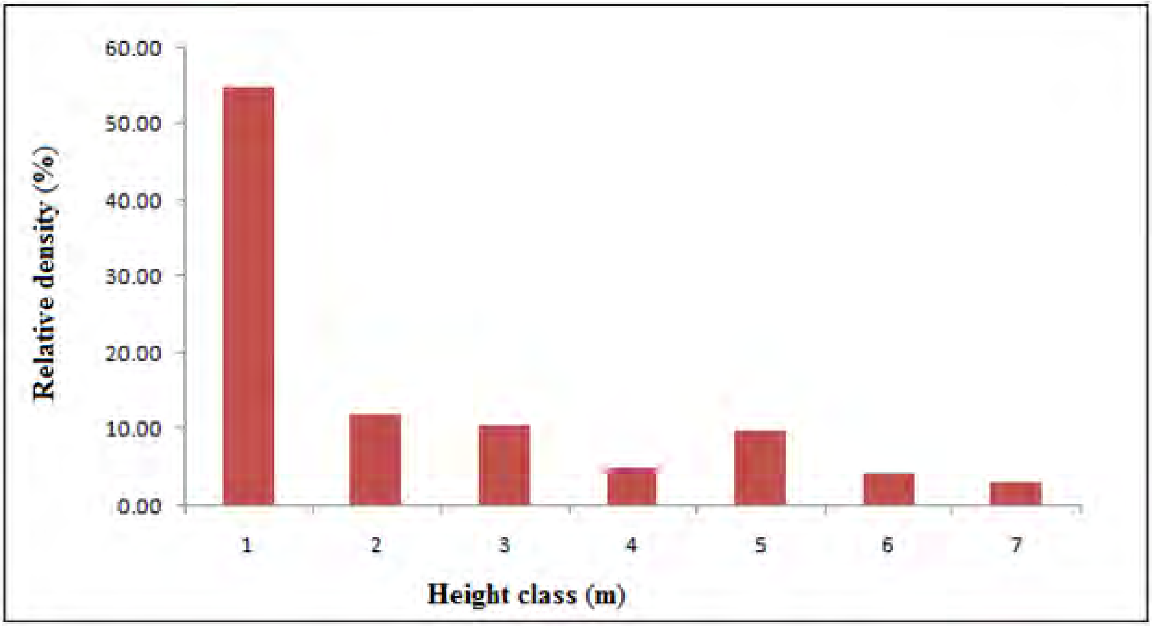


Figure 13 Relative density of trees and shrubs distributed along Height classes in Bore-Anferara-Wadera forest
Height class 1 = 3 – 8 m; 2 = 8.1 – 13 m; 3 = 13.1 – 18 m; 4 = 18.1 – 23 m; 5 = 23.1 – 28 m; 6 = 28.1 – 33 m; 7 = > 33 m

4.6.4 Vertical structure

The vertical structure of the woody species was analyzed using the International Union for Forestry Research Organization (IUFRO) classification scheme (Lamprecht, 1989). The IUFRO classification scheme classifies storey into upper, where the tree height is greater than 2/3 of the top height; middle, where the tree height is in between 1/3 and 2/3 of the top height and lower where the tree height is less than 1/3 of the top height. Density, species number, and ratio of individuals to species in the lower, middle and upper storey in Bore-Anferara-Wadera forest are given in Table 12.

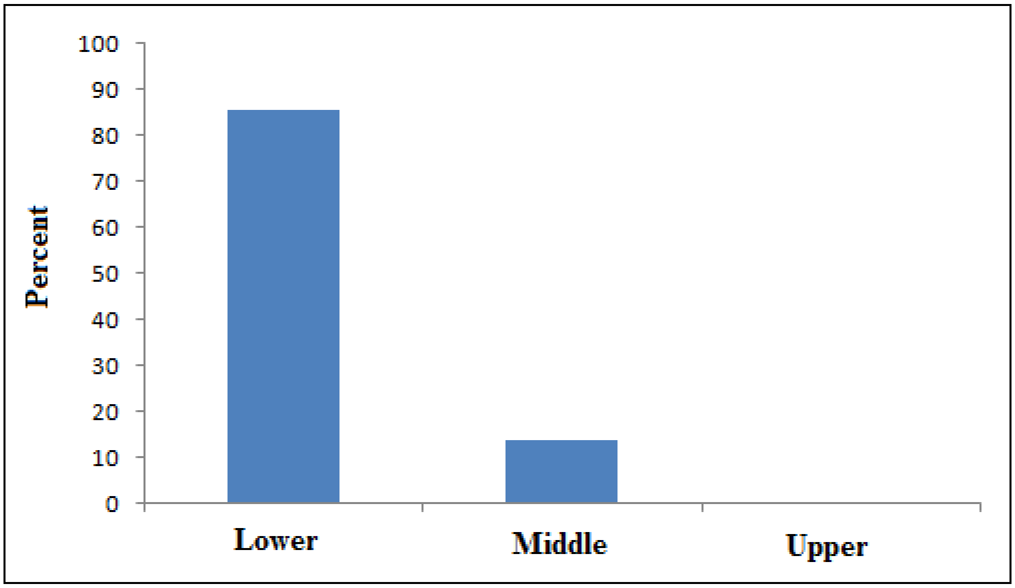


Figure 14 Percent density of trees in lower, middle and upper storey

Table 12 Density, species number, and ratio of individuals to species in the lower, middle and upper storey of Bore-Anferara-Wadera forest

Storey	**Height (m)**	**Stem**		**Species**		**Ratio of Individuals to species**
		Density	**%**	**No.**	**%**	
Lower	3.0 - 25	902.78	85.83	89	66.92	10.1:1
Middle	25.1 - 50	145.34	13.82	35	26.32	4.2:1
Upper	> 50	3.77	0.36	9	6.77	0.4:1

The top height for trees in Bore-Anferara-Wadera forest was 75 m. Tree species that occupied the upper storey in Bore-Anferara-Wadera forest include *Pouteria adolfi-friederici*, *Syzygium guineense* subsp. *afromontanum*, *Polyscias fulva*, *Ocotea kenyensis*, *Prunus africana*, *Ficus sur*, *Croton macrostachyus* and *Olea capensis* subsp.

macrocarpa. A small number of individuals occupied the upper storey as the ratio of individuals to species is lower (Figure 14 & Table 12). The middle layer of Bore-Anferara-Wadera forest was dominated by species like *Macaranga capensis*, *Psydrax schimperiana*, *Olea welwitschii*, *Podocarpus falcatus*, *Millettia ferruginea*, *Fagaropsis angolensis*, *Allophylus abyssinicus*, *Trichilia emetica*, *Celtis africana*, *Elaeodendron buchananii*, *Hagenia abyssinica*, *Teclea nobilis and Apodytes dimidiata*. The lower storey was largely dominated by shrubs and small trees such as *Maytenus addat*, *Nuxia congesta*, *Chionanthus mildbraedii*, *Galiniera saxifraga*, *Lepidotrichilia volkensii*, *Ehretia cymosa*, *Oncoba spinosa*, *Celtis toka*, *Psychotria orophila*, *Flacourtia indica*, *Maesa lanceolata*, *Senna didymobotrya*, *Dombeya torrid*, *Ficus thonningii*, *Coffea arabica*, *Erythrococca trichogyne*. The highest proportion of species was concentrated in the lower storey (66.92%) followed by the middle (26.32%) and upper storey (6.77%) (Table 12).

4.6.5 Basal area (BA)

The total basal area of Bore-Anferara-Wadera forest was 75.23 m^2 ha^{-1} (Appendix 5). About 54.37 (72.31%) of the total basal area was contributed by five large-sized tree species, i.e., *Syzygium guineense* subsp. *afromontanum*, *Pouteria adolfi-friederici*, *Ocotea kenyensis*, *Olea capensis* subsp. *macrocarpa* and *Macaranga capensis* (Table 13).

Table 13 Basal area and percent contribution of the five tree species in Bore-Anferara-Wadera forest

Species name	**Basal area (m^2 ha^{-1})**	**%**
Syzygium guineense subsp. *afromontanum*	23.73	31.55
Pouteria adolfi-friederici	14.70	19.55
Ocotea kenyensis	7.50	9.98
Olea capensis subsp. *macrocarpa*	4.98	6.63
Macaranga capensis	3.46	4.60
Total	**54.37**	**72.31**

The contribution of each DBH class to the total basal area is presented in Figure 15 and Table 14. About 51.70% of all individuals had DBH less than 10 cm (DBH class 1). The percentage contribution of this class to the total basal area, however, was only 2.13% (Table 14). On the other hand, individuals in the DBH classes > 50 cm had a density of about 8.88% of the total, but they accounted for about 74.6% of the total basal area of the forest (Table 14).

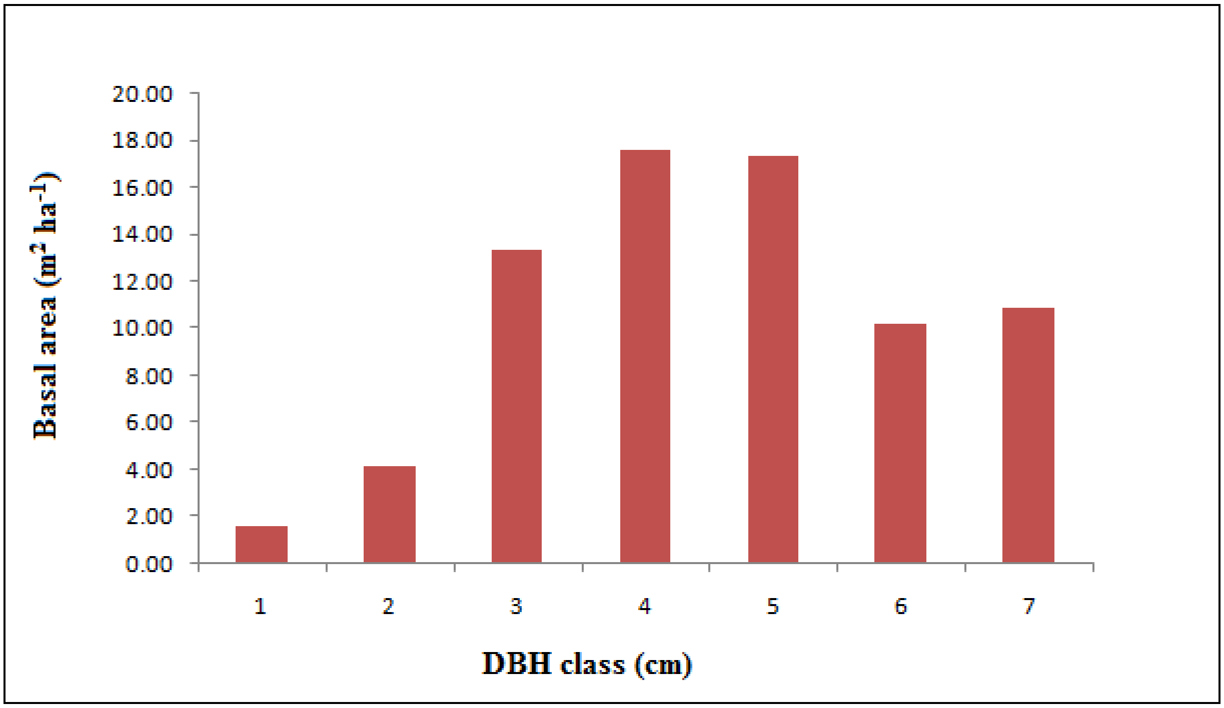


Figure 15 Basal area distributions along DBH classes of Bore-Anferara-Wadera forest.

DBH class 1 = 2.1 – 10 cm; 2 = 10.1 – 20 cm; 3 = 20.1 – 50 cm; 4 = 50.1 – 80 cm; 5 = 80.1 – 110 cm; 6 = 110.1 – 140 cm; 7 = > 140 cm.

Table 14 Contribution of different DBH classes to the total density and basal area per hectare in Bore-Anferara-Wadera forest

DBH class	**Density**		**Basal area**	
	Stem/ha	**%**	**m^2/ha**	**%**
1	541.62	51.70	1.60	2.13
2	254.51	24.30	4.14	5.50
3	158.33	15.12	13.36	17.76
4	54.86	5.24	17.62	23.42
5	24.60	2.35	17.39	23.12
6	8.73	0.83	10.22	13.59
7	4.86	0.46	10.89	14.47
	1047.52	100.00	75.23	100.00

4.6.6 Frequency

Teclea nobilis was found to be the most frequent species in Bore-Anferara-Wadera forest occurring in 99% of all plots sampled followed by *Psychotria orophila* (97.32%), *Psydrax schimperiana* (90.18%), *Elaeodendron buchananii* (86.61%) and *Syzygium guineense* subsp. *afromontanum* (80.36%). The least frequent species include, among others, *Dodonea angustifolia*, *Phyllanthus ovalifolius*, *Protea gaguedi*, *Ricinus communis*, *Ritchiea albersii*, *Schrebera alata*, *Terminalia schimperiana* each constituting 0.89% (Appendix 6). The frequency distribution of trees and shrubs in Bore-Anferara-Wadera forest showed that higher proportions of species were found to be distributed in the first frequency class followed by a rapid decline in distribution of species across successive higher frequency classes (Figure 16).

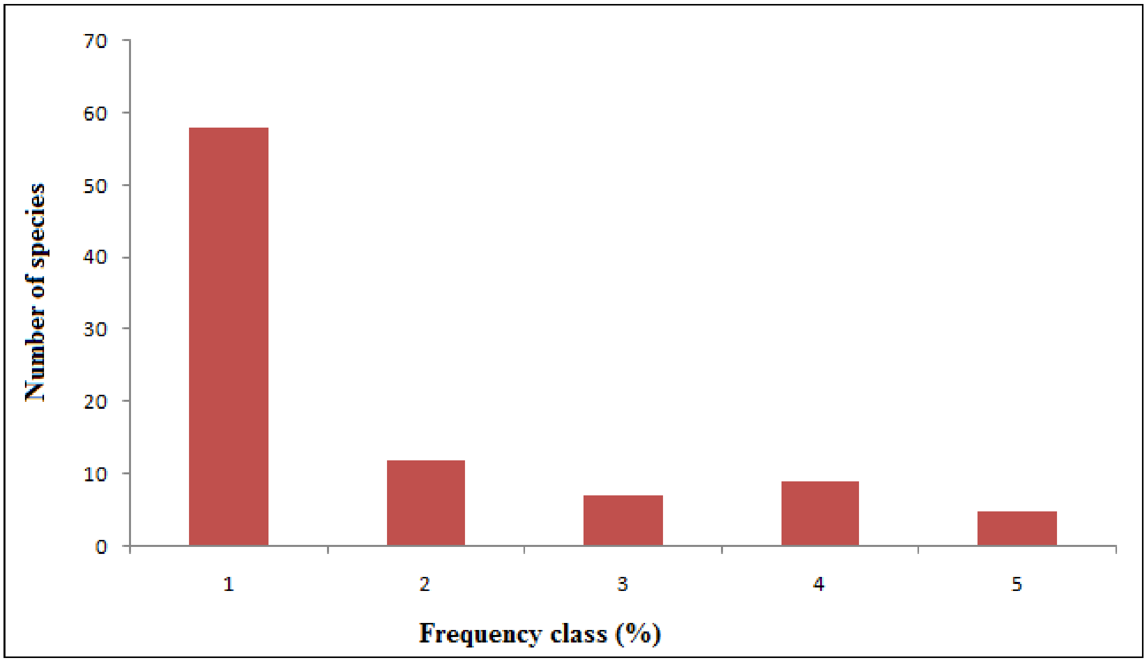


Figure 16 Frequency distributions of trees and shrubs in Bore-Anferara-Wadera forest. Frequency class 1 = 0 – 20; 2 = 20.1 – 40; 3 = 40.1 – 60; 4 = 60.1 – 80 and 5 = 80.1 – 100

4.6.7 Importance Value Index (IVI)

In Bore-Anferara-Wadera forest, the highest IVI value was recorded for *Syzygium guineense* subsp. *afromontanum* (41.27) followed by *Pouteria adolfi-friederici* (28.60), *Dracaena afromontana* (20.66), *Ocotea kenyensis* (20.15), *Teclea nobilis* (16.81), *Olea capensis* subsp. *macrocarpa* (13.10), *Psychotria orophila* (12.87) and *Macaranga capensis* (11.26). These eight species contributed about 54.91% of the total importance values whereas the remaining 83 species had combined IVI values of 45.09% (Appendix 7).

4.7 Population structure

The patterns of diameter class distribution indicate the general trends of population dynamics and recruitment process of a species. Analysis of the population structure of 30 common tree and shrub species revealed three general patterns in Bore-Anferara-Wadera forest (Figure 17a-c).

The first pattern was an inverted J-shaped distribution formed by species with high frequency distribution of individuals in the lower DBH classes followed by a gradual decrease towards the higher DBH classes (Figure 17a). This pattern of distribution was represented by *Pouteria adolfi-friederici*, *Dracaena afromontana*, *Ocotea kenyensis*, *Psydrax schimperiana*, *Millettia ferruginea*, *Celtis africana*, *Apodytes dimidiata*, *Vepris dainellii*, *Elaeodendron buchananii*, *Lepidotrichilia volkensii*, *Fagaropsis angolensis*, *Scolopia theifolia*, *Ehretia cymosa*, *Psychotria orophila* and *Nuxia congesta*. The second pattern was a bell-shaped distribution formed by species where the frequency distribution of individuals in the lower and higher DBH classes was lower than the middle classes (Figure 17b). Species such as *Syzygium guineense* subsp. *afromontanum*, *Olea capensis* subsp. *macrocarpa*, *Macaranga capensis*, *Allophylus abyssinicus*, and *Polyscias fulva* were characterized by this distribution pattern in the forest. The third pattern was formed by species having an irregular distribution over DBH classes where some DBH classes had a small number of individuals while other DBH classes had a large number of individuals (Figure 17c). This pattern of distribution was represented by *Trichilia emetica*, *Prunus africana*, *Ficus sur*, *Podocarpus falcatus* and *Croton macrostachyus* in the forest.

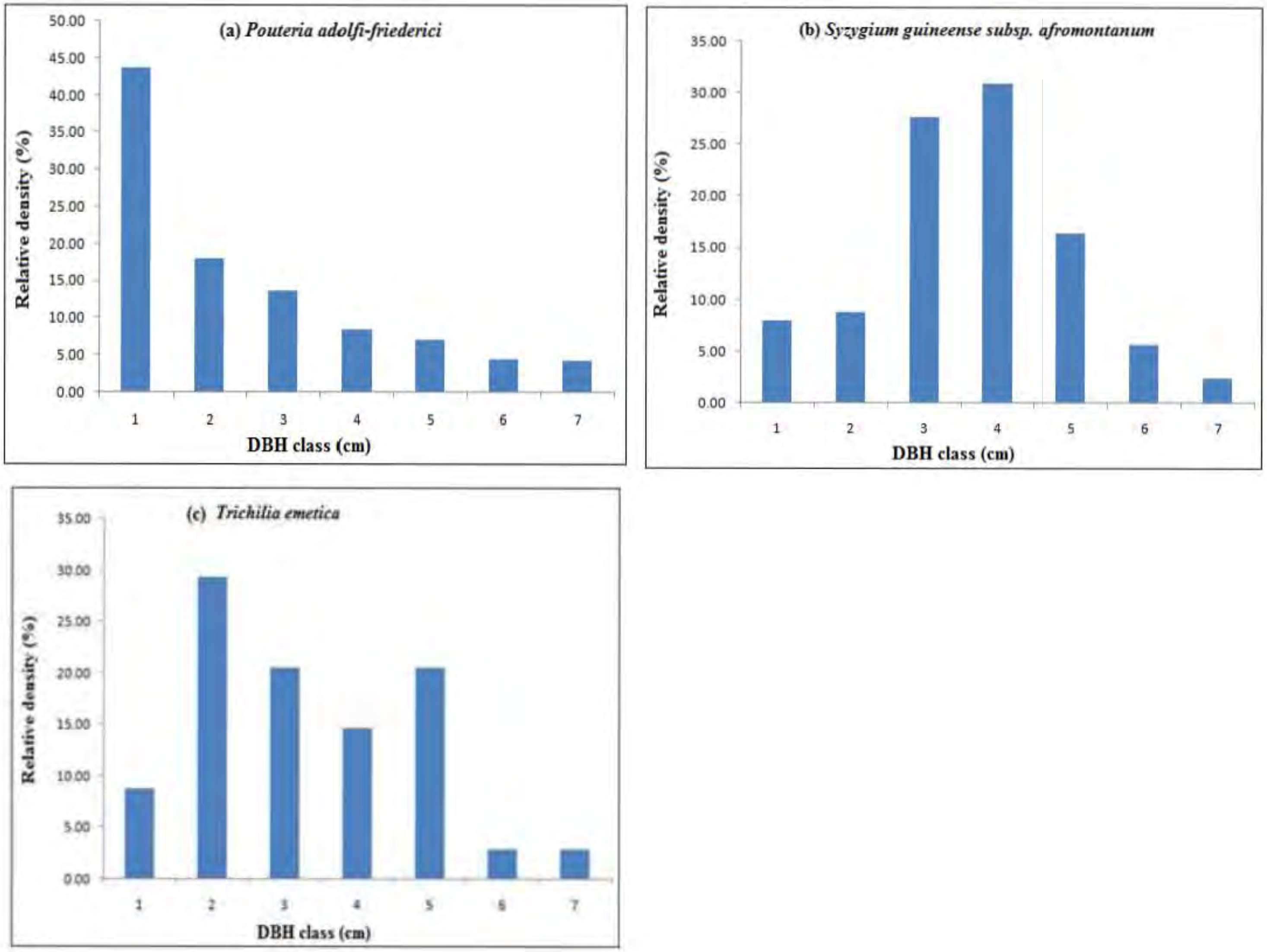


Figure 17a-c Representative patterns of species population structures in Bore-Anferara-Wadera forest.

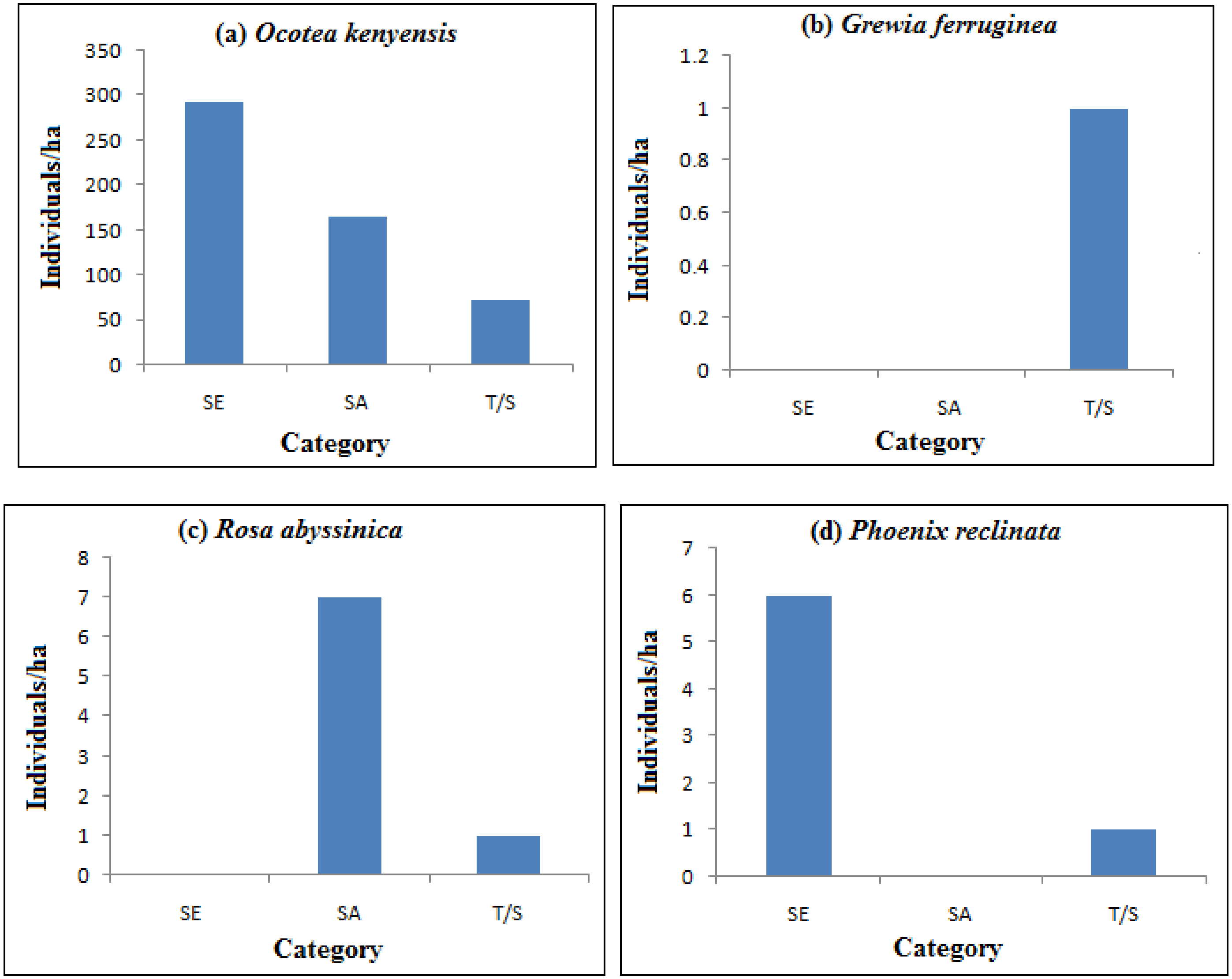
4.8 Regeneration status of Bore-Anferara-Wadera forest

The composition and density of seedlings and saplings are indicators of future regeneration status of any forest. The distribution of seedlings, saplings, and mature trees/shrubs indicates four general patterns (Figure 18a-f). The first type include species represented by all three stages, the second type include species represented by mature trees/shrubs only, the third type contains species represented by either sapling and mature trees/shrubs or seedlings and mature trees/shrubs and the fourth type include species represented by either seedlings only or seedlings and saplings. The density of seedlings, saplings, and mature tree/shrub species in Bore-Anferara-Wadera forest is given in Appendix 8.

Seven species (10.61%) were not represented by both seedlings and saplings (Figure 18 b) and only a small number of mature trees/shrubs were recorded for these species. On the other hand, two species (3.03%) and three species (4.55%) had no seedlings and saplings (Figure 18c & d), respectively. Moreover, two species (3.03%) were not represented by saplings and mature trees/shrubs (Figure 18e) and three species (4.55%) had no mature trees/shrubs (Figure 18f). According to these results, tree/shrub species were grouped into three priority classes for conservation activities. These are Class 1 species with no seedlings and saplings, Class 2 species with the absence of either seedlings or saplings, Class 3 species with no saplings and/or mature trees/shrubs (Table 15).

Table 15 Classification of tree species in the different conservation priority classes

Priority class 1	**Priority class 2**	**Priority class 3**
Acacia bussei	*Cordia africana*	*Flacourtia indica*
Celtis toka	*Phoenix reclinata*	*Lobelia giberroa*
Ficus thonningii	*Rosa abyssinica*	*Rhamnus prinoides*
Grewia ferruginea	*Pterolobium stellatum*	*Toddalia asiatica*
Pavetta oliveriana	*Vernonia rueppellii*	*Zanthoxylum usambarense*
Schefflera abyssinica		
Terminalia schimperiana		


(a) *Ocotea kenyensis*
Individuals/ha
350
300
250
200
150
100
50
0
SE
SA
T/S
Category
(b) *Grewia ferruginea*
Individuals/ha
1.2
1
0.8
0.6
0.4
0.2
0
SE
SA
T/S
Category
(c) *Rosa abyssinica*
Individuals/ha
8
7
6
5
4
3
2
1
0
SE
SA
T/S
Category
(d) *Phoenix reclinata*
Individuals/ha
7
6
5
4
3
2
1
0
SE
SA
T/S
Category

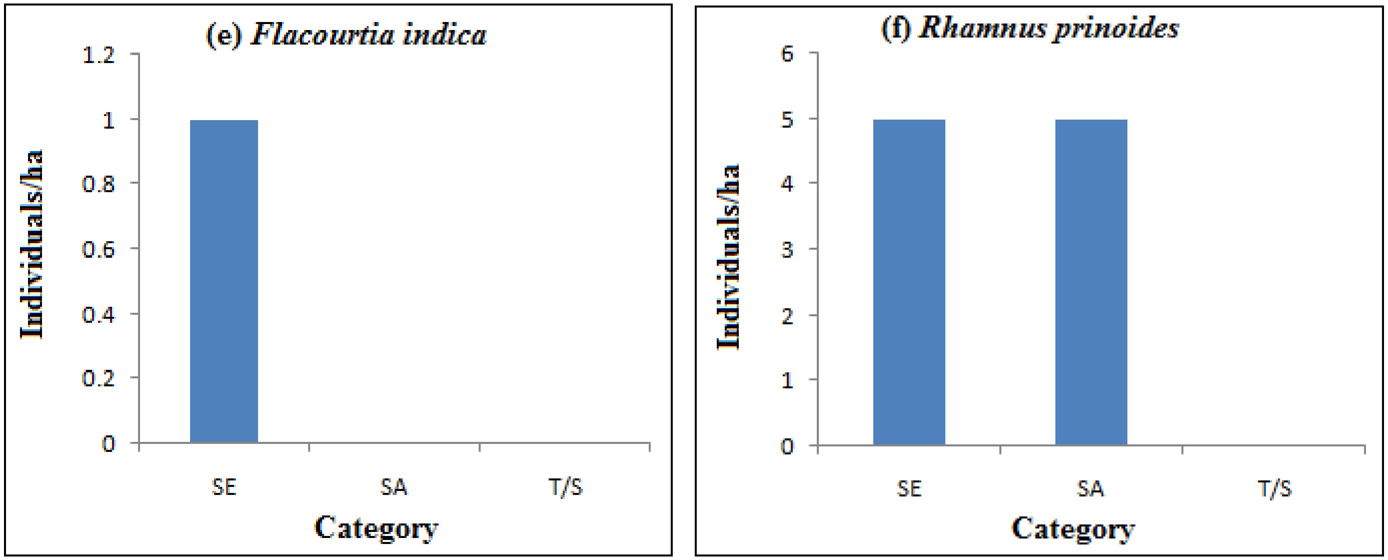


Figure 18a-f Seedling (SE), Sapling (SA) and Tree/shrub (T/S) distributions of some selected species in Bore-Anferara-Wadera forest.

4.9 Land use/land cover dynamics

4.9.1 Land use/land cover type change for 1986, 2000 and 2014

Most of the forest and shrub land areas existed in the 1986 and 2000 maps were reduced in the 2014 map (Table 16 & Figure 19-21). On the contrary, the latter map showed the predominance of agricultural land, built up area and wetland. In general, a continuous change was taking place for most LULC types over the whole study period (1986 – 2014).

In 1986, the dominant land use/land cover classes were shrubland and forest land which covered an area of 133,535.40 ha (38.36%) and 122,524.87 ha (35.19%), respectively. The least coverage was wetland accounted for 13,126.28 ha (3.77%). Agricultural land, built up area and grazing land covered an area of about 43,686.54 ha (12.55%), 14,424.48 ha (4.14%) and 20,850.57 ha (5.99%), respectively during this period (Table 16). Of all LULC classes, 79.54% of the study area was covered by green vegetation such as forest land, grazing land and shrubland while the remaining 20.46% was covered by agricultural land, built up area and wetland in 1986 (Table 16). During 2000 the dominant LULC classes were still shrubland and forest land covering an area of 105,408.62 ha (30.28%) and 101,063.03 ha (29.03%), respectively. The grazing land had the least area coverage of about 22,875.61 ha (6.57%). Built up area, agricultural land and wetland covered an area of 33,242.53 ha (9.55%), 59,082.32 ha (16.97%) and 26,476.03 ha (7.60%), respectively. From overall LULC classes, 65.88% of the study area was covered by green vegetation such as forest land, grazing land and shrubland while the remaining 34.12%

was covered by agricultural land, built up area and wetland in 2000 (Table 16). In 2014, the dominant LULC class was found to be agricultural land covering an area of 95,155.31 ha (27.33%) followed by forest land 69,851.12 ha (20.06%) and shrubland 68,610.22 ha (19.71%). Built up area and wetland covered an area of about 55,753.56 ha (16.01%) and 35,291.5 ha (10.14%), respectively. Grazing land (23,486.43 ha, 6.75%) still occupied the least area compared with other land cover types in this period. About 46.52% of the study area was covered by green vegetation such as forest land, grazing land and shrubland while the remaining 53.48% was covered by agricultural land, built up area and wetland during this period (Table 16). Over the whole period, forest land and shrubland were decreasing whereas agricultural land and built up area were increasing.

Table 16 Area of LULC types during 1986, 2000 and 2014

LULC type	**1986**		**2000**		**2014**	
	Area (ha)	**%**	**Area (ha)**	**%**	**Area (ha)**	**%**
Agricultural land	43686.54	12.55	59082.32	16.97	95155.31	27.33
Built up area	14424.48	4.14	33242.53	9.55	55753.56	16.01
Forest	122524.87	35.19	101063.03	29.03	69851.12	20.06
Grazing land	20850.57	5.99	22875.61	6.57	23486.43	6.75
Shrub land	133535.40	38.36	105408.62	30.28	68610.22	19.71
Wetland	13126.28	3.77	26476.03	7.60	35291.5	10.14
Total	348148.14	100.00	348148.14	100.00	348148.14	100.00

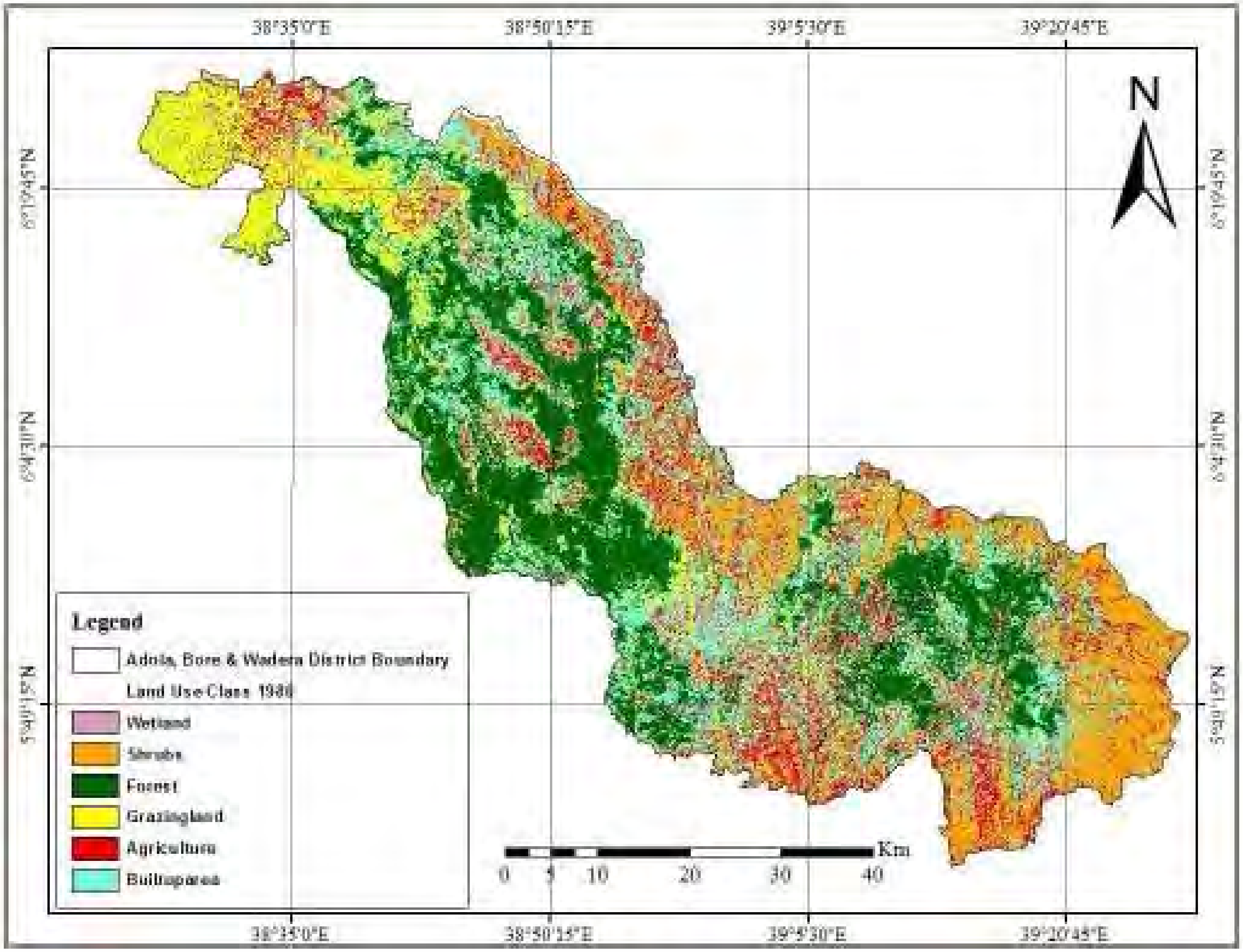


Figure 19 Land use/land cover map of Bore-Anferara-Wadera forest (1986).

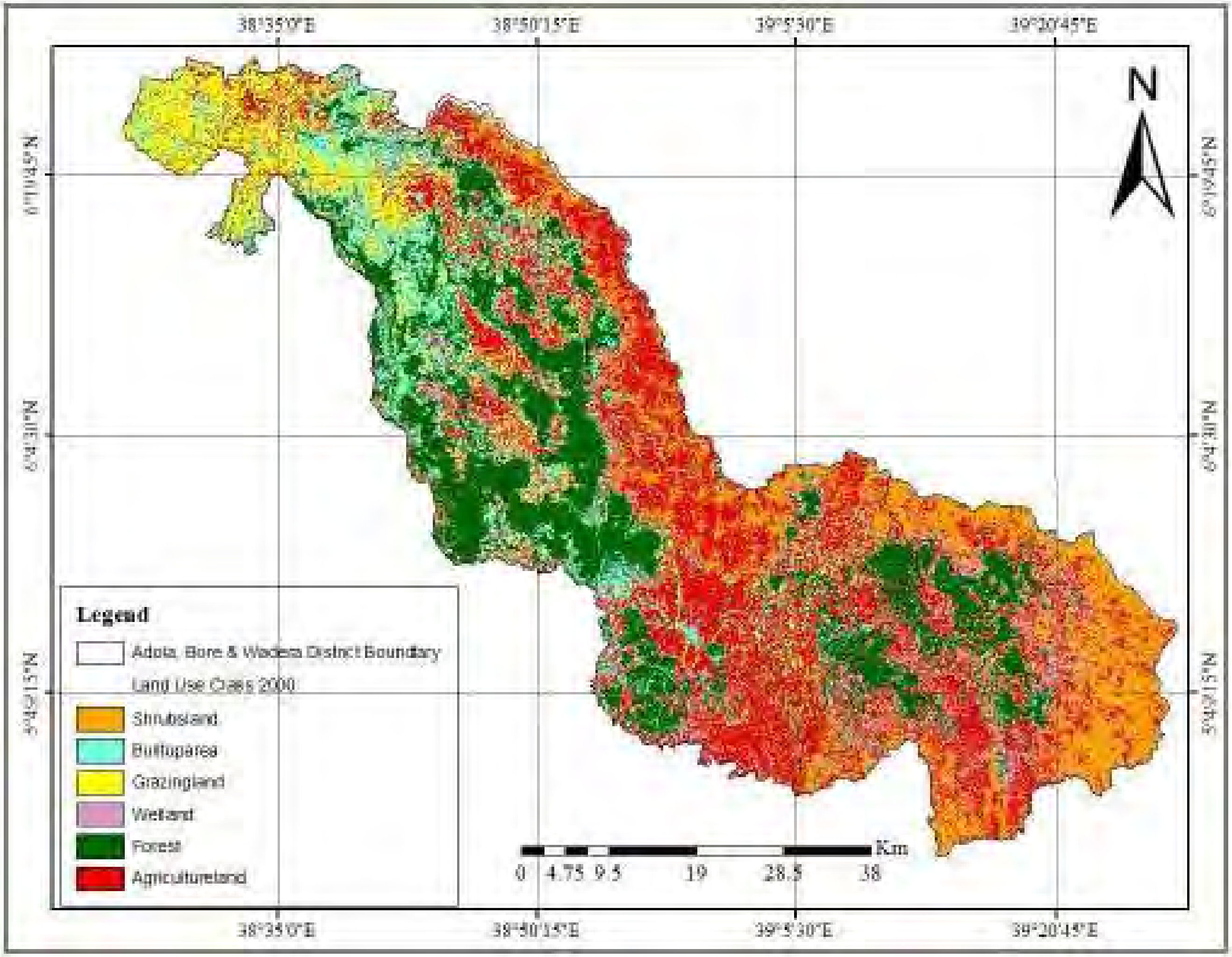


Figure 20 Land use/land cover map of Bore-Anferara-Wadera forest (2000).

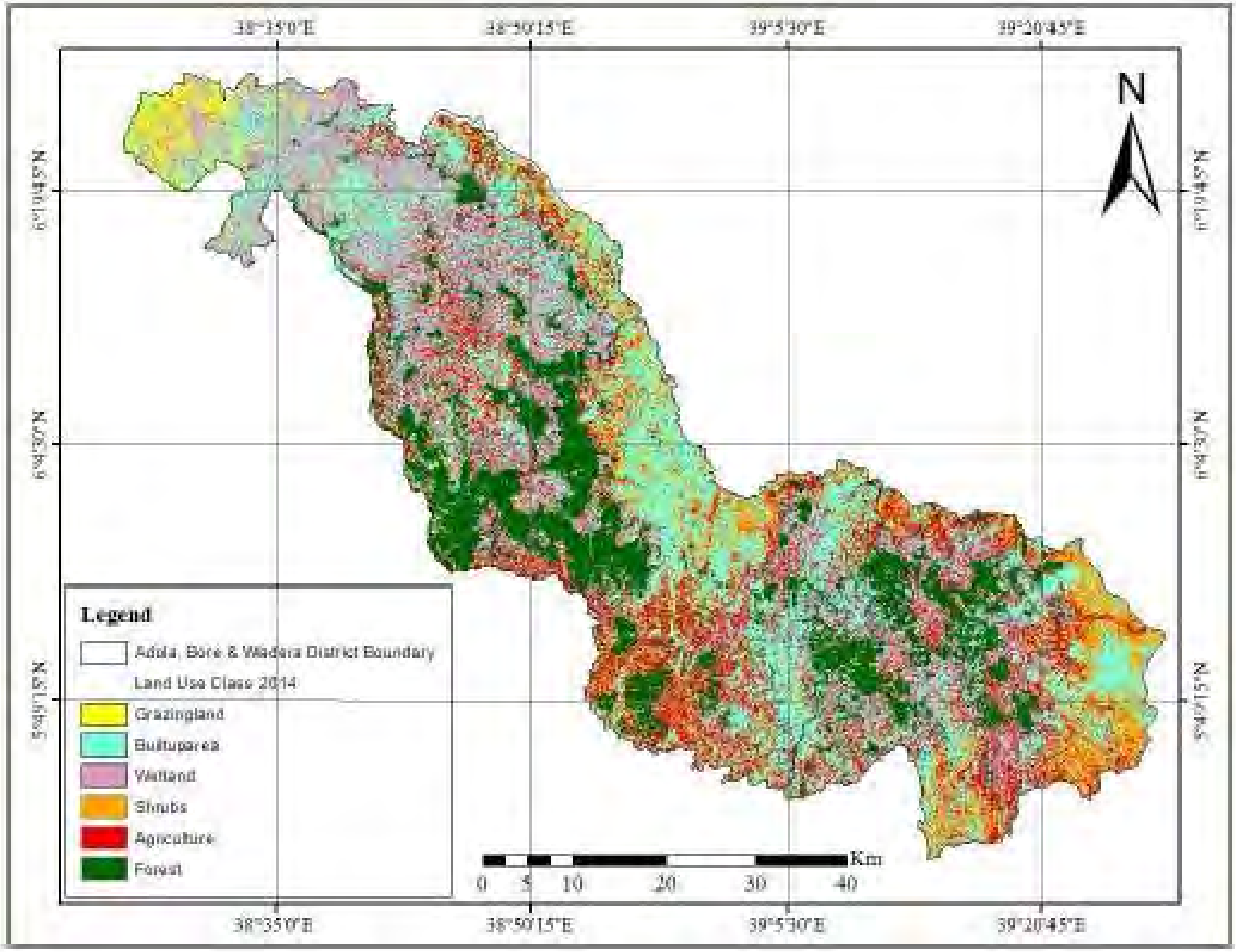


Figure 21 Land use/land cover map of Bore-Anferara-Wadera forest (2014).

4.9.2 Land use/land cover change from 1986 to 2000

The land use/land cover change matrix indicated the direction and extent of changes from 1986 to 2000 (Table 17 and Figure 22). In this period, built up area showed the highest change with a net gain of 18,819 ha (14,424 ha in 1986 to 33,243 ha in 2000) followed by agricultural land that gained 15,395 ha (43,687 ha in 1986 to 59,082 ha in 2000). A considerable area of agricultural land (5, 165 ha), forest (10, 783 ha), wetland (840 ha) and shrubland (5, 093 ha) were converted to the built up area. The area of grazing land and wetland also increased by 2,024 and 13,350 ha, respectively, whereas forest land (-21,462 ha) and shrubland (-28,126 ha) decreased considerably. The diagonal values in the table show the unchanged area coverage of that particular LULC class during the study period (Table 17 and Figure 22).

Table 17 LULC Matrices of Bore-Anferara-Wadera forest (1986 and 2000)

LULC Class		**Final State 2000 (ha)**						
		Agricultural land	Forest	Wetland	Built up area	Grazing land	Shrub-land	Total
Initial State 1986 (ha) (ha)	Agricultural land	**13302**	3410	5691	7874	3071	10339	43687
	Forest	9037	**89648**	7367	11016	645	4812	122525
	Wetland	3068	1189	**5062**	2731	910	166	13126
	Built up area	2709	233	1891	**2901**	4900	1790	14424
	Grazing land	4640	4850	2397	1838	**4273**	2853	20851
	Shrubland	26326	1733	4068	6883	9076	**85449**	133535
	Total	59082	101063	26476	33243	22875	105409	348148

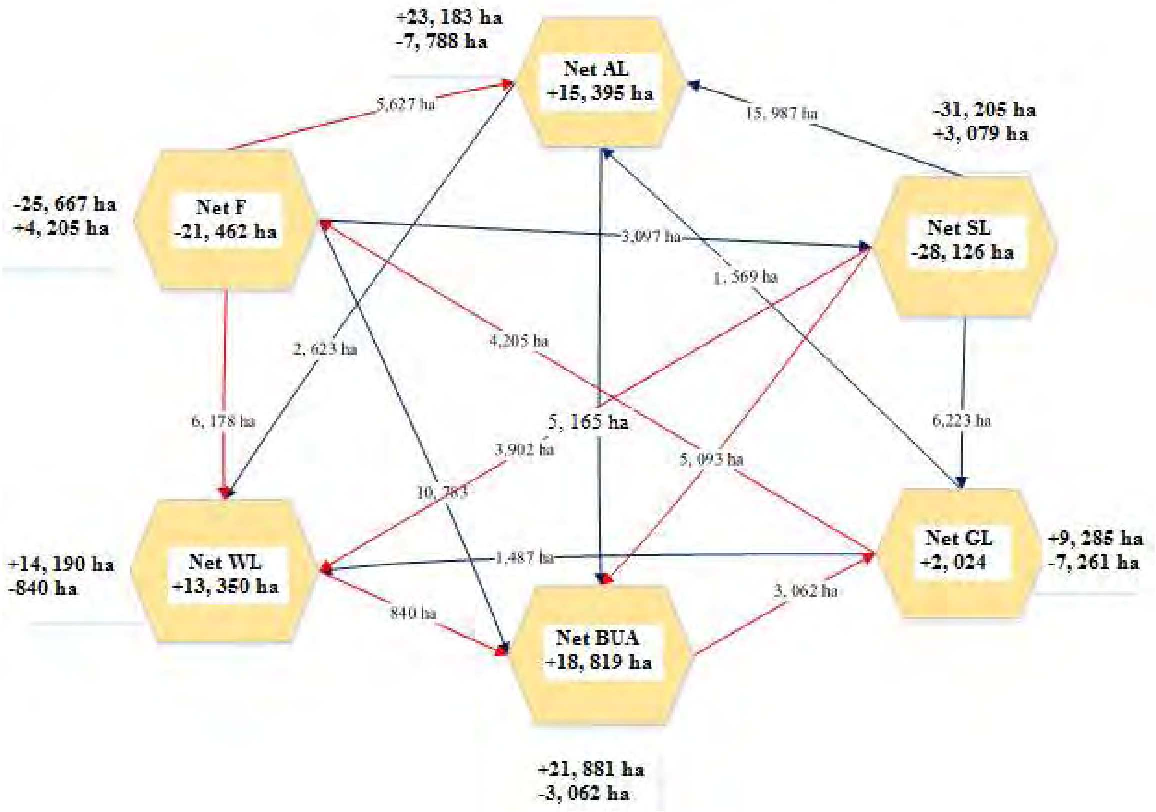


Figure 22 Land use and land cover dynamics of 1986 - 2000

4.9.3 Land use/land cover change from 2000 to 2014

In the period 2000 to 2014, agricultural land increased by 36,073 ha (59,082 ha in 2000 to 95,155 ha in 2014) followed by the built up area that increased by 22,511 ha (33,243 ha in 2000 to 55,754 ha in 2014). Very large amount of shrubland (16, 319 ha) was converted to agricultural land. In addition, a substantial amount of forest (10, 361 ha), wetland (2, 237 ha), built up area (2, 470 ha) and grazing land (4, 686 ha) were also converted to agricultural land. On the contrary, forest land and shrubland decreased considerably from 101,063 ha in 2000 to 69,851 ha in 2014 (-31,212 ha) and 105,409 ha in 2000 to 68,610 ha in 2014 (-36,799 ha), respectively (Table 18 and Figure 23).

Table 18 LULC Matrices of Bore-Anferara-Wadera forest (2000 and 2014)

LULC Class		**Final State 2014 (ha)**						
		Agricultural land	Forest	Wetland	Built up area	Grazing land	Shrub-land	Total
Initial State 2000 (ha)	Agricultural land	**23339**	908	8372	10667	3474	12322	59082
	Forest	11269	**65906**	4399	12996	1756	4737	101063
	Wetland	10609	543	**5828**	6533	1198	1765	26476
	Built up area	13137	1853	7831	**6830**	1422	2170	33243
	Grazing land	8160	41	1988	5311	**5538**	1837	22875
	Shrubland	28641	600	6874	13417	10098	**45779**	105409
	Total	95155	69851	35292	55754	23486	68610	348148

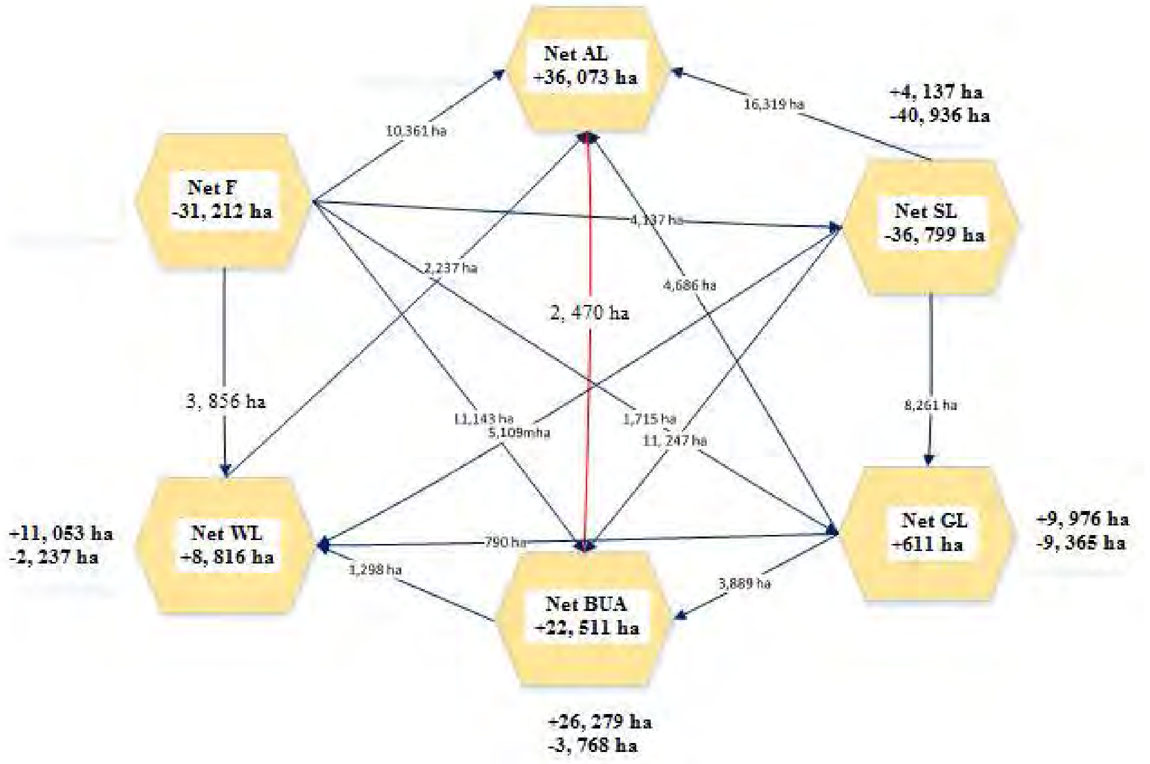


Figure 23 Land use and land cover dynamics of 2000 - 2014

4.9.4 Rate of land use/land cover change

Results showed that the area of agricultural land, built up area, grazing land and wetland increased while that of forest land and shrubland declined considerably in the whole period (1986 – 2014) (Table 19). In the first period (1986 – 2000), built up area increased by 18,819 ha, which is approximately equivalent to 1,344.21 ha/yr, while the increment of agricultural land was 15,395 ha (1,099.64 ha/yr) (Table 19). In the second period (2000 – 2014), the rate of increase in agricultural land was 36,073 ha (2,576.64 ha/yr), which is twice the first period. The rate of change for the built up area was 25,511 ha (1,607.92 ha/yr). In contrast, forest land and shrubland considerably declined both in the first and second periods. The largest decline was observed for shrubland that reduced by -

28,126 ha (-2,009 ha/yr) and -36,799 ha (-2,628.35 ha/yr) in the first and second periods, respectively. The other land use/land cover classes also increased both in the first and second periods (Table 19). Agricultural land was found to show the highest rate of change by 1,838.14 ha/yr increment followed by built up area 1,476 ha/yr, whereas the highest negative value was observed for shrubland (-2,318.75 ha/yr) and forest land (-1,881.21 ha/yr) during the whole period (1986 – 2014) (Table 19).

Table 19 Rate of changes in LULC classes (1986 – 2014)

LULC class	**1986 - 2000**		**2000 - 2014**		**1986 - 2014**	
	Area change (ha)	Rate of change (ha/yr)	Area change (ha)	Rate of change (ha/yr)	Area change (ha)	Rate of change (ha/yr)
Agricultural land	15395	1099.64	36073	2576.64	51468	1838.14
Built up area	18819	1344.21	22511	1607.92	41330	1476
Forest	-21462	-1533	-31212	-2229.43	-25674	-1881.21
Grazing land	2024	144.57	611	43.64	2635	94.11
Shrub land	-28126	-2009	-36799	-2628.5	-64925	-2318.75
Wetland	13350	953.57	8816	629.71	22166	791.64

Note: The negative values indicate the magnitude of the decline in that particular land use and land cover type.

4.9.5 Accuracy assessment of 1986, 2000 and 2014 maps

The overall classification accuracy assessment results for the three reference years are presented in Appendix 9. The lowest overall classification accuracy was 86.3% for 1986 and the highest was 89.5% for 2014. The accuracy level is well over the minimum requirement of 85% as set by Anderson *et al*. (1976) for effective land use/land cover change analysis. The highest Kappa statistic was 0.87 for 2014 and the lowest was 0.84 for 1986, all of which showed strong agreement with the ground information.

CHAPTER FIVE

5 DISCUSSION, CONCLUSION AND RECOMMENDATIONS

5.1 Discussion

5.1.1 Floristic composition

The results of this study show that Bore-Anferara-Wadera forest is rich in plant species diversity which is indicated by the presence of 136 species. The observed diverse flora of Bore-Anferara-Wadera forest is in agreement with the general pattern of high species diversity in the tropical montane forests. According to Coetzee (1978), among the Afromontane forest regions, the East African Mountains have the richest and most diverse flora. In Ethiopia, the available floristic data are either site-specific (e.g., Lisanework Nigatu and Mesfin Tadesse, 1989; Tadesse Woldemariam, 2003) or covering a wide range of vegetation types (Friis, 1992), as a result, it is difficult to make a direct comparison with other similar studies. However, the overall species richness of forests can give a general impression of their diversity. Accordingly, Bore-Anferara-Wadera forest has higher species richness than some other Afromonatne forests in Ethiopia such as Jibat (131 species) (Tamrat Bekele, 1994), Masha (130 species) (Abreham Assefa *et al.*, 2013), Tara Gedam (111 species) and Abebaye (88 species) (Haileab Zegeye *et al.*, 2011), Zegie (113 species) (Alemnew Alelign *et al.*, 2007) and more or less similar to the Afromontane and transitional rainforest in southwestern Ethiopia, i.e., 139 species (Kumelachew Yeshitila and Tamrat Bekele, 2002). However, its species richness was much lower than that reported for Bonga forest (243 species) (Ensermu Kelbessa and Teshome Soromessa, 2008), Yayu forest (220 species) (Tadesse Woldemariam *et al.*, 2008), Mana Angetu (212 species) (Ermias Lulekal *et al.*, 2008) and Magada forest (197 species) (Genene Bekele and Reddy, 2015). The patterns of plant diversity distribution over different forests could be attributed to a number of environmental factors that operate over multiple temporal and spatial scales (Shmida and Wilson, 1985; Brockway, 1998). Climate and topography appear to have broad effects on diversity across the landscape, while biological factors and availability of suitable environmental gradients seem to influence diversity more at the site level (Lovett *et al.*, 2000; Pausas and Austin, 2001; Tuomisto *et al.*, 2003). Moreover, forests with a high degree of human interference and disturbances for prolonged periods show relatively lower species richness than others (Urban *et al.*, 2000). According to Chen *et al.* (2003), environmental heterogeneity, regeneration success and competition are also important factors that shape species composition of forests.

The family Fabaceae has the highest representation of species (11 species, 8%) in Bore-Anferara-Wadera forest which could be attributed to the fact that it is the second largest family in the flora area of Ethiopia and Eritrea (Hedberg and Edwards, 1989). This could also be related to its efficient and successful dispersal strategies as well as better adaptation to a wide range of ecological conditions. Fabaceae was also found to be dominant in other Afromontane forests in Ethiopia like Belete (8 species, 10%) (Kitessa Hundera & Tsegaye Gadissa, 2008), Hugumburda (5 species, 6%) (Ermias Aynekulu *et al.*, 2011), Komto (12 species, 12%) (Fekadu Gurmessa *et al.*, 2012), and Jibat (14 species, 8%) (Tesfaye Burju *et al.*, 2013). Results also showed that the number of herbaceous species is lower when compared with other growth forms. This could be attributed partly to the closed canopy of the forest that prevents light from reaching the ground. The forest floor of Afromontane rainforests is usually dark and poor in species composition (Hedberg *et al.*, 2009). On the other hand, human interference and opening up of the forest canopy due to selective cutting of trees for different purposes increased the growth of herbaceous species in the forest understorey in other Ethiopian Afromontane forests (Ermias Lulekal *et al.*, 2008; Haile Yineger *et al.*, 2008; Fekadu Gurmessa *et al.*, 2012).

Bore-Anferara-Wadera forest is relatively poor in the diversity of plant species endemic to Ethiopia. However, with a record of six endemic plant species to Ethiopia Bore-Anferara-Wadera forest is better than, for example, Yayu forest with three endemic plant species (Tadesse Woldemariam *et al.*, 2008) and similar to Wondo Genet Afromontane forest with seven endemic plant species (Mamo Kebede *et al.*, 2013). The proportion of endemic plant species in other Afromontane forests of Ethiopia is high, ranging between 11-15% of the total number of species (Friis *et al.*, 2001). For example, Hussien Adal (2014) reported that 8.7% (31 species) of the plant species found in Borena Saint National Park were endemic. According to Kumelachew Yeshitila and Simon Shibru (2002), the southwest moist montane forests are poor in trees/shrubs endemicity compared to dry Afromontane forests. This is in agreement with the result of this study. All endemic plant species found in Bore-Anferara-Wadera forest are already on the Red List of endemic species of Ethiopia and Eritrea. Among these, *Lippia adoensis*, *Millettia ferruginea*, *Vepris dainellii* and *Vernonia rueppellii* were species with least concern while *Maytenus addat* is near threatened and *Kotschya recurvifolia* is in the vulnerable category (Vivero *et al.*, 2005). In this study, sixteen new plant species were recorded for the Sidamo floristic region of the Flora of Ethiopia and Eritrea, which accounts for 11.8% of the total species (Table 4).

5.1.2 Community diversity of the plant community types

Measures of community diversity play a major role in ecology and conservation biology (Magurran, 2004) since it is an important parameter of a plant community which is related to ecosystem dynamics and environmental quality (Liu & Brakenhielm, 1996). The four plant community types of Bore-Anferara-Wadera forest showed variation in their species richness, diversity, and evenness. This variation among the plant community types was the direct reflection of the effects of the environmental variables where these community types occurred. Comparatively, community 3 (mixed forest) was the most diverse and richest in its species composition than the other communities. This might be attributed to the combined effects of topographic, climatic and edaphic differences, as the representative plots were composed of three different forest patches (Anferara, Kilenso, and Danissa), and the concomitant occurrence of species adapted to these different environmental conditions. According to Suratman (2012), the difference in terrain, gradient and slope direction causes differences in the soils, water, and microclimate conditions which cause differences in species adaptability. Parthasarathy (2001) also reported that species diversity in a given forest area depends on climate and geographic location.

The difference in Shannon diversity index between community 1 (Anferara forest; H' = 3.38) and community 2 (KIlenso forest; H' = 3.49) might be related to the higher anthropogenic disturbances in these forest patches. According to field observation and focal group discussion, Anferara forest was heavily affected by the local people who were involved in clearing forests for expansion of farmland, selective cutting of mature woody species for timber production and for making farm implements and wooden pots that could be used for traditional gold sieving activities while in Kilenso forest the local people were involved in charcoal production covering a wider area, cutting of trees for house construction and expansion of farmland. Feyera Senbeta and Demel Teketay (2003) have shown that local climatic variations and forest disturbances are mentioned among the factors most responsible for variations in species diversity and evenness in a given forest. Engelbrecht *et al*. (2007) also stated that forest disturbances affect species diversity, amongst others, due to their effect on the removal of some preferred species, and the resulting change in the light environment of the understorey species.

Community 4 (Danissa forest) is the second in its diversity and the third in its species richness. This might be related to intermediate or reduced disturbance since most of the plots were located on a strong slope (average 22%) that could not be easily accessible by the local people (to exploit through selective cutting) and grazing animals. A similar result was reported by Abiyou Tilahun *et al*. (2011) on Menagesha Amba Mariam forest. Bongers *et al*. (2009) explained that intermediate levels of forest disturbance may promote community diversity by facilitating regeneration of some species.

Species evenness exhibits the relative abundance of a species in the sample plot. The lower evenness value observed in community 1 (Anferara forest) indicates the dominance of a few species such as *Acanthus eminens*, *Dracaena afromontana*, *Isoglossa* *somalensis*, *Lepidotrichilia volkensii*, *Ocotea kenyensis*, *Poecilostachys oplismenoides*, *Pouteria adolfi-friederici*, *Psychotria orophila*, *Syzygium guineense* subsp. *afromontanum* and *Teclea nobilis* in the community. Conversely, the high evenness values observed in community 3 (mixed forest) and community 4 (Danissa forest) shows little dominance by any single species but repeated coexistence of species over all plots in these communities. The overall Shannon – Wiener diversity index (H' = 3.84) of Bore-Anferara-Wdaera forest indicated high diversity in the study area. This finding was found to be higher than that of other Ethiopian Afromontane forests like Chilimo (H' = 2.72) (Tadesse Woldemariam *et al*., 2000), Zegie (H' = 3.72), (Alemnew Alelign *et al*., 2007), Tara Gedam (H'=2.98) (Haileab Zegeye *et al.*, 2011) and Wondo Genet (H' = 3.60) (Mamo Kebede *et al.*, 2013) forests. According to Kent and Coker (1992), the Shannon-Wiener diversity index normally varies between 1.5 and 3.5, and rarely exceeds 4.5. Thus, the H' value of Bore-Anferara-Wader forest resides within the normal range that indicates high plant species diversity of the study area, which may be related to the relatively high degree of topographic, climatic and edaphic factors differences in the area. The observed overall species richness (J = 0.78) in Bore-Anferara-Wadera forest is also higher than in Chilimo (J=0.68) (Tadesse Woldemariam *et al*., 2000) and Tara Gedam (J=0.65) (Haileab Zegeye *et al.*, 2011), indicating a relatively much more equitable distribution of individuals among various species in the study area.

Comparisons of species composition among the four forest patches (communities) indicated that Anferara forest patch shared the highest number of species with Kilenso and Mixed forests whereas Danissa forest shared the least number of species with the other forest patches. In other words, Anferara forest patch showed the least species turnover (the least β-diversity) while Danissa forest patch showed the highest species turnover since similarity coefficients and beta diversity are inversely related (Table 7). The higher number of shared species in Anferara forest might be related to the narrow geographical distance between the two forest patches (Anferara and Kilenso) where most of the plots forming these communities may have relatively similar environmental factors. Terborgh and Andresen (1998) and Pyke *et al*. (2001) also reported that environmental variables change with geographical distance which leads to the floristic dissimilarity. Compared to other forest patches, Danissa forest is relatively far away in its location from the two forest patches with a narrow altitudinal range of the plots forming this community may have created different environmental conditions responsible for the highest species turnover (β-diversity) of the community. As a result, this community formed a relatively different species composition.

5.1.3 Plant community – environmental variables relationship

In community ecology, discrete habitats or plant community types are distinguished in terms of species composition or environmental variables (Van Tongeren, 1995; McCune and Grace, 2002). Differences in species composition observed among the four plant communities identified in Bore-Anferara-Wadera forest could be attributed to variations in environmental gradients. According to Whittaker *et al*. (2003), vegetation patterns among communities can best be explained by differences in environmental gradients. Plant community distribution is an expression of physical gradients (elevation, soil heterogeneity and microclimate), biotic responses to these gradients and human-induced and/or environmental disturbances in a region (Lima and Zollner, 1996; Urban *et al*., 2000). Various ordination and clustering methods have been found useful in providing detailed information on the patterns of vegetation composition and the response of plant species to underlying environmental gradients (Gauch, 1982; Van Tongeren, 1995; McCune and Grace, 2002).

In this study, both multivariate techniques (ordination and cluster analysis) consistently indicated patterns of floristic groupings, their relationships with environmental variables and the presence of floristic heterogeneity. Floristic heterogeneity is based on the difference in altitude, slope, local landscape, exposure, soil physical and chemical properties, geological history, and anthropogenic factors (Hedberg, 1964; Friis, 1992; Tadesse Woldemariam, 2003).

The major distinguishing feature of the identified plant communities in Bore-Anferara-Wadera forest is the difference in dominant plant species in each community. It appears that species frequently show marked preferences for specific environmental conditions. The species compositions of plant communities identified in Bore-Anferara-Wadera forest are similar with Afromontane rainforest vegetation of southwest Ethiopia. The dominant plant species in community 1 (Anferara) and community 2 (Kilenso), *Dracaena afromontana*, *Syzygium guineense* subsp. *afromontanum* and *Ocotea kenyensis* are the characteristic species of Afromontane rainforest (Friis, 1992), where the canopy layer is occupied by mixed broad-leaved species, e.g., *Croton macrostachyus*, *Polyscias fulva* and *Olea capensis* subsp. *macrocarpa*. *Pouteria adolfi-friederici*, *Psychotria orophila* and *Teclea nobilis*, the dominant species of community 3 (Mixed) and community 4

(Danissa), are also the characteristic species of Afromontane rainforest vegetation (Friis, 1992). The majority of the tree and shrub species recorded within these community types are also characteristic species of this forest type. Although the lower altitude in all the plant communities is higher than the upper limit of the altitudinal range of the transitional rainforest (500 – 1500 m a.s.l.), plant species belonging to the transitional rain forest include *Elaeodendron buchananii*, *Phoenix reclinata*, and *Vepris dainellii*.

Even though the upper canopies of the identified plant communities were dominated by different species in Bore-Anferara-Wadera forest, various degrees of combination of similar species were observed across communities. This can be related to the overlapping of altitudinal ranges among different communities, and the resulting relatively small differences in environmental gradients across adjacent plots forming different communities. Condit *et al.* (2002) reported that small differences in species composition across plant communities may reflect adaptations of species to similar environmental conditions coupled with efficient dispersal of the respective species forming the communities.

From the ordination result, it has been found that most of the variation in the patterns of plant species composition was explained by the first CCA axis (Table 9). Among the variables altitude was highly correlated with axis one (Table 9), thus altitude was the major environmental variable in explaining variations in plant species distribution and patterns of plant community formation in the study area. Although the forest patches have some overlapping ranges of altitude, there is apparent variation in species composition along the altitudinal gradient between the forest patches. This could be related to a continuous change in environmental variables along the altitudinal gradient (Lovett and Wasser, 1993). The importance of altitude as a major determinant of vegetation variation agrees well with reports of similar vegetation studies made along altitudinal gradients (Zerihun Woldu *et al*., 1989; Lisanework Nigatu and Mesfin Tadesse, 1989; Kumelachew Yeshitila and Tamrat Bekele, 2002; Haile Yeneger *et al.*, 2008; Motuma Didita *et al.*, 2010; Tadesse Woldemariam *et al.*, 2008; Schmitt et al., 2010; Ermias Lulekal, 2014). Hamilton (1982) has also reported the role of altitudinal gradients in shaping distribution of vegetation in East African Mountains. Altitude affects atmospheric pressure, moisture and temperature in an area which in turn influence the growth and development of plants, and the corresponding patterns of vegetation distribution (Hedberg, 1964). Altitudinal variations will always be accompanied by varying degrees of differences in microclimatic conditions which influence temperature and soil moisture, resulting in variations in community structure. Bolstad *et al.* (1998) have also explained that plants in mountainous regions respond to small-scale variations related to altitudinal changes that affect microclimatic conditions.

Considering multiple gradients may improve understanding of mechanisms that influence plant community patterns in mountainous regions of the tropics. Slope, though not highly correlated with the first axis of the CCA ordination diagram, it was correlated with CCA second axis (Figure 11 and Table 9) which explained some amount of variation in the patterns of species distribution and plant community formation. Variation in slope also affects the composition and structure of forest vegetation since the soils on steeper slopes are influenced by bedrock and tends to be less acidic and less moist (Tewolde Berhan Gebre Egziabher, 1986; cited in Tadesse Woldemariam, 2003). Other environmental variables, related to physical and chemical properties of soil were also correlated in various degrees with CCA axes 1-4, indicating that these variables had their own share in determining patterns of plant species composition and plant community formation in the study area. However, these variables are directly or indirectly affected by altitude and slope.

In addition, Bore-Anferara-Wadera forest is also subjected to human interference mainly for charcoal and timber production, and hence disturbance is apparent. In my field observation and focal group discussions, the local people were involved in the charcoal production and selective felling of larger trees for timber production in most of the forest patches in order to earn additional income to support their livelihoods.

From the species composition, altitudinal ranges, types of dominant species in the upper canopies of the forest and climatic conditions of the area indicated that Bore-Anferara-Wadera forest belongs to the category of moist evergreen Afromontane forest. According to Friis (1992), moist evergreen Afromontane forests lie between altitudinal ranges of 1500 – 2600 m a.s.l., annual rainfall between 700 – 1500 mm and consists of upper and middle canopies dominated by species like *Pouteria adolfi-friederici*, *Syzygium guineense* subsp. *afromontanum*, *Ocotea kenyensis*, *Dracaena afromontana*, and *Psychotria orophila*, which were all found to be satisfied in Bore-Anferara-Wadera forest.

5.1.4 Vegetation structure

Tree and shrub density

In Bore-Anferara-Wadera forest, most plant species were represented by a small number of individuals distributed in different plots, a common pattern for most tropical forests (Valencia *et al*., 1994). However, Bore-Anferara-Wadera forest showed relatively high woody species density (1,047 individuals ha^{-1}) when compared with some other Afromontane forests such as Komto (952 individuals ha^{-1}) (Fekadu Gurmessa *et al.*, 2012), Jibat (702 individuals ha^{-1}) (Tesfaye Burju *et al.*, 2013) and Gedo (781 individuals ha^{-1}) (Birhanu Kebede *et al.*, 2014), while its density was found to be lower than some other forests such as Belete (1482 individuals ha^{-1}) (Kitessa Hundera and Tsegaye Gadissa, 2008), Masha-Anderacha (1,709 individuals ha^{-1}) (Kumilachew Yeshitila and Taye Bekele, 2003) and Tara Gedam (3,001 individuals ha^{-1}) (Haileab Zegeye *et al.*, 2011) forests. This could be attributed to variations in topographic gradients and habitat preferences of species forming the forest, and the degree of anthropogenic disturbances (Whittaker *et al*., 2003).

The ratio of density of trees and shrubs with DBH > 10 cm to DBH > 20 cm in Bore-Anferara-Wadera forest (2.03) was found to be higher than that of some other Afromontane forests such as Wof Washa (1.5) (Tamrat Bekele, 1993), Denkoro (1.9) (Abate Ayalew *et al.*, 2006), Dodola (1.5) (Kitessa Hundera *et al.*, 2007), Komto (1.5) (Fekadu Gurmessa *et al.*, 2012) and Dense (1.5) (Ermias Lulekal, 2014), indicating a higher predominance of small-sized individuals in Bore-Anferara-Wadera forest than the forests compared. It is evident that Bore-Anferara-Wadera forest had been exposed to a long history of heavy anthropogenic disturbance as indicated by the absence of large-sized individuals. Grubb *et al*. (1963) have shown that lower values for the ratio of small-sized individuals to large-sized individuals could be indicators of forests which developed under natural conditions and with minimum disturbances, while higher values indicate a predominance of small-sized individuals that start to grow following excessive cuttings or other anthropogenic disturbances.

DBH and Height class distribution

The patterns of DBH class distributions indicate the general trends of population dynamics and recruitment processes of species in a given forest. The results on DBH class distribution of trees and shrubs across the seven DBH classes in Bore-Anferara-Wadera forest indicate an inverted J-shape distribution (Figure 12). This is a general pattern of normal population structure where the majority of the species had the highest number of individuals at lower DBH classes with a gradual decrease towards the higher classes. Such inverted J-shape distributions across DBH classes are indicative of an overall healthy regenerating forest (Silvertown and Doust, 1993). Similar pattern of population structure was reported in previous studies from different Afromontane forests of Ethiopia (e.g. Feyera Senbeta and Demel Teketay, 2003; Kitessa Hundera *et al.*, 2007; Haile Yineger *et al.*, 2008; Haileab Zegeye *et al.*, 2011; Fekadu Gurmessa *et al.*, 2012; Tesfaye Burju *et al.*, 2013; Ermis Lulekal, 2014).

The percent distribution of trees in Bore-Anferara-Wadera Forest in the DBH classes ≤ 50 cm was found to be higher (91.42) than that of some other Afromontane forests in Ethiopia such as Jibat (87.5) (Tesfaye Burju *et al.*, 2013) and Dense (70.1) (Ermias Lulekal, 2014), indicating a better distribution of trees in the lower DBH classes (1-3) of Bore-Anferara-Wadera forest than the forests it was compared. Hence, the result still confirms that there is the dominance of small-sized individuals in Bore-Anferara-Wadera forest than the forests used for comparison. On the other hand, in the higher DBH classes (> 80 cm) better species distribution was observed for Jibat (6.5) (Tesfaye Burju *et al.*, 2013) and Dense (18.6) (Ermias Lulekal, 2014) than Bore-Anferara-Wadera (3.48) forest. The presence of a large number of small to medium-sized individuals and the relatively small proportion of large-sized trees indicates that the Bore-Anferara-Wadera forest is in a stage of secondary development. According to Aye *et al*. (2014), the presence of higher tree density in small DBH classes compared to large DBH classes is a secondary forest characteristic. A similar result was also reported for Denkoro (Abate Ayalew *et al.*, 2006), Belete (Kitessa Hundera and Tsegaye Gadissa, 2008) and Komto (Fekadu Gurmessa *et al.*, 2012) forests. Distribution of individuals in different height classes also reveals a similar trend in Bore-Anferara-Wadera forest. Height can be used as an indicator of the age of the forest. The percent distribution of trees and shrubs decreased with increasing height classes showing an inverted J-shape pattern of distribution with a slight depression in the 18 – 23 m class, indicating selective cutting of individuals in that particular height class (Figure 13). The decline in the number of individuals in each height class towards the highest classes indicates the dominance of small-sized individuals in the forest, which was a characteristic of the high rate of regeneration. For instance, about 54.96% of individuals are represented in the height class of 3 – 8 m, 12.11% in the height class 8 – 13 m and only 3.03% reaches a height of above 33 m.

Vertical structure

The result obtained from analysis of the vertical structure of Bore-Anferara-Wadera forest indicates that the highest proportion of species is concentrated in the lower storey followed by the middle and upper storey. Hence, the result still confirms that there is the dominance of small-sized individuals in the forest indicating that it is in a stage of secondary development. A similar result was reported by Ensermu Kelbessa and Teshome Soromessa (2008) and Fekadu Gurmessa *et al*. (2012) in Bonga and Komto forests, respectively. However, the vertical structure of the forest could be influenced by the character of the physical environment (Kent and Coker, 1992) and the microclimate (Grubb *et al.*, 1963; Stoutjesdijk and Barkman, 1992). In addition to physical environments, human factors can modify the vertical structure of the forest. According to focus group discussions with local people, there is a long history of repeated forest fire that destroyed the forest and logging by people coming from other areas due to resettlement. These human activities must have contributed to the reduction of upper storey trees, since most of these tree species are used for timber, e.g., *Pouteria adolfi-friederici* and *Ocotea kenyensis*.

Basal area

The total basal area for Bore-Anferara-Wadera forest (75.23 m^2 ha^{-1}) is higher than the normal basal area for virgin tropical forests in Africa (22 – 37 m^2 ha^{-1}) (Lamprecht, 1989). According to Gentry (1988), the density of woody plants in the montane forests of

Africa is relatively high as compared to other tropical montane forests. This may have contributed to the high basal area in Bore-Anferara-Wadera forest. Feyera Senbeta (2006) also reported higher basal area of the Afromontane rainforests in Ethiopia than those reported for tropical forests due to a high density of individuals in the forests studied. Comparison of the basal area of this forest with other Afromontane forests in Ethiopia shows that it is higher than Chilimo (30.1 m^2 ha^{-1}), Menagesha (36.1 m^2 ha^{-1}) and Jibat (49.8 m^2 ha^{-1}) (Tamrat Bekele, 1993), Denkoro (45 m^2 ha^{-1}) (Abate Ayalew *et al.*, 2006), Komto (50.72 m^2 ha^{-1}) (Fekadu Gurmessa *et al.*, 2012) and lower than Wof Washa (101.8 m^2 ha^{-1}) (Tamrat Bekele, 1993), Masha-Anderacha (81.9 m^2 ha^{-1}) (Kumelachew Yeshitila and Taye Bekele, 2003), Dodola (129 m^2 ha^{-1}) (Kitessa Hundera *et al.*, 2007) and Mana Angetu (94 m^2 ha^{-1}) (Ermias Lulekal *et al.*, 2008). This may be due to variations in the conservation of the forests, exposure to deforestation and geographical location of the forests.

The results of a basal area analysis provide a better measure of the relative dominance of woody species in a forest (Cain and Castro, 1959). Thus, about 31.55% (23.73 m^2/ha) of the total basal area of Bore-Anferara-Wadera forest was contributed by *Syzygium guineense* subsp. *afromontanum*. Tree species with the largest contribution in the basal area can be considered the most important species, and hence, *Syzygium guineense* subsp. *afromontanum* was found to be important in Bore-Anferara-Wadera forest. The predominance of this tree species in the forest was due to the low market demand for its timber. *Syzygium guineense* subsp. *afromontanum* was also the dominant species in Belete and Komto forests for the same reason (Kitessa Hundera and Tsegaye Gadissa, 2008; Fekadu Gurmessa *et al.*, 2012). Comparison of the contribution of the different DBH classes to the total basal area shows that more than 74% of the basal area was occupied by the upper DBH (above 50cm) whose density was only 8.88% of the total individuals in the forest, and this was due to the presence of very few, large-sized individuals of the canopy tree. Similar results were observed in other Afromontane forests in Ethiopia such as Masha-Anderacha (Kumelachew Yeshitila and Taye Bekele, 2003), Dodola (Kitessa Hundera *et al.*, 2007), and Komto (Fekadu Gurmessa *et al.*, 2012).

Frequency

Frequency is the number of sampling plots in which a given species occurred in the study area. It gives an indication for homogeneity and heterogeneity of given vegetation (Lamprecht, 1989). A higher number of species in lower frequency classes and lower number of species in higher frequency classes indicate a high degree of floristic heterogeneity while a small number of species in lower frequency classes and a large number of species in higher frequency classes show floristic homogeneity or similar species composition (Aye *et al.*, 2014). Accordingly, this study reveals high percentage number of species in the lower frequency classes and low percentage number of species in the higher frequency classes indicating the existence of floristic heterogeneity in the forest. The frequency occurrence of most woody species is generally low. Of the five most frequent woody species observed in this study, only one of them is dominant while the other four species are not dominant or have little dominance. Fewer occurrences of the dominant trees in Bore-Anferara-Wadera forest agrees with the assertion of Myers and Shelton (1980) that where there is a great difference in the life-form of species, the largest species is often dominant and it rarely happens that the largest species is also numerically the commonest. According to Rey *et al.* (2000), high frequency of a species always depends on factors which relate to habitat preferences, adaptation, the degree of exploitation and availability of suitable environmental conditions for regeneration.

Importance Value Index

Importance Value Index (IVI) is an important parameter that reveals the ecological significance of species in a given ecosystem (Lamprecht, 1989). It combines data from three parameters, i.e., relative frequency, relative density and relative dominance (Kent and Coker, 1992), and hence many ecologists consider it as the most realistic aspect of vegetation study (Curtis and McIntosh, 1950). The IVI output from the study showed higher values for *Syzygium guineense* subsp. *afromontanum* and *Pouteria adolfi-friederici* than for any other species in Bore-Anferara-Wadera forest. High basal area values of these species were the major factors explaining these high IVI values.

According to Mueller-Dombois and Ellenberg (1974) and Curtis and McIntosh (1950), the relative ecological significance and/or dominance of a tree species in a forest ecosystem could best be identified from IVI analysis. Thus, in this study, the IVI results also confirmed that these two species (*Syzygium guineense* subsp. *afromontanum* and *Pouteria adolfi-friederici*) are the most important/dominant species in the study area.

5.1.5 Population structure

The results obtained from population structure analysis of woody species in Bore-Anferara-Wadera forest showed three major representative patterns of the density distribution of trees and shrubs across different DBH classes. According to Popma *et al.* (1988), the patterns of species population structure can be interpreted as an indication of variation in population dynamics and regeneration status in the forest. The first pattern (Figure 17a) represented by *Pouteria adolfi-friederici*, indicates the presence of the highest density in the lower DBH classes with a gradual decrease in density towards the higher DBH classes. This population pattern is an indicator of a stable population structure and healthy regeneration of the forest (Silvertown and Doust, 1993; Feyera Senbeta, 2006; Getachew Tesfaye *et al.*, 2010). This type of population structure is characteristic of species with high potential for reproduction and shade-tolerant canopy trees that maintain a more or less constant rate of recruitment. There is a large probability that the death of an adult tree will be replaced by the growth of individuals from the smaller size classes and seems to be a self-maintaining plant population. *Dracaena afromontana*, *Ocotea kenyensis*, and *Psydrax schimperiana* also demonstrate such an inverted J-shaped distribution, suggesting good reproduction and recruitment potential in the forest.

The second type (Figure 17b) was represented by *Syzygium guineense* subsp. *afromontanum* that shows a bell-shaped distribution pattern. It follows a Gauss-type distribution pattern having a low frequency in the first and second DBH classes, a gradual increase in the number of individuals in the middle classes followed by a subsequent decrease in density towards the higher DBH classes. This type of distribution pattern indicates a poor reproduction and recruitment of species (Tamrat Bekele, 1993; Feyera Senbeta *et al.*, 2007), and also a decline in the number of large-sized trees. Selective cutting of large-sized individuals for various purposes, mainly for timber and construction purposes, could be the reason for the decline in the number of large-sized trees. Feyera Senbeta (2006) also reported poor reproduction of *Syzygium guineense* in the Afromontane forests of Ethiopia as a result of either most trees are not producing seeds due to age or there are seed losses due to predators after reproduction since its fruits are usually used as food by many animals and also humans. *Olea capensis* subsp. *macrocarpa*, *Macaranga capensis*, and *Allophylus abyssinicus* also show this type of distribution pattern. Ermias Lulekal *et al*. (2008) and Fekadu Gurmessa *et al*. (2012) reported similar results from different Afromontane forests of Ethiopia.

The third type (Figure 17c) was represented by *Trichilia emetica*. It is formed by species having an irregular distribution over DBH classes. Some DBH classes were poorly represented indicating selective removal of specific sized individuals while other DBH classes were well represented. Most of the species with irregular distributions are tree species that are hunted by the local people selectively. For example, *Prunus africana* is required for house construction at its medium size (Personal communication with the local people). Feyera Senbeta (2006) has shown that logging has been extremely selective and mostly confined to a few highly valuable timber tree species in Ethiopia. This could be one reason for some tree species to have distorted population structure in the Afromontane forests. In addition, livestock grazing and trampling (observed in the forest) which affects the seedlings under the mother tree could be another reason for such irregularities. *Ficus sur*, *Podocarpus falcatus*, and *Croton macrostachyus* also show this population pattern. Similar results were reported from different Afromontane forests in Ethiopia (Kumelachew Yeshitila and Taye Bekele, 2003; Kitessa Hundera and Tsegaye Gadissa, 2008; Motuma Didita *et al*., 2010; Tesfaye Burju *et al.*, 2013). Generally, the observed representative population structures of species in Bore-Anferara-Wadera Forest are indicators to the ultimate need of overall conservation activities, giving priorities to those species with poor reproduction and hampered recruitment. In addition, the existence of certain missed out individuals at different DBH classes indicate selective cutting of preferred-sized individuals for various purposes by the local people which require due attention.

5.1.6 Regeneration status of Bore-Anferara-Wadera forest

The regeneration status of tree/shrub species in any forest is determined by recruitment of saplings and seedlings (Dhar *et al.*, 1997; Samant *et al.*, 2002). In Bore-Anferara-Wadera forest, the density of seedlings and saplings shows huge variation. Species such as *Grewia ferruginea*, *Acacia bussei*, *Celtis toka*, *Ficus thonningii*, *Pavetta oliveriana*, *Schefflera abyssinica* and *Terminalia schimperiana* were represented by a few number of mature trees/shrubs, suggesting that these species might be on the verge of local extinction as there are no individuals at the seedling and sapling stage to replace the adult plants. The major reasons for poor or hampered regeneration include unfavorable environmental factors like rocky land and poorly developed soil, seed predation, human disturbance particularly livestock grazing and trampling and ability of a species to reproduce in the forest. On the contrary, prevention of livestock grazing and overexploitation could improve the regeneration status of woody species. Other species such as *Cordia africana*, *Phoenix reclinata*, *Rosa abyssinica*, *Pterolobium stellatum*, *Vernonia rueppellii*, *Flacourtia indica*, *Lobelia giberroa*, *Rhamnus prinoides*, *Toddalia asiatica* and *Zanthoxylum usambarense* were represented by seedlings and/or saplings and very few mature plants, indicating the possibility of replacement of mature plants in the future and hence show relatively better regeneration status. Similar findings were reported in different Afromontane forests of Ethiopia (Simon Shibru and Girma Balcha, 2004; Dereje Denu, 2006; Ensermu Kelbessa and Teshome Soromessa, 2008; Haileab Zegeye *et al.*, 2011; Fekadu Gurmessa *et al.*, 2012; Gebremicael Fisaha *et al.*, 2013). With regard to conservation priority, we suggest that priority should be given to the forests containing naturally and locally threatened species giving particular emphasis to species with no seedlings and saplings (priority class 1) in order to save them from local extinction and species under priority classes 2 & 3 also need proper management practices.

5.1.7 Floristic similarity of Bore-Anferara-Wadera forest with other Afromontane forests

The similarity analysis between Bore-Anferara-Wadera forest and eight other Afromontane forests in Ethiopia revealed high Sorensen's similarity values between Bore-Anferara-Wadera and Belete forests (0.47) and Bore-Anferara-Wadera and Magada forests (0.41) which indicate that Bore-Anferara-Wadera forest is floristically more related to Belete and Magada forests than to the other forests compared. Belete forest is located in Jimma zone of Oromia National Regional State, southwestern Ethiopia

(Kitessa Hundera and Tsegaye Gadissa, 2008). The observed high floristic similarity between Bore-Anferara-Wadera and Belete forests may be attributed to the similarity in altitudinal ranges. The lowest elevation point (1850 m a.s.l. in Belete and 1828 m a.s.l. in Bore-Anferara-Wadera) and the highest elevation point (2250 m a.s.l. in Belete and 2304 m a.s.l. in Bore-Anferara-Wadera) show almost similar readings in both forests. In addition, *Pouteria adolfi-friederici* and *Syzygium guineense* subsp. *afromontanum* were the dominant upper canopy tree species in Belete forest (Kitessa Hundera and Tsegaye Gadissa, 2008), which were also found to be true in Bore-Anferara-Wadera forest. Therefore, the overlap in altitudinal ranges may have resulted in species composition similarity between the two forests. Many studies (e.g., Kumelachew Yeshitila and Tamrat Bekele, 2002; Haile Yeneger *et al.*, 2008; Motuma Didita *et al.*, 2010; Schmitt *et al.*, 2010) have also emphasized the importance of altitude as one of the major determinants of plant species distribution patterns in tropical forests.

The observed high floristic similarity between Bore-Anferara-Wadera and Magada forests could be attributed to their geographical proximity and the relatively similar altitudinal ranges and climatic conditions. Magada forest is found in Borena zone of Oromia National Regional State, southern Ethiopia. The altitude of Magada forest ranges from 1750 to 2100 m a.s.l. (Genene Bekele and Reddy, 2015). Both Magada and Bore-Anferara-Wadera forest areas receive relatively similar mean annual rainfall (809 mm in Magada and 814 mm in Bore-Anferara-Wadera) in a bimodal rainfall pattern with similar rainy seasons. Thus, geographical proximity and overlap in altitudinal ranges and climatic conditions may have resulted in the observed good floristic similarity between the two forests. In general, our results indicate that Bore-Anferara-Wadera forest exhibits relatively better floristic similarity with moist Afromontane forests (MAF) (Belete, Magada, Mana Angetu, Gurra Farda and Masha) than with the dry Afromontane forests (DAF) (Egdu, Dodola, and Hugumburda) of Ethiopia.

5.1.8 Land use/land cover dynamics of 1986 – 2014

Ecosystems are not static but exposed to certain natural and human-induced disturbances that caused temporal changes. As a result, changes due to disturbances are likely to affect community structure, composition, diversity and distribution (Jentsch *et al*., 2002). Such responses become stronger with increasing magnitude and frequency of the disturbances. Classification accuracy assessment results of the year 1986, 2000 and 2014 are given in Appendix 9. The highest Kappa coefficient was 0.87 for 2014 and the lowest was 0.84 for 1986. Rahman *et al.* (2006) have shown that Kappa values are categorized into three groups: a value greater than 0.80 (80%) represents strong agreement, a value between 0.40 and 0.80 (40 to 80%) represents moderate agreement, and a value below 0.40 (40%) represents poor agreement. Therefore, this finding showed that there is strong agreement (higher Kappa statistics) between the classification map and the ground reference information confirming that the classified land use/land cover types are truly on the ground. Thus, six land use/land cover classes were identified from the supervised classification of the three images (1986, 2000 and 2014), namely, agricultural land, forest land, built up area, grazing land, shrubland and wetland.

5.1.8.1 Land use/land cover trends in the two study periods

The total area covered by green vegetation (forest land, shrubland and grazing land) in 1986 and 2000 was greater than the total area covered by the remaining land use/land cover types (agricultural land, built up area and wetland) which was the highest in 1986 (79.54%) and the least in 2000 (65.88%) while in 2014 the reverse holds true, i.e., the area covered by green vegetation is lower than that covered by other land use/land cover types. Due to the expansion of agricultural land and built up area the natural vegetation cover was greatly deteriorated and degraded. During the study periods (1986 – 2014), the area covered by forest declined at a rate of 1881.21 ha/yr which is equivalent to 1.33% of the national annual forest cover loss (141,000 ha/yr) reported by FAO (2010). Major deforestation took place over the last 28 years that resulted in 43% forest cover decline in the study area. In addition, a considerable amount of shrubland (48.6%) was also declined in the same period. This finding agrees well with Gete Zeleke and Hurni (2001) and Mohammed Assen (2011) who reported a 26.7% and 58.6% decline of natural forest cover in northwestern Ethiopia and Lake Alemaya watershed eastern Ethiopia, between 1957 and 1995 and 1965 and 2007, respectively. Mengistie Kindu *et al.* (2013) also reported 11.8% reduction of natural forest cover between 1973 and 2012 in Munessa-Shashemene landscape of the Ethiopian highlands. The finding that there is a major decline in the area of the shrubland is in line with that of Belay Tegene (2002), who reported a 58% decline of the cover type in the Derekolli catchment of South Wollo, between 1957 and 1986. A dramatic expansion of agricultural land and built up area occurred due to the transformation of a considerable amount of forest land, grazing land, shrubland, and wetland into these land cover types (Table 17 & 18 and Figure 22 & 23).

A similar result was reported by Amanuel Abate (2013) where agricultural land and built up area increased at the expense of grassland, forestland, bushland and reverine forest in Nadda Asendabo watershed, southwest Ethiopia.

Transformation of such large area of land into agricultural land and built up area was largely due to population expansion. Population expansion resulted in a scarcity of land for farming and house construction which compelled farmers to use all available and marginal lands. This led to a rapid increase and wide area coverage of agricultural land and settlement area. For instance, agricultural land increased by 4.42% and 10.36% in the two periods (1986 – 2000 and 2000 – 2014), respectively, where most of the expansion occurred in the later period and similarly built up area increased by 5.41% and 6.46% in the two periods (Table 16). This finding is in agreement with Abate Shiferaw (2011) who reported that agricultural land increased by 18% in Borena woreda of south Wollo due to rapid population growth, between 1972 and 2003. Hurni *et al.* (2005) also reported that changes in population densities in the upper Nile Basin of Ethiopia due to rapid population growth had an effect on land use/land cover types, resulting in the shrinking of forests and grasslands and expansion of cultivated areas.

In general, it was found out that agricultural land, built up area, wetland and grazing land increased by 14.78%, 11.87%, 6.37% and 0.76% whereas shrubland and forestland decreased by 18.65% and 15.13%, respectively in the whole study periods (Table 16). According to focus group discussion with local people, the majority of the study area was covered by dense vegetation some years before but now it has been declining at an alarming rate due to illegal logging, unsustainable exploitation of the forest resources for fuelwood, house construction and charcoal production as well as the occurrence of frequent forest fire. Belay Tegene (2002) has shown that the major cause for the rapid decline of the shrubland was the use of woody vegetation for firewood and charcoal production in Derekolli catchment of South Wollo. Illegal wood cutting and timber logging for sawmills industry were the major cause for forest cover decline in the central rift valley of Ethiopia (Gessesse Dessie and Christiansson, 2008).

5.1.8.2 Driving forces of land use/land cover change in the study area

Land use/land cover change is the result of a number of natural and anthropogenic factors. It is found to be the main threat to terrestrial ecosystems. Habitat loss and fragmentation due to LULCC coupled with climate change will remain a major issue of biodiversity conservation in the twenty-first century (Lovejoy and Hannah, 2005; Olson *et al.*, 2008). In Bore-Anferara-Wadera forest, the main causes of the observed rapid land use/land cover change over the last 28 years include rapid population growth, deforestation, land tenure system and forest fire.

Population growth

The total population of the three study districts was 253,463 in 1994 (CSA, 1996) and it increases to 373,870 in 2007 (CSA, 2007). Based on the 1994 population, on average, 9262 people were added per year to reach 373,870 in 2007, indicating a high rate of population growth in the study area. There has been a little adjustment of family size since children are considered an asset in the struggle for survival, as well as a security in old age. Therefore, population pressure was certainly the most important driving force for the observed land use/land cover dynamics. According to Pahari & Marai (1999), population pressure is often considered to be an important driver of deforestation. This is because population growth increases the demand for land for cultivation and settlement, trees for fuel wood and charcoal production as well as construction materials. Among others, Kibrom Tekle and Hedlund (2000), Wondeamlak Bewket (2002), Messay Mulugeta (2011), Tegegne Sishaw and Aklilu Amsalu (2012) and Yohannes Kidane *et al.* (2012) have shown that population growth is a major driving force in land use/land cover changes and that it has contributed to land degradation. Environmental consequences of population growth thus have been a reduction in fallow periods and soil exhaustion, cultivation of shallow soils and steep slopes followed by accelerated erosion and over-exploitation of forests and shrublands (Abate Shiferaw, 2011).

People coming from other areas due to resettlement as well as those coming from adjacent areas with different farming practices severely affected the natural vegetation of the study area. These people cut the forests to get land for farming and house construction and practicing shifting cultivation, slash burning, and poor land administration while the native people were growing crops in small pieces of land without clearing the forests and taking care of the natural vegetation. Hurni *et al.* (2005) also noted the effect of people coming from the adjacent area on land use/land cover change in the upper Nile Basin of Ethiopia. Similarly, Ouedraogo *et al*. (2012) indicated that migrants had larger farmlands and used environmentally harmful practices (shifting cultivation, slash and burning techniques) in their land use systems while the native people tended to take more care of land and environment by intensifying the production within the same croplands instead of cutting forest to make space for new croplands in Southern Burkina Faso.

Deforestation

Deforestation is one of the leading driving forces of biodiversity loss (MEA, 2005). According to discussions with experts from Borena-Guji Forest and Wildlife Enterprise, illegal timber logging, charcoal production, and firewood collection have increased in recent years due to the high demand of these forest products in the nearby towns and villages. Poor farmers and landless youth are largely involved in the illegal cutting of trees for various purposes to generate household income. Hence, an increasing demand for forest products, in the form of timber, construction materials, firewood and charcoal within and outside the study area has resulted in land use/land cover change in the study area. The drastic decline of forest and shrubland in this study can also be attributed to an extensive use of woody vegetation for the above-mentioned forest products. Meseret Mideksa (2009) reported that an increasing demand of forest products, in the form of firewood and charcoal production within and outside the forest area was the drivers of deforestation in Adaba-Dodola Forest Priority Area. Similarly, Eyayu Molla *et al.* (2010) also reported that illegal timber logging, firewood collection, and charcoal production from the natural forests to generate household income was the factor for the natural vegetation cover decline in northwestern Ethiopia.

Even though Ethiopia has adopted a forest policy in April 2007 to conserve and develop forest resources for sustainable supply of forest products to satisfy the demands of the society and contribute to the development of the national economy, there is a weak state control over most of the regional state forests giving opportunity that leads to an open access regime which in turn created unbalanced utilization where harvest exceeds natural regeneration, thereby, resulting in reduction of forest coverage every year (Sisay Nune, 2007). This weak enforcement of forest laws and policies aggravated illegal timber logging in the study area resulting in forest destruction. Mwavu and Witkowski (2008) and Bernard *et al*. (2010) also reported that weak environment laws and policies implementation was the basic driver of land use/land cover change in northwestern and southwestern Uganda.

Land tenure system

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 (Badege Bishaw, 2001). Although the current government offer land use right to the rural community, a large number of people coming from other areas acquired land informally in the study area and lack confidence to rehabilitate their land resources in fear of the possible confiscation (personal communication with elder farmers). This has resulted in ecological damage, inappropriate or over-intensive land use and poor land management practices that accelerate land use/land cover change. The lack of appropriate land use policies was taken as a major reason for land use/land cover changes and land degradation in different parts of Ethiopia

(Gete Zeleke and Hurni, 2001; Belay Tegene, 2002; Eyayu Molla *et al.*, 2010; Abate Shiferaw, 2011; Messay Mulugeta, 2011; Solomon Belay *et al.*, 2014).

Forest fire

According to focus group discussions with local people and experts from Borena-Guji Forest and Wildlife Enterprise, a forest fire occurred in all time intervals and a common phenomenon in the study area. For example, it occurred in the years 1992, 2000, 2004 and 2011 destroying thousands of hectares of forest in the area. Nowadays, fires are mainly of anthropogenic origin. They are lit intentionally to clear forests for expansion of agricultural land, settlement, and charcoal production as well as accidentally after honey gathering (Sillero-Zubiri and Macdonald, 1997; Wesche *et al*., 2000). The incidence of fire is also common in the Bale Mountains. Yohannes Kidane *et al*. (2012) reported that forest fire is frequent in the Afromontane shrubs and herbaceous formations located on the eastern slopes of the Bale Mountains resulting in land use/land cover change.

5.1.8.3 Implications of land use/land cover change in the study area

The observed LULC changes did not take place without negative consequences. For example, as agricultural land and built up area expanded at the expense of other land use and land cover types, forest and shrubland severely declined, resulting in the destruction of various plant species and loss of habitat for wild animals. The increasing loss of natural forest covers in Bore-Anferara-Wadera forest results in the decline of biodiversity. According to BLI (2012), the study area is part of the Eastern Afromontane Biodiversity hotspot region of the Ethiopian highlands which contains high diversity and composition of plant and animal species including endemic plants that require high conservation priority. The ongoing forest and shrubland conversion in the study area could directly contribute to the loss of high conservation priority plant and animal species. The rapid population explosion in the study area also put additional pressure on the remaining natural forest and shrubland areas resulting in complete destruction of the resources unless critical measures have been taken. In addition, the ongoing pressure on the natural forests supports expectations of Ciais *et al.* (2009) who predicted that the African forest carbon stocks would remain vulnerable. Such dynamics could change the carbon stocks and release a substantial amount of carbon from the forest to the atmosphere (IPCC, 2000).

Land degradation could be partly explained by the removal of land cover by deforestation and overgrazing. Land cover is one of the major factors that determine the rate of soil loss due to erosion (Wondeamlak Bewket, 2002). Land use/land cover change could also affect soil quality or soil intactness (the ability of soils to stay in place) and increase the risk of erosion and flooding (Abate Shiferaw, 2011). Discussions with local farmers also confirmed the presence of soil erosion due to flooding on their farm plots during rainy seasons. Small rills observed in the cultivated lands during the field visits of 2014 also approve the presence of soil erosion. Such negative changes in the land use history result in shallow soil depth, loss of soil fertility and a decline in crop productivity affecting the livelihood of the community. Mulugeta Demelash and Star (2010) have shown that soil erosion reduces root depth, removes soil organic matter and nutrients, and decreases water-holding capacity of the soil resulting in the decline of land productivity in northwestern highlands of Ethiopia.

5.2 Conclusion

Bore-Anferara-Wadera forest is one of the remnant vegetation in southern Ethiopia. Anferara forest is also one of the Eastern Afromontane Key Biodiversity Areas which contain vulnerable species of animals and plants.

This study has shown the presence of high plant diversity, species composition, and richness in the study area which might be attributed to habitat heterogeneity and availability of suitable environmental gradients that suit different plant species associations. Although a low number of endemic plant species is recorded, the number of newly recorded plant species in the floristic region is relatively large indicating further botanical exploration would result in additional new records in the floristic region. Four plant community types were identified in this study. The patterns of plant species distribution and plant community formation is mainly influenced by altitude and slope among other environmental factors.

Analysis of vegetation structure has shown the predominance of small-sized individuals in the lower DBH and height classes with gradual decline towards the higher classes indicating good reproduction potential of the forest. Such a trend is probably caused by past disturbance of the original vegetation resulting in a stage of secondary development. The population structure of representative tree and shrub species also indicates that some species have abnormal population structure with no or few individuals at different size classes due to selective removal of preferred sized individuals by the local people for different purposes. Moreover, assessment of the regeneration status of the forest reveals that a large number of species are represented by few mature plants only without any sapling and seedling stage indicating that these species are on the verge of local extinction, demanding high conservation priority.

Bore-Anferara-Wadera forest is floristically similar with Belete and Magada forests due to the similarity in altitudinal ranges, climatic conditions as well as geographical proximity, but in general it exhibits relatively better floristic similarity with moist Afromontane forests than with the dry Afromontane forests of Ethiopia.

The results of LULC image classification have indicated that shrubland and forest land were the dominant LULC types in the initial study year (1986), whereas agricultural land was the dominant LULC type followed by others in various proportions in 2014. Change detection analysis for the study periods has shown a continuous expansion of agricultural land and built up area at the expense of mainly forest and shrubland. Forest and shrubland accounted for 35.19% and 38.36% of the study area in 1986, declined to 29.03% and 30.28% in 2000 and further dropped to 20.06% and 19.71% in 2014, respectively. These figures suggest that forest and shrubland were declining at the rate of 0.44% and 0.58% per year between 1986 and 2000 and 0.64% and 0.78% per year between 2000 and 2014, indicating severe destruction of the vegetation cover of the area. The other land use/land cover types have revealed minor expansion over the study periods. Rapid population growth, deforestation, land tenure system and forest fire were the major diving forces for the observed land use/land cover change over the last 28 years in the study area.

5.3 Recommendations

The results of this work show that the forest is still under disturbance and the high level of dependency of the local people for the forest products and the threat on the forest due to increasing population pressure and the demand for more land; it would be very difficult to stop anthropogenic disturbances on the forest. Therefore, the following recommendations are made for effective conservation and sustainable utilization of the forest resources:

1. Considerable numbers of tree and shrub species in the forest have shown abnormal population structure as well as poor regeneration status. To prevent local extinction of these species, the involvement of many stakeholders including the local government authorities, the local communities and NGO's in establishing nursery sites to grow and plant indigenous tree species is very important and appropriate. In addition, prevention of livestock grazing in the forest would promote the natural regeneration process of many species.

2. Land use/land cover dynamics data should be used as the main input during land use planning for sustainable land use management and resource allocation in the study area in the future.

3. Carry out further studies on ethnobotany to document the indigenous knowledge on the use of medicinal plants to treat various types of human and livestock diseases, soil seed bank to assess the natural regeneration potential of species as

well as the relationship with above ground vegetation and carbon sequestration to evaluate the amount of carbon absorbed by the forest to mitigate climate change.

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APPENDICES

Appendix 1 Floristic list of Bore-Anferara-Wadera forest, Southern Ethiopia

No	**Scientific name**	**Family**	**Habit***	**Local name (Oromifa)**
1	*Acacia abyssinica* Hochst. ex Benth.	Fabaceae	T	Ambo
2	*Acacia bussei* Harms ex Sjostedt	Fabaceae	T	Hallo
3	*Acanthus eminens* C. B. Clarke	Acanthaceae	S	Gogodu
4	*Achyranthes aspera* L.	Amaranthaceae	H	Darguu
5	*Acokanthera schimperi* (A. DC.) Schweinf.	Apocynaceae	T	Qereruu
6	*Agarista salicifolia* (Comm. ex Lam.) Hook. f.	Ericaceae	T	Gerii
7	*Albizia gummifera* (J.F. Gmel.) C.A. Sm.	Fabaceae	T	Gerbi
8	*Allophylus abyssinicus* (Hochst.) Radlk.	Sapindaceae	T	Sareji
9	*Apodytes dimidiata* E. Mey. ex Arn.	Icacinaceae	T	Meiae
10	*Arundinaria alpina* K. Schum.	Poaceae	S	Lamae
11	*Asparagus africanus* Lam.	Asparagaceae	L	Hidoo
12	*Asplenium protensum* Schrad.	Aspleniaceae	H	Baldoo

13	*Bersama abyssinica* Fresen.	Melianthaceae	T	Tibro
14	*Bridelia micrantha* (Hochst.) Baill.	Euphorbiaceae	T	Bobeeaa
15	*Brucea antidysenterica* J. F. Mill.	Simaroubaceae	S	Guduba-lafaa
16	*Calpurnia aurea* (Ait.) Benth.	Fabaceae	S	Cheketa
17	*Carex chlorosaccus C.B.Clarke*	Cyperaceae	H	Qunii
18	*Carissa spinarum* L.	Apocynaceae	S	Agamsa
19	*Celtis africana* Burm. f.	Ulmaceae	T	Moto koma
20	*Celtis toka* (Forssk.) Hepper & Wood	Ulmaceae	T	Kelkelcha
21	*Chionanthus mildbraedii* (Gilg & Schellenb.) Stearn	Oleaceae	S	Wolichu
22	*Christella gueinziana* (Mett.) Holttum	Thelypteridaceae	H	Terchaa
23	*Clematis hirsuta* Perr. & Guill	Ranunculaceae	L	Feetii
24	*Clutia abyssinica* Jaub. & Spach.	Euphorbiaceae	S	Kutaadigaa
25	*Clutia lanceolata* Forssk.	Euphorbiaceae	S	Gurii
26	*Coffea arabica* L.	Rubiaceae	S	Buna
27	*Combretum molle* R. Br. ex G. Don	Combretaceae	T	Rukensaa
28	*Cordia africana* Lam.	Boraginaceae	T	Wodessaa

29	*Croton macrostachyus* Del.	Euphorbiaceae	T	Mokonisa
30	*Cyathula uncinulata* (Schrad.) Schinz	Amaranthaceae	H	Naenkoo
31	*Dalbergia lactea* Vatke	Fabaceae	L	Tirroo
32	*Desmodium repandum* (Vahl) DC.	Fabaceae	H	Dangie
33	*Dodonea angustifolia* L. f.	Sapindaceae	S	Itiecha
34	*Dombeya torrida* (J.F. Gmel.) P. Bamps	Sterculiaceae	S	Danigola
35	*Dracaena afromontana* Mildbr.	Dracaenaceae	T	Worko
36	*Ehretia cymosa* Thonn.	Boraginaceae	T	Uraga
37	*Elaeodendron buchananii* (Loes.) Loes.	Celastraceae	T	Tillo
38	*Embelia schimperi* Vatke	Myrsinaceae	L	Haanquu
39	*Erythrococca abyssinica* Pax	Euphorbiaceae	S	Dugoo
40	*Erythrococca trichogyne* (Muell Arg.) Prain	Euphorbiaceae	S	Fuoo
41	*Euclea racemosa* Murr. subsp. *schimperi* (A. DC.) White	Ebenaceae	S	Miaesaa
42	*Fagaropsis angolensis* (Engl.) Dale	Rutaceae	T	Siisaa
43	*Ficus sur* Forssk.	Moraceae	T	Haru
44	*Ficus thonningii* Blume	Moraceae	S	Denbii

45	*Flacourtia indica* (Burm. f) Merr.	Flacourtiaceae	S	Hagela
46	*Galiniera saxifraga* (Hochst.) Bridson	Rubiaceae	S	Kudumi
47	*Gnidia stenophylla* Gilg	Thymelaeaceae	H	Hantichaa
48	*Gouania longispicata* Engl.	Rhamnaceae	L	Subena
49	*Grewia ferruginea* Hochst. ex A. Rich.	Tiliaceae	S	Xaaxessaa
50	*Hagenia abyssinica* (Brace) J.F. Gmel.	Rosaceae	T	Hitto
51	*Halleria lucida* L.	Scrophulariaceae	S	Qertatenaa
52	*Hippocratea africana* (Willd.) Loes.	Celastraceae	L	Titiitaa
53	*Hippocratea goetzei* Loes.	Celastraceae	L	Arayee
54	*Hippocratea pallens* Planch. ex Oliver	Celastraceae	L	Teruu
55	*Hypericum revolutum* Vahl	Guttiferae	S	Edera
56	*Hypoestes forskaolii* (Vahl) R. Br.	Acanthaceae	H	Xenii
57	*Ilex mitis* (L.) Radlk.	Aquifoliaceae	T	Hangadii
58	*Indigofera hochstetteri* Bak.	Fabaceae	H	Errie
59	*Isoglossa somalensis* Lindau	Acanthaceae	H	Darguu
60	*Jasminum abyssinicum* Hochst. ex DC.	Oleaceae	L	Dikii, Xookonu

61	*Jasminum grandiflorum* L. subsp. *floribundum* (R.Br. ex Fresen.) P.S. Green	Oleaceae	L	Dekicha
62	*Kotschya recurvifolia* (Taub.) F. White**	Fabaceae	S	Unknown
63	*Laggera crispata* (Vahl) Hepper & Wood	Asteraceae	H	Saesikoo
64	*Landolphia buchananii* (Hall.f.) Stapf	Apocynaceae	L	Hoppii
65	*Leonotis ocymifolia* (Burm. f.) Iwarsson	Lamiaceae	S	Xutiyee
66	*Lepidotrichilia volkensii* (Gürke) Leroy	Meliaceae	T	Sakoro
67	*Lippia adoensis* Hochst. ex Walp.**	Verbenaceae	S	Udoo
68	*Lobelia giberroa* Hemsl.	Lobeliaceae	S	Merangaa
69	*Macaranga capensis* (Baill.) Sim	Euphorbiaceae	T	Worayu
70	*Maesa lanceolata* Forssk.	Myrsinaceae	T	Abbayyi
71	*Maytenus addat* (Loes.) Sebsebe**	Celastraceae	S	Kombolcha
72	*Maytenus arbutifolia* (A. Rich.) Wilczek	Celastraceae	S	Kombolcha
73	*Maytenus undata* (Thunb.) Blakelock	Celastraceae	S	Akooluu
74	*Microglossa pyrifolia* (Lam.) Kuntze	Asteraceae	S	Gubaa
75	*Millettia ferruginea* (Hochst.) Bak. subsp. *darassana* (Cuf.) Gillett **	Fabaceae	T	Dadaatu

76	*Mimusops kummel* A. DC.	Sapotaceae	T	Bururii
77	*Momordica foetida* Schumach.	Cucurbitaceae	L	Surupaa
78	*Myrsine africana* L.	Myrsinaceae	S	Eerii
79	*Nuxia congesta* R. Br. ex Fresen.	Loganiaceae	T	Udessa, Irba
80	*Ochna holstii* Engl	Ochnaceae	S	Baltikessa
81	*Ocotea kenyensis* (Chiov.) Robyns & Wilczek	Lauraceae	T	Deressa
82	*Olea capensis* L. subsp. *macrocarpa* (C.A. Wright.) Verdc.	Oleaceae	T	Gegema
83	*Olea europaea* L. subsp. *cuspidata* (Wall. ex G.Don) Cif.	Oleaceae	S	Ejersaa
84	*Olea welwitschii* (Knobl.) Gilg & Schellenb.	Oleaceae	T	Sawa
85	*Olinia rochetiana* A. Juss.	Oliniaceae	T	Qediidaa
86	*Oncoba spinosa* Forssk.	Flacourtiaceae	S	Akoku
87	*Pavetta abyssinica* Fresen.	Rubiaceae	S	Bunuti
88	*Pavetta oliveriana* Hiern	Rubiaceae	S	Taamee
89	*Pellaea quadripinnata* (Forssk.) Prantl	Sinopteridaceae	H	Tercha
90	*Peponium vogelii* (Hook.f.) Engl.	Cucurbitaceae	L	Surupa
91	*Periploca linearifolia* Quart.-Dill. & A. Rich.	Asclepiadaceae	L	Hirkebeta

92	*Phoenix reclinata* Jacq.	Arecaceae	T	Maexii
93	*Phyllanthus ovalifolius* Forssk.	Euphorbiaceae	S	Unknown
94	*Phytolacca dodecandra* L 'Herit.	Phytolaccaceae	S	Haranja
95	*Pittosporum viridiflorum* Sims	Pittosporaceae	T	Aaraa
96	*Podocarpus falcatus* (Thunb.) R. B. ex Mirb.	Podocarpaceae	T	Birbirsa
97	*Poecilostachys oplismenoides* (Hack.) W. D. Clayton	Poaceae	H	Daaffa
98	*Polyscias fulva* (Hiern) Harms	Araliaceae	T	Tella
99	*Pouteria adolfi-friederici* (Engl.) Baehni	Sapotaceae	T	Guduba
100	*Premna schimperi* Engl.	Lamiaceae	S	Xulengie
101	*Protea gaguedi* J. F. Gmel.	Proteaceae	S	Ayudoo
102	*Prunus africana* (Hook. f.) Kalkm.	Rosaceae	T	Suke
103	*Psychotria orophila* Petit	Rubiaceae	S	Wontifulas, Buna boye
104	*Psydrax schimperiana* (A. Rich.) Bridson	Rubiaceae	T	Galo
105	*Pteridium aquilinum* (L.) Kuhn subsp. *capense* (Thunb.) C.Chr.	Hypolepidaceae	H	Trechaa
106	*Pterolobium stellatum* (Forssk.) Brenan	Fabaceae	S	Kejiimaa

107	*Rhamnus prinoides* L'Herit.	Rhamnaceae	S	Gesho
108	*Rhoicissus tridentata* (L.f.) Wild & Drummond	Vitaceae	L	Laluu
109	*Ricinus communis* L.	Euphorbiaceae	S	Koboo
110	*Ritchiea albersii* Gilg	Capparidaceae	S	Unknown
111	*Rosa abyssinica* Lindley	Rosaceae	S	Gora
112	*Rytigynia neglecta* (Hiern) Robyns	Rubiaceae	S	Mikie
113	*Schefflera abyssinica* (Hochst. ex A. Rich.) Harms	Araliaceae	T	Geteme, Orone
114	*Schrebera alata* (Hochst.) Welw.	Oleaceae	S	Tuwamie
115	*Scolopia theifolia* Gilg	Flacourtiaceae	T	Keraa
116	*Scutia myrtina* (Burm.f.) Kurz	Rhamnaceae	S	Bechoo
117	*Senna didymobotrya* (Fresen.) Irwin & Barneby	Fabaceae	S	Asenemeka
118	*Setaria megaphylla* (Steud.) Th. Dur. & Schinz	Poaceae	H	Sokooruu
119	*Smilax aspera* L.	Smilacaceae	L	Seritii
120	*Solanum anguivi* Lam.	Solanaceae	S	Hadiihallala
121	*Solanum marginatum* L. f.	Solanaceae	S	Hidii
122	*Syzygium guineense* (Willd.) DC subsp. *afromontanum* F. White	Myrtaceae	T	Bedessa

123	*Tacazzea conferta* N.E. Br.	Asclepiadaceae	L	Anenotuu
124	*Teclea nobilis* Del.	Rutaceae	S	Hadheessaa
125	*Tectaria gemmifera* (Fee) Alston	Dryopteridaceae	H	Terchaa
126	*Terminalia schimperiana* Hochst.	Combretaceae	T	Debekaa
127	*Thalictrum rhynchocarpum* Dill. & A. Rich.	Ranunculaceae	H	Hachercha
128	*Toddalia asiatica* (L.) Lam.	Rutaceae	S	Goraa-oukaa
129	*Trichilia emetica* Vahl	Meliaceae	T	Anuunuu
130	*Urera hypselodendron* (A. Rich.) Wedd.	Urticaceae	L	Hajiijaa
131	*Vepris dainellii* (Pic.Serm.) Kokwaro**	Rutaceae	S	Arabe
132	*Vernonia amygdalina* Del.	Asteraceae	S	Ebicha
133	*Vernonia rueppellii* Sch. Bip. ex Walp.**	Asteraceae	S	Rejii
134	*Vernonia unionis* Sch. Bip. ex. Walp.	Asteraceae	H	Soyemaa
135	*Zanthoxylum chalybeum* Engl.	Rutaceae	S	Gedaa
136	*Zanthoxylum usambarense* (Engl.) Kokwaro	Rutaceae	T	Gedaa

* Habit T = tree, S = shrub, L = Liana, H = herb

** Endemic species

Appendix 2 Density of trees and shrubs with DBH > 2 cm, 10 cm and 20 cm in Bore-Anferara-Wadera forest

Species name	**DBH > 2**		**DBH > 10**		**DBH > 20**	
	Density	**%**	**Density**	**%**	**Density**	**%**
Acacia abyssinica Hochst. ex Benth.	0.10	0.01	0.10	0.02	0.10	0.04
Acacia bussei Harms ex Sjostedt	0.79	0.08	0.79	0.16	0.40	0.16
Acanthus eminens C.B. Clarke	15.08	1.44	0.00	0.00	0.00	0.00
Acokanthera schimperi (A. DC.) Schweinf.	3.87	0.37	1.98	0.40	0.79	0.32
Agarista salicifolia (Comm. ex Lam.) Hook. f.	0.10	0.01	0.10	0.02	0.10	0.04
Albizia gummifera (J.F. Gmel.) C.A. Sm.	1.19	0.11	0.99	0.20	0.79	0.32
Allophylus abyssinicus (Hochst.) Radlk.	8.13	0.78	6.65	1.33	4.96	2.01
Apodytes dimidiata E. Mey. ex Arn.	7.64	0.73	5.26	1.05	2.78	1.13
Arundinaria alpina K. Schum.	0.10	0.01	0.00	0.00	0.00	0.00
Bersama abyssinica Fresen.	9.72	0.93	2.08	0.42	0.99	0.40
Bridelia micrantha (Hochst.) Baill.	2.08	0.20	1.49	0.30	1.19	0.48
Brucea antidysenterica J. F. Mill.	0.10	0.01	0.00	0.00	0.00	0.00
Calpurnia aurea (Ait.) Benth.	3.27	0.31	0.79	0.16	0.00	0.00

Carissa spinarum L.	2.08	0.20	0.00	0.00	0.00	0.00
Celtis africana Burm. f.	9.33	0.89	4.86	0.97	2.18	0.88
Celtis toka (Forssk.) Hepper & Wood	1.19	0.11	1.19	0.24	0.60	0.24
Chionanthus mildbraedii (Gilg & Schellenb.) Stearn	18.15	1.73	3.27	0.65	0.40	0.16
Clutia abyssinica Jaub. & Spach.	0.20	0.02	0.00	0.00	0.00	0.00
Coffea arabica L.	0.60	0.06	0.00	0.00	0.00	0.00
Combretum molle R. Br. ex G. Don	0.10	0.01	0.10	0.02	0.10	0.04
Cordia africana Lam.	1.29	0.12	1.19	0.24	0.79	0.32
Croton macrostachyus Del.	19.35	1.85	17.76	3.55	13.49	5.47
Dodonea angustifolia L. f.	0.10	0.01	0.00	0.00	0.00	0.00
Dombeya torrida (J.F. Gmel.) P. Bamps	0.10	0.01	0.00	0.00	0.00	0.00
Dracaena afromontana Mildbr.	143.55	13.70	59.33	11.86	9.62	3.90
Ehretia cymosa Thonn.	12.80	1.22	7.84	1.57	2.98	1.21
Elaeodendron buchananii (Loes.) Loes.	25.79	2.46	12.30	2.46	2.28	0.93
Erythrococca trichogyne (Muell Arg.) Prain.	6.55	0.63	0.89	0.18	0.00	0.00
Euclea racemosa Murr. subsp. *schimperi* (A. DC.) White	0.10	0.01	0.00	0.00	0.00	0.00

Fagaropsis angolensis (Engl.) Dale	7.84	0.75	2.98	0.59	1.39	0.56
Ficus sur Forssk.	9.92	0.95	6.94	1.39	3.97	1.61
Ficus thonningii Blume	0.50	0.05	0.30	0.06	0.10	0.04
Flacourtia indica (Burm. f.) Merr.	0.10	0.01	0.10	0.02	0.00	0.00
Galiniera saxifraga (Hochst.) Bridson	9.03	0.86	3.57	0.71	0.40	0.16
Grewia ferruginea Hochst. ex A. Rich.	0.60	0.06	0.30	0.06	0.00	0.00
Hagenia abyssinica (Brace) J.F. Gmel.	0.40	0.04	0.30	0.06	0.30	0.12
Halleria lucida L.	0.10	0.01	0.00	0.00	0.00	0.00
Hypericum revolutum Vahl	0.10	0.01	0.00	0.00	0.00	0.00
Ilex mitis (L.) Radlk.	8.73	0.83	0.99	0.20	0.00	0.00
Lepidotrichilia volkensii (Gürke) Leroy	40.87	3.90	22.02	4.40	1.69	0.68
Lobelia giberroa Hemsl.	0.20	0.02	0.00	0.00	0.00	0.00
Macaranga capensis (Baill.) Sim	44.05	4.20	36.31	7.26	23.41	9.49
Maesa lanceolata Forssk.	5.75	0.55	1.98	0.40	0.30	0.12
Maytenus addat (Loes.) Sebsebe	1.88	0.18	0.79	0.16	0.50	0.20
Maytenus arbutifolia (A. Rich.) Wilczek	3.67	0.35	0.89	0.18	0.60	0.24

Maytenus undata (Thunb.) Blakelock	0.79	0.08	0.00	0.00	0.00	0.00
Microglossa pyrifolia (Lam.) Kuntze	0.10	0.01	0.00	0.00	0.00	0.00
Millettia ferruginea (Hochst.) Bak.	10.12	0.97	5.65	1.13	2.38	0.97
Mimusops kummel A. DC.	3.47	0.33	1.79	0.36	0.20	0.08
Nuxia congesta R. Br. ex Fresen.	8.53	0.81	3.67	0.73	1.29	0.52
Ochna holstii Engl.	0.99	0.09	0.30	0.06	0.00	0.00
Ocotea kenyensis (Chiov.) Robyns & Wilczek	71.53	6.83	50.00	9.99	32.94	13.35
Olea capensis L. subsp. *macrocarpa* (C.A. Wright.) Verdc.	30.65	2.93	24.11	4.82	17.76	7.20
Olea europaea L. subsp. *cuspidata* (Wall. ex G.Don) Cif.	0.10	0.01	0.00	0.00	0.00	0.00
Olea welwitschii (Knobl.) Gilg & Schellenb.	1.19	0.11	0.60	0.12	0.40	0.16
Olinia rochetiana A. Juss.	0.69	0.07	0.50	0.10	0.40	0.16
Oncoba spinosa Forssk.	1.19	0.11	0.00	0.00	0.00	0.00
Pavetta abyssinica Fresen.	0.40	0.04	0.00	0.00	0.00	0.00
Pavetta oliveriana Hiern	1.09	0.10	0.60	0.12	0.30	0.12
Phoenix reclinata Jacq.	1.29	0.12	1.19	0.24	0.30	0.12
Phyllanthus ovalifolius Forssk.	0.10	0.01	0.00	0.00	0.00	0.00

Phytolacca dodecandra L 'Herit.	0.10	0.01	0.10	0.02	0.10	0.04
Pittosporum viridiflorum Sims	1.09	0.10	0.50	0.10	0.00	0.00
Podocarpus falcatus (Thunb.) R. B. ex Mirb.	17.46	1.67	9.03	1.80	5.85	2.37
Polyscias fulva (Hiern) Harms	9.52	0.91	8.53	1.71	6.35	2.57
Pouteria adolfi-friederici (Engl.) Baehni	56.85	5.43	31.55	6.31	21.13	8.57
Premna schimperi Engl.	2.08	0.20	0.30	0.06	0.00	0.00
Protea gaguedi J. F. Gmel.	0.10	0.01	0.10	0.02	0.10	0.04
Prunus africana (Hook. f.) Kalkm.	10.22	0.98	7.84	1.57	6.45	2.61
Psychotria orophila Petit	83.53	7.97	8.04	1.61	0.60	0.24
Psydrax schimperiana (A. Rich.) Bridson	38.00	3.63	18.85	3.77	7.94	3.22
Pterolobium stellatum (Forssk.) Brenan	0.99	0.09	0.00	0.00	0.00	0.00
Rhamnus prinoides L'Herit.	0.20	0.02	0.00	0.00	0.00	0.00
Ricinus communis L.	0.10	0.01	0.00	0.00	0.00	0.00
Ritchiea albersii Gilg	0.10	0.01	0.10	0.02	0.10	0.04
Rosa abyssinica Lindley	0.69	0.07	0.00	0.00	0.00	0.00
Rytigynia neglecta (Hiern) Robyns	2.18	0.21	0.00	0.00	0.00	0.00

Schefflera abyssinica (Hochst. ex A. Rich.) Harms	0.69	0.07	0.40	0.08	0.30	0.12
Schrebera alata (Hochst.) Welw.	0.10	0.01	0.00	0.00	0.00	0.00
Scolopia theifolia Gilg	42.96	4.10	16.96	3.39	3.17	1.29
Scutia myrtina (Burm.f.) Kurz	0.10	0.01	0.00	0.00	0.00	0.00
Senna didymobotrya (Fresen.) Irwin & Barneby	0.10	0.01	0.00	0.00	0.00	0.00
Syzygium guineense (Willd.) DC subsp. *afromontanum* F. White	63.39	6.05	58.33	11.66	52.68	21.36
Teclea nobilis Del.	117.76	11.24	26.88	5.37	2.68	1.09
Terminalia schimperiana Hochst.	1.29	0.12	1.29	0.26	1.19	0.48
Toddalia asiatica (L.) Lam	0.40	0.04	0.00	0.00	0.00	0.00
Trichilia emetica Vahl	3.37	0.32	3.08	0.61	2.08	0.84
Vepris dainellii (Pic.Serm.) Kokwaro	33.73	3.22	13.39	2.68	2.78	1.13
Vernonia amygdalina Del.	0.10	0.01	0.00	0.00	0.00	0.00
Vernonia rueppellii Sch. Bip. ex Walp.	0.69	0.07	0.10	0.02	0.00	0.00
Zanthoxylum usambarense (Engl.) Kokwaro	0.20	0.02	0.10	0.02	0.00	0.00
Total	1047.52	100.00	500.30	100.00	246.63	100.00

Appendix 3 Distribution of trees and shrubs per hectare across DBH classes in Bore-Anferara-Wadera forest

Species name	**DBH class (cm)**							
	2.1 - 10	**10.1 - 20**	**20.1 - 50**	**50.1 - 80**	**80.1 - 110**	**110.1 - 140**	**> 140**	**%**
Acacia abyssinica Hochst. ex Benth.			0.10					0.01
Acacia bussei Harms ex Sjostedt		0.40	0.40					0.08
Acanthus eminens C.B. Clarke	10.81	4.27						1.44
Acokanthera schimperi (A. DC.) Schweinf.	1.88	1.19	0.79					0.37
Agarista salicifolia (Comm. ex Lam.) Hook. f.			0.10					0.01
Albizia gummifera (J.F. Gmel.) C.A. Sm.	0.20	0.20	0.79					0.11
Allophylus abyssinicus (Hochst.) Radlk.	1.49	1.69	3.57	1.19	0.20			0.78
Apodytes dimidiata E. Mey. ex Arn.	2.28	2.48	1.98	0.69	0.10			0.72
Arundinaria alpina K. Schum.	0.10							0.01
Bersama abyssinica Fresen.	7.64	1.09	0.79	0.10	0.10			0.93
Bridelia micrantha (Hochst.) Baill.	0.60	0.30	1.19					0.20

Brucea antidysenterica J. F. Mill.	0.10							0.01
Calpurnia aurea (Ait.) Benth.	2.48	0.79						0.31
Carissa spinarum L.	2.08							0.20
Celtis africana Burm. f.	4.46	2.68	2.08	0.10				0.89
Celtis toka (Forssk.) Hepper & Wood		0.60	0.60					0.11
Chionanthus mildbraedii (Gilg & Schellenb.) Stearn	14.88	2.88	0.40					1.73
Clutia abyssinica Jaub. & Spach.	0.20							0.02
Coffea arabica L.	0.60							0.06
Combretum molle R. Br. ex G. Don			0.10					0.01
Cordia africana Lam.	0.10	0.40	0.79					0.12
Croton macrostachyus Del.	1.59	4.27	11.81	1.29	0.10			1.82
Dodonea angustifolia L. f.	0.10							0.01
Dombeya torrida (J.F. Gmel.) P. Bamps	0.10							0.01
Dracaena afromontana Mildbr.	84.23	49.70	7.84	1.09	0.40	0.10	0.10	13.71
Ehretia cymosa Thonn.	4.96	4.86	2.98					1.22

Elaeodendron buchananii (Loes.) Loes.	13.49	10.02	2.28					2.46
Erythrococca trichogyne (Muell Arg.) Prain.	5.65	0.89						0.63
Euclea racemosa Murr. subsp. *schimperi* (A. DC.) White	0.10							0.01
Fagaropsis angolensis (Engl.) Dale	4.86	1.59	1.39					0.75
Ficus sur Forssk.	2.98	2.98	2.38	0.50	0.50	0.10	0.20	0.92
Ficus thonningii Blume	0.20	0.20		0.10				0.05
Flacourtia indica (Burm. f.) Merr.		0.10						0.01
Galiniera saxifraga (Hochst.) Bridson	5.46	3.17	0.40					0.86
Grewia ferruginea Hochst. ex A. Rich.	0.30	0.30						0.06
Hagenia abyssinica (Brace) J.F. Gmel.	0.10		0.30					0.04
Halleria lucida L.	0.10							0.01
Hypericum revolutum Vahl	0.10							0.01
Ilex mitis (L.) Radlk.	7.74	0.99						0.83
Lepidotrichilia volkensii (Gürke) Leroy	18.75	20.34	1.69					3.90
Lobelia giberroa Hemsl.	0.20							0.02

Macaranga capensis (Baill.) Sim	7.74	12.90	19.05	3.87	0.30	0.10		4.20
Maesa lanceolata Forssk.	3.77	1.69	0.30					0.55
Maytenus addat (Loes.) Sebsebe	1.09	0.30	0.20	0.20	0.10			0.18
Maytenus arbutifolia (A. Rich.) Wilczek	2.78	0.30	0.60					0.35
Maytenus undata (Thunb.) Blakelock	0.79							0.08
Microglossa pyrifolia (Lam.) Kuntze	0.10							0.01
Millettia ferruginea (Hochst.) Bak.	4.46	3.27	2.18	0.20				0.97
Mimusops kummel A. DC.	1.69	1.59	0.20					0.33
Nuxia congesta R. Br. ex Fresen.	4.86	2.38	1.19	0.10				0.82
Ochna holstii Engl.	0.69	0.30						0.09
Ocotea kenyensis (Chiov.) Robyns & Wilczek	21.53	17.06	22.12	7.74	2.38	0.50	0.20	6.84
Olea capensis L. subsp. *macrocarpa* (C.A. Wright.) Verdc.	6.55	6.35	10.12	5.06	2.28	0.30		2.93
Olea europaea L. subsp. *cuspidata* (Wall. ex G.Don) Cif.	0.10							0.01
Olea welwitschii (Knobl.) Gilg &	0.60	0.20	0.40					0.11

Schellenb.								
Olinia rochetiana A. Juss.	0.20	0.10	0.40					0.07
Oncoba spinosa Forssk.	1.19							0.11
Pavetta abyssinica Fresen.	0.40							0.04
Pavetta oliveriana Hiern	0.50	0.30	0.30					0.10
Phoenix reclinata Jacq.	0.10	0.89	0.30					0.12
Phyllanthus ovalifolius Forssk.	0.10							0.01
Phytolacca dodecandra L 'Herit.			0.10					0.01
Pittosporum viridiflorum Sims	0.60	0.50						0.10
Podocarpus falcatus (Thunb.) R. B. ex Mirb.	8.43	3.17	3.77	0.99	0.60	0.50		1.67
Polyscias fulva (Hiern) Harms	0.99	2.18	2.58	2.68	0.89	0.20		0.91
Pouteria adolfi-friederici (Engl.) Baehni	25.30	10.42	7.94	4.76	3.57	2.68	2.18	5.43
Premna schimperi Engl.	1.79	0.30						0.20
Protea gaguedi J. F. Gmel.			0.10					0.01
Prunus africana (Hook. f.) Kalkm.	2.38	1.39	2.68	2.28	0.89	0.50	0.10	0.98

Psychotria orophila Petit	75.50	7.44	0.60					7.98
Psydrax schimperiana (A. Rich.) Bridson	19.15	10.91	7.64	0.20	0.10			3.63
Pterolobium stellatum (Forssk.) Brenan	0.99							0.09
Rhamnus prinoides L'Herit.	0.10							0.01
Ricinus communis L.	0.10							0.01
Ritchiea albersii Gilg			0.10					0.01
Rosa abyssinica Lindley	0.69							0.07
Rytigynia neglecta (Hiern) Robyns	2.18							0.21
Schefflera abyssinica (Hochst. ex A. Rich.) Harms	0.30	0.10	0.20	0.10				0.07
Schrebera alata (Hochst.) Welw.	0.10							0.01
Scolopia theifolia Gilg	25.99	13.79	3.17					4.11
Scutia myrtina (Burm.f.) Kurz	0.10							0.01
Senna didymobotrya (Fresen.) Irwin & Barneby	0.10							0.01
Syzygium guineense (Willd.) DC subsp. *afromontanum* F. White	5.06	5.65	17.86	19.64	10.32	3.37	1.49	6.06

Teclea nobilis Del.	90.87	24.21	2.58	0.10				11.25
Terminalia schimperiana Hochst.		0.10	1.19					0.12
Toddalia asiatica (L.) Lam	0.40							0.04
Trichilia emetica Vahl	0.30	0.99	0.69	0.50	0.69	0.20		0.32
Vepris dainellii (Pic.Serm.) Kokwaro	20.34	10.62	2.78					3.22
Vernonia amygdalina Del.	0.10							0.01
Vernonia rueppellii Sch. Bip. ex Walp.	0.60	0.10						0.07
Zanthoxylum usambarense (Engl.) Kokwaro	0.10	0.10						0.02

Appendix 4 Percentage distribution of trees and shrubs across height classes in Bore-Anferara-Wadera forest

Species name	**Height class (m)**						
	3 - 8	**8.1 - 13**	**13.1 - 18**	**18.1 - 23**	**23.1 - 28**	**28.1 - 33**	**> 33**
Acacia abyssinica Hochst. ex Benth.			0.01				
Acacia bussei Harms ex Sjostedt		0.05	0.02	0.01			
Acanthus eminens C.B. Clarke	0.03				0.02		
Acokanthera schimperi (A. DC.) Schweinf.	0.16	0.04	0.14	0.02			
Agarista salicifolia (Comm. ex Lam.) Hook. f.		0.01			0.03		
Albizia gummifera (J.F. Gmel.) C.A. Sm.	0.01	0.01	0.03	0.03	0.28		0.01
Allophylus abyssinicus (Hochst.) Radlk.	0.09	0.09	0.14	0.12	0.13	0.01	0.05
Apodytes dimidiata E. Mey. ex Arn.	0.21	0.07	0.18	0.09		0.05	0.01
Arundinaria alpina K. Schum.			0.01		0.06		
Bersama abyssinica Fresen.	0.74	0.09	0.02		0.02	0.01	0.02
Bridelia micrantha (Hochst.) Baill.	0.04	0.04	0.06	0.05			
Brucea antidysenterica J. F. Mill.	0.01						
Calpurnia aurea (Ait.) Benth.	0.23	0.10					

Carissa spinarum L.	0.19	0.01			0.09		
Celtis africana Burm. f.	0.30	0.10	0.28	0.09	0.02	0.02	0.01
Celtis toka (Forssk.) Hepper & Wood		0.03	0.04	0.03	0.01		
Chionanthus mildbraedii (Gilg & Schellenb.) Stearn	1.40	0.30	0.05	0.02			
Clutia abyssinica Jaub. & Spach.	0.02						
Coffea arabica L.	0.07						
Combretum molle R. Br. ex G. Don			0.01		0.01		
Cordia africana Lam.	0.01	0.02	0.07	0.02	0.54		
Croton macrostachyus Del.	0.19	0.15	0.41	0.33		0.06	0.17
Dodonea angustifolia L. f.	0.01						
Dombeya torrida (J.F. Gmel.) P. Bamps	0.01				0.05		
Dracaena afromontana Mildbr.	12.82	0.51	0.26	0.08	0.09	0.03	0.02
Ehretia cymosa Thonn.	0.51	0.31	0.28	0.05	0.16		
Elaeodendron buchananii (Loes.) Loes.	1.18	0.36	0.58	0.23			
Erythrococca trichogyne (Muell Arg.) Prain	0.56	0.07					

Euclea racemosa Murr. subsp. *schimperi* (A. DC.) White	0.01				0.03		
Fagaropsis angolensis (Engl.) Dale	0.43	0.08	0.12	0.12	0.19		0.01
Ficus sur Forssk.	0.20	0.13	0.16	0.09		0.09	0.09
Ficus thonningii Blume	0.03	0.01	0.02				
Flacourtia indica (Burm. f.) Merr.	0.01						
Galiniera saxifraga (Hochst.) Bridson	0.60	0.21	0.05	0.01			
Grewia ferruginea Hochst. ex A. Rich.	0.05	0.01			0.02		
Hagenia abyssinica (Brace) J.F. Gmel.		0.01	0.01				
Halleria lucida L.	0.01						
Hypericum revolutum Vahl	0.02				0.01		
Ilex mitis (L.) Radlk.	0.72	0.10	0.02				
Lepidotrichilia volkensii (Gürke) Leroy	2.56	1.11	0.24	0.01			
Lobelia giberroa Hemsl.	0.02				1.23		
Macaranga capensis (Baill.) Sim	0.49	0.48	0.89	0.59		0.30	0.23
Maesa lanceolata Forssk.	0.37	0.16	0.02		0.01		

Maytenus addat (Loes.) Sebsebe	0.09	0.06				0.02	0.01
Maytenus arbutifolia (A. Rich.) Wilczek	0.34	0.02	0.04	0.02			
Maytenus undata (Thunb.) Blakelock	0.09						
Microglossa pyrifolia (Lam.) Kuntze	0.01				0.09		
Millettia ferruginea (Hochst.) Bak.	0.42	0.20	0.15	0.04		0.07	0.02
Mimusops kummel A. DC.	0.15	0.12	0.05	0.01	0.02		
Nuxia congesta R. Br. ex Fresen.	0.45	0.16	0.14	0.03		0.01	0.01
Ochna holstii Engl.	0.09		0.03		1.52		
Ocotea kenyensis (Chiov.) Robyns & Wilczek	1.63	0.87	1.10	0.55	0.69	0.80	0.42
Olea capensis L. subsp. *macrocarpa* (C.A. Wright.) Verdc.	0.53	0.27	0.58	0.47		0.29	0.13
Olea europaea L. subsp. *cuspidata* (Wall. ex G.Don) Cif.	0.01				0.01		
Olea welwitschii (Knobl.) Gilg & Schellenb.	0.03	0.03	0.01	0.01	0.01	0.01	0.02
Olinia rochetiana A. Juss.	0.03		0.01	0.02			
Oncoba spinosa Forssk.	0.11	0.01					
Pavetta abyssinica Fresen.	0.04						

Pavetta oliveriana Hiern	0.05		0.05	0.01			
Phoenix reclinata Jacq.	0.01	0.07	0.05				
Phyllanthus ovalifolius Forssk.	0.03						
Phytolacca dodecandra L 'Herit.	0.01						
Pittosporum viridiflorum Sims	0.05	0.02	0.03	0.01	0.22		
Podocarpus falcatus (Thunb.) R. B. ex Mirb.	0.85	0.14	0.18	0.14	0.26	0.18	0.07
Polyscias fulva (Hiern) Harms	0.07	0.04	0.17	0.12	0.81	0.06	0.22
Pouteria adolfi-friederici (Engl.) Baehni	2.10	0.55	0.48	0.39		0.60	0.65
Premna schimperi Engl.	0.18	0.02					
Protea gaguedi J. F. Gmel.	0.01				0.20		
Prunus africana (Hook. f.) Kalkm.	0.26	0.03	0.07	0.09	0.01	0.21	0.16
Psychotria orophila Petit	7.42	0.47	0.15	0.01	0.31	0.01	
Psydrax schimperiana (A. Rich.) Bridson	1.64	0.66	0.72	0.28		0.03	0.02
Pterolobium stellatum (Forssk.) Brenan	0.01						
Rhamnus prinoides L'Herit.	0.02						
Ricinus communis L.	0.01						

Ritchiea albersii Gilg	0.01						
Rosa abyssinica Lindley	0.07						
Rytigynia neglecta (Hiern) Robyns	0.25						
Schefflera abyssinica (Hochst. ex A. Rich.) Harms	0.03	0.01	0.03				
Schrebera alata (Hochst.) Welw.	0.01				0.11		
Scolopia theifolia Gilg	1.88	0.98	0.94	0.24			
Scutia myrtina (Burm.f.) Kurz	0.01						
Senna didymobotrya (Fresen.) Irwin & Barneby	0.01				2.46		
Syzygium guineense (Willd.) DC subsp. *afromontanum* F. White	0.49	0.19	0.41	0.47	0.06	1.53	0.64
Teclea nobilis Del.	8.88	1.88	0.51	0.06		0.02	
Terminalia schimperiana Hochst.		0.06	0.05	0.02	0.05		
Trichilia emetica Vahl	0.04	0.05	0.03	0.04	0.02	0.06	0.07
Vepris dainellii (Pic.Serm.) Kokwaro	2.20	0.52	0.47	0.08			
Vernonia amygdalina Del.		0.01					
Vernonia rueppellii Sch. Bip. ex Walp.	0.07						

Zanthoxylum usambarense (Engl.) Kokwaro	0.02						

Appendix 5 Basal area (m^2 ha^{-1}) of trees and shrubs in Bore-Anferara-Wadera forest

Species name	**Basal area (m^2 ha^{-1})**	**Percent contribution**
Acacia abyssinica Hochst. ex Benth.	0.01	0.01
Acacia bussei Harms ex Sjostedt	0.03	0.04
Acanthus eminens C.B. Clarke	0.04	0.05
Acokanthera schimperi (A. DC.) Schweinf.	0.07	0.09
Agarista salicifolia (Comm. ex Lam.) Hook. f.	0.00	0.00
Albizia gummifera (J.F. Gmel.) C.A. Sm.	0.08	0.10
Allophylus abyssinicus (Hochst.) Radlk.	0.82	1.09
Apodytes dimidiata E. Mey. ex Arn.	0.53	0.71
Arundinaria alpina K. Schum.	0.00	0.00
Bersama abyssinica Fresen.	0.21	0.28
Bridelia micrantha (Hochst.) Baill.	0.09	0.12

Brucea antidysenterica J. F. Mill.	0.00	0.00
Calpurnia aurea (Ait.) Benth.	0.01	0.02
Carissa spinarum L.	0.00	0.01
Celtis africana Burm. f.	0.25	0.34
Celtis toka (Forssk.) Hepper & Wood	0.05	0.07
Chionanthus mildbraedii (Gilg & Schellenb.) Stearn	0.10	0.14
Clutia abyssinica Jaub. & Spach.	0.00	0.00
Coffea arabica L.	0.00	0.00
Combretum molle R. Br. ex G. Don	0.01	0.02
Cordia africana Lam.	0.05	0.07
Croton macrostachyus Del.	1.69	2.24
Dodonea angustifolia L. f.	0.00	0.00
Dombeya torrida (J.F. Gmel.) P. Bamps	0.00	0.00
Dracaena afromontana Mildbr.	2.59	3.44
Ehretia cymosa Thonn.	0.28	0.37
Elaeodendron buchananii (Loes.) Loes.	0.30	0.40

Erythrococca trichogyne (Muell Arg.) Prain	0.03	0.04
Euclea racemosa Murr. subsp. *schimperi* (A. DC.) White	0.00	0.00
Fagaropsis angolensis (Engl.) Dale	0.13	0.17
Ficus sur Forssk.	1.72	2.29
Ficus thonningii Blume	0.03	0.03
Flacourtia indica (Burm. f.) Merr.	0.00	0.00
Galiniera saxifraga (Hochst.) Bridson	0.10	0.13
Grewia ferruginea Hochst. ex A. Rich.	0.00	0.01
Hagenia abyssinica (Brace) J.F. Gmel.	0.03	0.04
Halleria lucida L.	0.00	0.00
Hypericum revolutum Vahl	0.00	0.00
Ilex mitis (L.) Radlk.	0.04	0.05
Lepidotrichilia volkensii (Gürke) Leroy	0.48	0.64
Lobelia giberroa Hemsl.	0.00	0.00
Macaranga capensis (Baill.) Sim	3.46	4.60
Maesa lanceolata Forssk.	0.06	0.07

Maytenus addat (Loes.) Sebsebe	0.14	0.18
Maytenus arbutifolia (A. Rich.) Wilczek	0.06	0.08
Maytenus undata (Thunb.) Blakelock	0.00	0.00
Microglossa pyrifolia (Lam.) Kuntze	0.00	0.00
Millettia ferruginea (Hochst.) Bak.	0.29	0.38
Mimusops kummel A. DC.	0.04	0.06
Nuxia congesta R. Br. ex Fresen.	0.15	0.20
Ochna holstii Engl.	0.01	0.01
Ocotea kenyensis (Chiov.) Robyns & Wilczek	7.50	9.98
Olea capensis L. subsp. *macrocarpa* (C.A. Wright.) Verdc.	4.98	6.63
Olea europaea L. subsp. *cuspidata* (Wall. ex G.Don) Cif.	0.00	0.00
Olea welwitschii (Knobl.) Gilg & Schellenb.	0.03	0.04
Olinia rochetiana A. Juss.	0.04	0.05
Oncoba spinosa Forssk.	0.00	0.01
Pavetta abyssinica Fresen.	0.00	0.00
Pavetta oliveriana Hiern	0.02	0.03

Phoenix reclinata Jacq.	0.04	0.05
Phyllanthus ovalifolius Forssk.	0.00	0.00
Phytolacca dodecandra L 'Herit.	0.02	0.02
Pittosporum viridiflorum Sims	0.01	0.01
Podocarpus falcatus (Thunb.) R. B. ex Mirb.	1.69	2.24
Polyscias fulva (Hiern) Harms	2.00	2.65
Pouteria adolfi-friederici (Engl.) Baehni	14.70	19.55
Premna schimperi Engl.	0.01	0.01
Protea gaguedi J. F. Gmel.	0.01	0.01
Prunus africana (Hook. f.) Kalkm.	2.25	2.99
Psychotria orophila Petit	0.34	0.45
Psydrax schimperiana (A. Rich.) Bridson	1.00	1.32
Pterolobium stellatum (Forssk.) Brenan	0.00	0.00
Rhamnus prinoides L'Herit.	0.00	0.00
Ricinus communis L.	0.00	0.00
Ritchiea albersii Gilg	0.00	0.00

Rosa abyssinica Lindley	0.00	0.00
Rytigynia neglecta (Hiern) Robyns	0.00	0.00
Schefflera abyssinica (Hochst. ex A. Rich.) Harms	0.04	0.06
Schrebera alata (Hochst.) Welw.	0.00	0.00
Scolopia theifolia Gilg	0.52	0.69
Scutia myrtina (Burm.f.) Kurz	0.00	0.00
Senna didymobotrya (Fresen.) Irwin & Barneby	0.00	0.00
Syzygium guineense (Willd.) DC subsp. *afromontanum* F. White	23.73	31.55
Teclea nobilis Del.	0.78	1.04
Terminalia schimperiana Hochst.	0.09	0.12
Toddalia asiatica (L.) Lam	0.00	0.00
Trichilia emetica Vahl	1.03	1.37
Vepris dainellii (Pic.Serm.) Kokwaro	0.39	0.51
Vernonia amygdalina Del.	0.00	0.00
Vernonia rueppellii Sch. Bip. ex Walp.	0.00	0.00

Zanthoxylum usambarense (Engl.) Kokwaro	0.00	0.00
Total	75.23	100.00

Appendix 6 Frequency distribution of trees and shrubs in Bore-Anferara-Wadera forest

Species name	**Percent frequency (%)**	**Relative frequency (RF)**
Acacia abyssinica Hochst. ex Benth.	0.89	0.04
Acacia bussei Harms ex Sjostedt	2.68	0.12
Acanthus eminens C.B. Clarke	67.86	3.10
Acokanthera schimperi (A. DC.) Schweinf.	14.29	0.65
Agarista salicifolia (Comm. ex Lam.) Hook. f.	0.89	0.04
Albizia gummifera (J.F. Gmel.) C.A. Sm.	16.96	0.78
Allophylus abyssinicus (Hochst.) Radlk.	51.79	2.37
Apodytes dimidiata E. Mey. ex Arn.	44.64	2.04
Arundinaria alpina K. Schum.	0.89	0.04
Bersama abyssinica Fresen.	77.68	3.55

Bridelia micrantha (Hochst.) Baill.	10.71	0.49
Brucea antidysenterica J. F. Mill.	1.79	0.08
Calpurnia aurea (Ait.) Benth.	18.75	0.86
Carissa spinarum L.	11.61	0.53
Celtis africana Burm. f.	34.82	1.59
Celtis toka (Forssk.) Hepper & Wood	8.04	0.37
Chionanthus mildbraedii (Gilg & Schellenb.) Stearn	39.29	1.80
Clutia abyssinica Jaub. & Spach.	3.57	0.16
Coffea arabica L.	7.14	0.33
Combretum molle R. Br. ex G. Don	0.89	0.04
Cordia africana Lam.	4.46	0.20
Croton macrostachyus Del.	46.43	2.12
Dodonea angustifolia L. f.	0.89	0.04
Dombeya torrida (J.F. Gmel.) P. Bamps	3.57	0.16
Dracaena afromontana Mildbr.	76.79	3.51
Ehretia cymosa Thonn.	28.57	1.31

Elaeodendron buchananii (Loes.) Loes.	86.61	3.96
Erythrococca trichogyne (Muell Arg.) Prain	25.00	1.14
Euclea racemosa Murr. subsp. *schimperi* (A. DC.) White	0.89	0.04
Fagaropsis angolensis (Engl.) Dale	48.21	2.20
Ficus sur Forssk.	34.82	1.59
Ficus thonningii Blume	4.46	0.20
Flacourtia indica (Burm. f.) Merr.	5.36	0.24
Galiniera saxifraga (Hochst.) Bridson	32.14	1.47
Grewia ferruginea Hochst. ex A. Rich.	2.68	0.12
Hagenia abyssinica (Brace) J.F. Gmel.	1.79	0.08
Halleria lucida L.	0.89	0.04
Hypericum revolutum Vahl	1.79	0.08
Ilex mitis (L.) Radlk.	25.00	1.14
Lepidotrichilia volkensii (Gürke) Leroy	67.86	3.10
Lobelia giberroa Hemsl.	1.79	0.08
Macaranga capensis (Baill.) Sim	53.57	2.45

Maesa lanceolata Forssk.	13.39	0.61
Maytenus addat (Loes.) Sebsebe	14.29	0.65
Maytenus arbutifolia (A. Rich.) Wilczek	67.86	3.10
Maytenus undata (Thunb.) Blakelock	14.29	0.65
Microglossa pyrifolia (Lam.) Kuntze	0.89	0.04
Millettia ferruginea (Hochst.) Bak.	15.18	0.69
Mimusops kummel A. DC.	16.96	0.78
Nuxia congesta R. Br. ex Fresen.	32.14	1.47
Ochna holstii Engl.	25.00	1.14
Ocotea kenyensis (Chiov.) Robyns & Wilczek	73.21	3.35
Olea capensis L. subsp. *macrocarpa* (C.A. Wright.) Verdc.	77.68	3.55
Olea europaea L. subsp. *cuspidata* (Wall. ex G.Don) Cif.	0.89	0.04
Olea welwitschii (Knobl.) Gilg & Schellenb.	7.14	0.33
Olinia rochetiana A. Juss.	9.82	0.45
Oncoba spinosa Forssk.	19.64	0.90
Pavetta abyssinica Fresen.	3.57	0.16

Pavetta oliveriana Hiern	4.46	0.20
Phoenix reclinata Jacq.	7.14	0.33
Phyllanthus ovalifolius Forssk.	0.89	0.04
Phytolacca dodecandra L 'Herit.	0.89	0.04
Pittosporum viridiflorum Sims	6.25	0.29
Podocarpus falcatus (Thunb.) R. B. ex Mirb.	63.39	2.90
Polyscias fulva (Hiern) Harms	37.50	1.71
Pouteria adolfi-friederici (Engl.) Baehni	79.46	3.63
Premna schimperi Engl.	8.93	0.41
Protea gaguedi J. F. Gmel.	0.89	0.04
Prunus africana (Hook. f.) Kalkm.	46.43	2.12
Psychotria orophila Petit	97.32	4.45
Psydrax schimperiana (A. Rich.) Bridson	90.18	4.12
Pterolobium stellatum (Forssk.) Brenan	7.14	0.33
Rhamnus prinoides L'Herit.	18.75	0.86
Ricinus communis L.	0.89	0.04

Ritchiea albersii Gilg	0.89	0.04
Rosa abyssinica Lindley	14.29	0.65
Rytigynia neglecta (Hiern) Robyns	35.71	1.63
Schefflera abyssinica (Hochst. ex A. Rich.) Harms	5.36	0.24
Schrebera alata (Hochst.) Welw.	0.89	0.04
Scolopia theifolia Gilg	16.07	0.73
Scutia myrtina (Burm.f.) Kurz	1.79	0.08
Senna didymobotrya (Fresen.) Irwin & Barneby	2.68	0.12
Syzygium guineense (Willd.) DC subsp. *afromontanum* F. White	80.36	3.67
Teclea nobilis Del.	99.11	4.53
Terminalia schimperiana Hochst.	0.89	0.04
Toddalia asiatica (L.) Lam	3.57	0.16
Trichilia emetica Vahl	26.79	1.22
Vepris dainellii (Pic.Serm.) Kokwaro	55.36	2.53
Vernonia amygdalina Del.	1.79	0.08

Vernonia rueppellii Sch. Bip. ex Walp.	2.68	0.12
Zanthoxylum usambarense (Engl.) Kokwaro	8.93	0.41
		100.00

Appendix 7 Importance Value Index (IVI) of trees and shrubs in Bore-Anferara-Wadera forest

Species name	**Relative Density (RD)**	**Relative Frequency (RF)**	**Relative Dominance (RDO)**	**IVI**	**%**
Acacia abyssinica Hochst. ex Benth.	0.01	0.04	0.01	0.06	0.02
Acacia bussei Harms ex Sjostedt	0.08	0.12	0.04	0.24	0.08
Acanthus eminens C.B. Clarke	1.44	3.10	0.05	4.59	1.53
Acokanthera schimperi (A. DC.) Schweinf.	0.37	0.65	0.09	1.11	0.37
Agarista salicifolia (Comm. ex Lam.) Hook. f.	0.01	0.04	0.00	0.05	0.02
Albizia gummifera (J.F. Gmel.) C.A. Sm.	0.11	0.78	0.10	0.99	0.33
Allophylus abyssinicus (Hochst.) Radlk.	0.78	2.37	1.09	4.23	1.41
Apodytes dimidiata E. Mey. ex Arn.	0.73	2.04	0.71	3.48	1.16
Arundinaria alpina K. Schum.	0.01	0.04	0.00	0.05	0.02

Bersama abyssinica Fresen.	0.93	3.55	0.28	4.75	1.58
Bridelia micrantha (Hochst.) Baill.	0.20	0.49	0.12	0.81	0.27
Brucea antidysenterica J. F. Mill.	0.01	0.08	0.00	0.09	0.03
Calpurnia aurea (Ait.) Benth.	0.31	0.86	0.02	1.19	0.40
Carissa spinarum L.	0.20	0.53	0.01	0.73	0.24
Celtis africana Burm. f.	0.89	1.59	0.34	2.82	0.94
Celtis toka (Forssk.) Hepper & Wood	0.11	0.37	0.07	0.55	0.18
Chionanthus mildbraedii (Gilg & Schellenb.) Stearn	1.73	1.80	0.14	3.67	1.22
Clutia abyssinica Jaub. & Spach.	0.02	0.16	0.00	0.18	0.06
Coffea arabica L.	0.06	0.33	0.00	0.38	0.13
Combretum molle R. Br. ex G. Don	0.01	0.04	0.02	0.07	0.02
Cordia africana Lam.	0.12	0.20	0.07	0.39	0.13
Croton macrostachyus Del.	1.85	2.12	2.24	6.21	2.07
Dodonea angustifolia L. f.	0.01	0.04	0.00	0.05	0.02
Dombeya torrida (J.F. Gmel.) P. Bamps	0.01	0.16	0.00	0.17	0.06
Dracaena afromontana Mildbr.	13.70	3.51	3.44	20.66	6.89

Ehretia cymosa Thonn.	1.22	1.31	0.37	2.89	0.96
Elaeodendron buchananii (Loes.) Loes.	2.46	3.96	0.40	6.82	2.27
Erythrococca trichogyne (Muell Arg.) Prain	0.63	1.14	0.04	1.81	0.60
Euclea racemosa Murr. subsp. *schimperi* (A. DC.) White	0.01	0.04	0.00	0.05	0.02
Fagaropsis angolensis (Engl.) Dale	0.75	2.20	0.17	3.12	1.04
Ficus sur Forssk.	0.95	1.59	2.29	4.83	1.61
Ficus thonningii Blume	0.05	0.20	0.03	0.29	0.10
Flacourtia indica (Burm. f.) Merr.	0.01	0.24	0.00	0.26	0.09
Galiniera saxifraga (Hochst.) Bridson	0.86	1.47	0.13	2.46	0.82
Grewia ferruginea Hochst. ex A. Rich.	0.06	0.12	0.01	0.19	0.06
Hagenia abyssinica (Brace) J.F. Gmel.	0.04	0.08	0.04	0.16	0.05
Halleria lucida L.	0.01	0.04	0.00	0.05	0.02
Hypericum revolutum Vahl	0.01	0.08	0.00	0.09	0.03
Ilex mitis (L.) Radlk.	0.83	1.14	0.05	2.03	0.68
Lepidotrichilia volkensii (Gürke) Leroy	3.90	3.10	0.64	7.64	2.55
Lobelia giberroa Hemsl.	0.02	0.08	0.00	0.10	0.03

Macaranga capensis (Baill.) Sim	4.20	2.45	4.60	11.26	3.75
Maesa lanceolata Forssk.	0.55	0.61	0.07	1.24	0.41
Maytenus addat (Loes.) Sebsebe	0.18	0.65	0.18	1.02	0.34
Maytenus arbutifolia (A. Rich.) Wilczek	0.35	3.10	0.08	3.53	1.18
Maytenus undata (Thunb.) Blakelock	0.08	0.65	0.00	0.73	0.24
Microglossa pyrifolia (Lam.) Kuntze	0.01	0.04	0.00	0.05	0.02
Millettia ferruginea (Hochst.) Bak.	0.97	0.69	0.38	2.04	0.68
Mimusops kummel A. DC.	0.33	0.78	0.06	1.16	0.39
Nuxia congesta R. Br. ex Fresen.	0.81	1.47	0.20	2.48	0.83
Ochna holstii Engl.	0.09	1.14	0.01	1.25	0.42
Ocotea kenyensis (Chiov.) Robyns & Wilczek	6.83	3.35	9.98	20.15	6.72
Olea capensis L. subsp. *macrocarpa* (C.A. Wright.) Verdc.	2.93	3.55	6.63	13.10	4.37
Olea europaea L. subsp. *cuspidata* (Wall. ex G.Don) Cif.	0.01	0.04	0.00	0.05	0.02
Olea welwitschii (Knobl.) Gilg & Schellenb.	0.11	0.33	0.04	0.48	0.16
Olinia rochetiana A. Juss.	0.07	0.45	0.05	0.57	0.19
Oncoba spinosa Forssk.	0.11	0.90	0.01	1.02	0.34

Pavetta abyssinica Fresen.	0.04	0.16	0.00	0.20	0.07
Pavetta oliveriana Hiern	0.10	0.20	0.03	0.33	0.11
Phoenix reclinata Jacq.	0.12	0.33	0.05	0.50	0.17
Phyllanthus ovalifolius Forssk.	0.01	0.04	0.00	0.05	0.02
Phytolacca dodecandra L 'Herit.	0.01	0.04	0.02	0.07	0.02
Pittosporum viridiflorum Sims	0.10	0.29	0.01	0.40	0.13
Podocarpus falcatus (Thunb.) R. B. ex Mirb.	1.67	2.90	2.24	6.80	2.27
Polyscias fulva (Hiern) Harms	0.91	1.71	2.65	5.28	1.76
Pouteria adolfi-friederici (Engl.) Baehni	5.43	3.63	19.55	28.60	9.53
Premna schimperi Engl.	0.20	0.41	0.01	0.62	0.21
Protea gaguedi J. F. Gmel.	0.01	0.04	0.01	0.06	0.02
Prunus africana (Hook. f.) Kalkm.	0.98	2.12	2.99	6.08	2.03
Psychotria orophila Petit	7.97	4.45	0.45	12.87	4.29
Psydrax schimperiana (A. Rich.) Bridson	3.63	4.12	1.32	9.07	3.02
Pterolobium stellatum (Forssk.) Brenan	0.09	0.33	0.00	0.42	0.14
Rhamnus prinoides L'Herit.	0.02	0.86	0.00	0.88	0.29

Ricinus communis L.	0.01	0.04	0.00	0.05	0.02
Ritchiea albersii Gilg	0.01	0.04	0.00	0.05	0.02
Rosa abyssinica Lindley	0.07	0.65	0.00	0.72	0.24
Rytigynia neglecta (Hiern) Robyns	0.21	1.63	0.00	1.84	0.61
Schefflera abyssinica (Hochst. ex A. Rich.) Harms	0.07	0.24	0.06	0.37	0.12
Schrebera alata (Hochst.) Welw.	0.01	0.04	0.00	0.05	0.02
Scolopia theifolia Gilg	4.10	0.73	0.69	5.52	1.84
Scutia myrtina (Burm.f.) Kurz	0.01	0.08	0.00	0.09	0.03
Senna didymobotrya (Fresen.) Irwin & Barneby	0.01	0.12	0.00	0.13	0.04
Syzygium guineense (Willd.) DC subsp. *afromontanum* F. White	6.05	3.67	31.55	41.27	13.76
Teclea nobilis Del.	11.24	4.53	1.04	16.81	5.60
Terminalia schimperiana Hochst.	0.12	0.04	0.12	0.29	0.10
Toddalia asiatica (L.) Lam.	0.04	0.16	0.00	0.20	0.07
Trichilia emetica Vahl	0.32	1.22	1.37	2.91	0.97
Vepris dainellii (Pic.Serm.) Kokwaro	3.22	2.53	0.51	6.26	2.09

Vernonia amygdalina Del.	0.01	0.08	0.00	0.09	0.03
Vernonia rueppellii Sch. Bip. ex Walp.	0.07	0.12	0.00	0.19	0.06
Zanthoxylum usambarense (Engl.) Kokwaro	0.02	0.41	0.00	0.43	0.14
	100.00	100.00	100.00	300.00	100.00

Appendix 8 Density of seedlings and saplings of tree species in Bore-Anferara-Wadera forest

Species name	**Seedlings**	**Saplings**
Acacia abyssinica Hochst. ex Benth.	0	0
Acacia bussei Harms ex Sjostedt	0	0
Acokanthera schimperi (A. DC.) Schweinf.	73	12
Agarista salicifolia (Comm. ex Lam.) Hook. f.	0	0
Albizia gummifera (J.F. Gmel.) C.A. Sm.	14	6
Allophylus abyssinicus (Hochst.) Radlk.	193	44
Apodytes dimidiata E. Mey. ex Arn.	28	13
Bersama abyssinica Fresen.	155	90

Bridelia micrantha (Hochst.) Baill.	2	1
Celtis africana Burm. f.	26	15
Celtis toka (Forssk.) Hepper & Wood	0	0
Combretum molle R. Br. ex G. Don	0	0
Cordia africana Lam.	1	0
Croton macrostachyus Del.	10	12
Dracaena afromontana Mildbr.	143	114
Ehretia cymosa Thonn.	13	9
Elaeodendron buchananii (Loes.) Loes.	126	97
Fagaropsis angolensis (Engl.) Dale	45	27
Ficus sur Forssk.	5	2
Hagenia abyssinica (Brace) J.F. Gmel.	0	0
Ilex mitis (L.) Radlk.	12	6
Lepidotrichilia volkensii (Gürke) Leroy	81	59
Macaranga capensis (Baill.) Sim	60	49
Maesa lanceolata Forssk.	8	5

Millettia ferruginea (Hochst.) Bak.	21	12
Mimusops kummel A. DC.	9	2
Nuxia congesta R. Br. ex Fresen.	10	11
Ocotea kenyensis (Chiov.) Robyns & Wilczek	292	166
Olea capensis L. subsp. *macrocarpa* (C.A. Wright.) Verdc.	101	87
Olea welwitschii (Knobl.) Gilg & Schellenb.	1	2
Olinia rochetiana A. Juss.	6	1
Phoenix reclinata Jacq.	6	0
Pittosporum viridiflorum Sims	1	1
Podocarpus falcatus (Thunb.) R. B. ex Mirb.	178	140
Polyscias fulva (Hiern) Harms	2	1
Pouteria adolfi-friederici (Engl.) Baehni	324	87
Prunus africana (Hook. f.) Kalkm.	143	77
Psydrax schimperiana (A. Rich.) Bridson	126	131
Schefflera abyssinica (Hochst. ex A. Rich.) Harms	0	0
Scolopia theifolia Gilg	562	128

Syzygium guineense (Willd.) DC subsp. *afromontanum* F. White	416	49
Terminalia schimperiana Hochst.	0	0
Trichilia emetica Vahl	3	3
Zanthoxylum usambarense (Engl.) Kokwaro	7	1
Total (Individuals/ha)	3203	1461

Appendix 9 Classification accuracy assessment of Bore-Anferara-Wadera forest

A) Classification accuracy assessment of 1986

Class Name	**Reference Totals**	**Classified Totals**	**Number Correct**	**Producers Accuracy (%)**	**Users Accuracy (%)**
Agriculture	9	9	7	100.00	77.78
Built up area	8	8	7	100.00	87.50
Forest land	8	9	8	112.50	100.00
Grazing land	6	6	5	100.00	83.33
Shrub land	7	6	6	85.71	85.71
Wetland	6	6	5	100.00	83.33
Total	44	44	39	100.00	86.28

Overall Classification Accuracy = 86.28%, Overall Kappa Statistics = 0.84

A) Classification accuracy assessment of 2000

Class Name	**Reference Totals**	**Classified Totals**	**Number Correct**	**Producers Accuracy (%)**	**Users Accuracy (%)**
Agriculture	9	9	8	100.00	80.00
Built up area	7	8	7	100.00	100.00
Forest land	8	8	7	100.00	100.00
Grazing land	7	6	5	125.00	100.00
Shrub land	7	7	6	100.00	80.00
Wetland	7	7	7	71.43	71.43
Total	45	45	39	100.00	88.92

Overall Classification Accuracy = 88.92%, Overall Kappa Statistics = 0.86

A) Classification accuracy assessment of 2014

Class Name	**Reference Totals**	**Classified Totals**	**Number Correct**	**Producers Accuracy (%)**	**Users Accuracy (%)**
Agriculture	8	8	7	100	87.5
Built up area	9	9	7	100	77.78
Forest land	9	9	8	100	88.89
Grazing land	7	7	6	100	85.71
Shrub land	7	7	6	100	85.71
Wetland	7	7	7	100	100
Total	47	47	41	100	89.45

Overall Classification Accuracy = 89.45%, Overall Kappa Statistics = 0.87