ADDIS ABABA UNIVERSITY, GRADUATE PROGRAMMES

COLLEGE OF NATURAL AND COMPUTATIONAL SCIENCES

DEPARTMENT OF PLANT BIOLOGY AND BIODIVERSITY MANAGEMENT

Forest Structure, Carbon Stocks and Leaf Litter Decomposition of two Selected Afromontane Forests in the Western Escarpment of Central Rift Valley and the Gibe Watershed, Ethiopia

BY

TALEMOS SETA SHANKA

Addis Ababa, Ethiopia
June 2017

Forest Structure, Carbon Stocks and Leaf Litter Decomposition of two Selected Afromontane Forests in the Western Escarpment of Central Rift Valley and the Gibe Watershed, Ethiopia

Talemos Seta Shanka

A Thesis Presented to the Graduate Programmes of the Addis Ababa University in Partial Fulfilment of the Requirements for the Degree of Doctor of Philosophy in Plant Biology and Biodiversity Management

Addis Ababa University

Addis Ababa, Ethiopia

June 2017

ADDIS ABABA UNIVERSITY
GRADUATE PROGRAMMES

This is to certify that the thesis prepared by Talemos Seta Shanka, entitled: *Forest Structure, Carbon Stocks and Leaf Litter Decomposition of two Selected Afromontane Forests in the Western Escarpment of Central Rift Valley and the Gibe Watershed, Ethiopia* submitted in Partial fulfilment of the requirements for the Degree of Doctor of Philosophy in Plant Biology and Biodiversity Management complies with the regulations of the University and meets the accepted standards with respect to originality and quality.

Signed by the Examining Committee:

Advisor: Sebsebe Demissew (Prof.)___Signature________________Date____________

Advisor: Zerihun Woldu (Prof.)_____Signature________________Date____________

Examiner: Mulugeta Lemenih (PhD) Signature________________Date____________

Examiner: Tamrat Bekele (PhD) Signature________________Date____________

Chair of the Department or Graduate Program Coordinator

Forest Structure, Carbon Stocks and Leaf Litter Decomposition of two Selected Afromontane Forests in the Western Escarpment of Central Rift Valley and the Gibe Watershed, Ethiopia

Talemos Seta Shanka

Addis Ababa University, 2017

Addis Ababa, Ethiopia

Dedicated

To

My Lovely Wife, Saba Kaleb

ABSTRACT

The Afromontane forests of Ethiopia have been under a serious degradation threat. Understanding forest structure, floristic compositions and their important roles in providing ecosystem functions like climate change mitigation through carbon sequestration and nutrient flow dynamics are important in targeting sustainable forest management strategies. Therefore, the major objectives of this study were to determine; i/ forest structure, plant community composition in relation to environmental factors and carbon stocks of Biteyu forest and ii/ quantifying forest structure, carbon stocks, litterfall and leaf litter decomposition dynamics, and nutrient release patterns from the Boter-Becho forest. Systematic sampling technique was employed for vegetation and environmental data collection. Thirty plots of 900 m^2 and 71 plots of the same quadrat size were selected from the Biteyu and Boter-Becho forests, respectively. Total carbon stocks (t ha^{-1}) were estimated from aboveground biomass, belowground bimass, soil and forest litter carbon pools. The appropriate allometric models were applied for aboveground and belowground biomass estimations. To determine litterfall dynamics in Boter-Becho forest, two sites of the forest namely, low disturbed (LD) and high disturbed (HD) were selected subjectively using field observations. A total of 20 litter traps, ten for each site were deployed. Two hundred forty collections were made from 4th March 2014 to 3rd February 2015, and monthly collections were oven dried at 80 ^{0}C for 24 hrs to constant weight. Similarly, the decomposition rate of leaf litter in Boter-Becho forest was investigated using the first order negative exponential decay equation. Different data analyses techniques were employed using R statistical software. The findings showed 190 plant species distributed among 154 genera and 73 families in Biteyu forest. Moreover, altitude and slope strongly affect the community structure of Biteyu forest. The estimates of total carbon stock in Biteyu and Boter-Becho forests was 166 ± 16.4 and 393 ± 24 t ha^{-} ,^{1} respectively. On the other hand, the amount of total litterfall estimated in Boter-Becho forest was 8.7 t ha^{-1} yr^{-1}. At the end of one year, 66.02% and 66.72% of the leaf litter was decomposed in LD and HD sites, respectively. Consequently, the annual decomposition rate constant measured for Boter-Becho forest was 1.405 $year^{-1}$(the average of the two sites). With regard to carbon and nutrient release in the Boter-Becho forest, there was no *significant difference ($P > 0.05$) in their mean concentration remaining except K between two sites. Similarly, no strong relationships were observed between initial leaf litter chemistry and C, N, P concentration ($P > 0.05$) released during the decomposition in both sites. There was high anthropogenic effect and high dependence of the local community in Biteyu forest so that forest conservation and restoration measures should be sought. Moreover, the variation in rainfall and temperature are responsible for controlling litterfall production, rate of litter decomposition, C and nutrient release pattern in the Boter-Becho forest. Therefore, the interventions, which reduce the climate change effect, would be very important in the maintenance of forest ecosystem functioning.*

Key-words/phrases*: Biteyu-forest, Boter-Becho forest, Carbon stocks, forest disturbance, leaf litter decomposition, Litterfall, Nutrient Release*

ACKNOWLEDGEMENTS

This longest journey with many ups and downs comes to an end with the help of Almighty God. My deepest gratitude goes to my Supervisors Professor Sebsebe Demissew and Professor Zerihun Woldu for their unreserved and tremendous advice from the beginning to the successful completion of my PhD dissertation. I also extend my gratitude to Dr. Ermias Aynekulu and Prof. Yadvinder Malhi for they have critically read my PhD dissertation and provided many suggestions for its improvement.

Dilla University is highly acknowledged for sponsoring me in the whole study period. I am grateful to the Department of PBBM, AAU, for the financial support from Thematic Research Project entitled" Restoration of Upper Awash and Gibe Watersheds and the Western Escarpments of the Central Rift Valley: Mechanisms for Re-establishing Biodiversity, Ecosystem functions and Livelihoods". Moreover, I am grateful to AAU for financial support of short-term training held in the University of Oxford, UK.

I have special thanks to the late Department Head, Professor Ensermu Kelbessa for his formal and informal advice, positive and unreserved support in activities important for my research work. I thank him and would like to say "**Rest in Peace**". Technical Herbarium staffs namely Woge, Melaku and Shewa were highly indebted for their assistance in plant specimen identification. My heartfelt appreciation goes to the secretary of the department for on time information exchange during my research activities.

I am deeply indebted to my beloved wife, Saba Kaleb, and my beloved son Robel Talemos for their patience and strength throughout my study period.

Oromia Forest and Wildlife Enterprise, Miskan Woreda Agricultural office, Holeta Research centre, Hawassa University college of Agriculture, local communities surrounding both forests, Boter-Becho District Officers were highly acknowledged for their positive and unreserved support. Many people have been instrumental in the completion of my PhD Dissertation and I am grateful to all of them even if they are not mentioned by name here.

Table of Contents

Contents **Pages**

List of Figures

List of Tables

List of Appendices

List of Acronyms

AAU = Addis Ababa University

AGB = Aboveground Biomass

AGBC = Carbon in Aboveground Biomass

BERSMP = Bale Eco-Region Sustainable Management Program

BGB = Belowground Biomass

BGBC= Carbon in Belowground Biomass

COP = Conference of the Parties

DAF = Dry Evergreen Afromontane Forest

DBH = Diameter at Breast Height

DPBBM = Department of Plant Biology and Biodiversity Management

FAO = Food and Agriculture Organizations of the United Nations

FRL = Forest Reference Level

GHG = Green House Gas

HD = High Disturbed sites

IPCC = Intergovernmental Panel on Climate Change

LD= Low Disturbed sites

MAF = Moist Evergreen Afromontane forest

$REDD^+$ = Reducing Emissions from Deforestation and Forest Degradation, as well as conservation, sustainable management of forests and enhancement of forest carbon stocks.

SUNARMA = Sustainable Natural Resource Management

SFM = Sustainable forest management

SNNPR= Southern Nations Nationalities and peoples' Region

SOC = Soil organic carbon

SOM = Soil organic matter

UNFCCC = United Nations Framework Convention on Climate Change

WBISPP = Wood Biomass Innventory and Strategic Planning Project

CHAPTER ONE

1. Introduction

1.1. Background of the study

There is no generally accepted measure to quantify or express forest structure. Foresters measure a variety of structural attributes such as aboveground biomass, abundance, basal area, canopy height and plant density. However, each of the attributes contributes to the overall forest structure, but do not individually describe it completely (Delang and Li, 2013). Forest structure is also expressed as all the morphological characters of the vegetation except the qualitative and quantitative properties of the plant taxa included in the vegetation (Barkman, 1979). According to this author, the attributes in the structural analysis include growth form, height class distribution, leaf type and leaf size class distribution. Moreover, Leemans (1989) in his study of stand structure in Swedish boreal forests analyzed density, basal area, maximum tree height, and minimum and maximum bole height.

In general, forest structure is both a product and driver of ecosystem processes and biological diversity. Recently, it has become apparent that changes in forest structure because of anthropogenic as well as natural factors lead to undesirable consequences for other components of forest ecosystem (Spies, 1998). Accordingly, living trees and soils in natural forests constitute major long-term stocks of organic carbon. However, variations in the forest structure affect carbon dynamics, which are necessary for predicting potential losses and carbon storage. Variation in structure caused by forest degradation disrupts the carbon cycle through direct effects on tree biomass and on soil organic matter (less litter production and higher rates of litter and organic matter decomposition) (Dirham, 1998) and it leads to lower species richness and to decreased variability in tree diameter.

An increase in carbon dioxide concentration in the atmosphere mainly leads to rising global surface temperature (Schimel, 1995). For instance, emission of 4 gigaton (1 gigaton = 1 billion metric ton) of carbon through anthropogenic activities increases the atmospheric concentration of CO_2 by approximately 1 parts per million (Lal, 2006, 2008, 2010). The rise of CO_2 and other GHGs is believed to be the most prominent cause of global warming and climate change as observed by hotter air temperature, lower amount of rainfall, longer period of drought (IPCCC, 2007). In connection with the global warming, there are increasing concerns about alpine glaciers retreating, sea levels rising and ecological and climate zones shifting. The consequences of these changes could be highly disruptive to present and future human societies as well as ecosystems that we depend on (Nair *et al*., 2009; UNFCCC, 2007).

Forests are known to be an important natural brake on climate change since they play considerable roles globally as both a carbon sink and source because of their large biomass per unit area of land (Gibbs *et al*., 2007). The carbon stocks in the forests originated from the atmosphere and are accumulated in the organic matter of soils and trees. According to Lal (2010), about 500 billion tons of carbon is stored in vegetation worldwide. The carbon continuously cycles between forests and the atmosphere through the decomposition of dead organic matter (Alexandrove, 2007) and therefore, changing carbon stocks in forests can affect the amount of carbon in the atmosphere.

However, deforestation and forest degradation alone account for 17.4% of the world's GHG emissions and the problem is acute mainly in tropical and subtropical forests where conversion of forest lands to arable and pasturelands have been increased (IPCC, 2007). Moreover, clearing tropical forests also destroys globally important carbon sinks that are currently sequestering CO_2 from the atmosphere and are critical to future climate stabilization (Stephens *et al.,* 2007). For instance, deforestation accounts for nearly 70% of total emissions in Africa (FAO, 2005). In addition, agricultural practices lead to a reduction in ecosystem carbon stocks mainly due to removal of AGB as harvest with subsequent burning and/or decomposition, loss of soil carbon as CO_2, and loss of soil C by erosion (Mutuo *et al*., 2005; Ramachandran *et al*., 2007).

Tropical forests have the largest potential to mitigate climate change amongst the world's forests through conservation of existing carbon pools (e.g. reduced impact logging), expansion of carbon sinks (e.g. reforestation, agro-forestry), and substitution of wood products for fossil fuels (Brown *et al*., 1996; Gorte, 2009). Tropical Montane forests, typically store 100-200 t C ha^{-1} only from the aboveground biomass. According to a 40-year study of African, Asian, and South American tropical forests by the University of Leeds, showed that tropical forests absorb about 18% of all carbon dioxide added by fossil fuels (NIACS, 2012). Thus, carbon storage and sequestration by terrestrial ecosystems, i.e. soil and vegetation is one of the options for reducing carbon emissions as a strategy for climate change mitigation.

Carbon sequestration refers to the natural process of removing excess carbon dioxide from the atmosphere and storing it in long-lived pools of carbon by fixing it or locking it up from being released back to the atmosphere (IPCC, 2007; Nair *et al*., 2009; Lal, 2010).

Moreover, the United Nations Framework Convention on Climate Change (UNFCCC) defines carbon sequestration as the process of removing carbon from the atmosphere and depositing it in a reservoir. It involves the transfer of atmospheric carbon dioxide and secure storage in long-lived pools (UNFCCC, 2007). Sequestration of atmospheric CO_2 in long-lived pools could reduce CO_2 concentration in the atmosphere and thereby the GHG-induced global warming (Lal, 2001; Nair *et al*., 2009). The process of carbon sequestration is assessed through either the amount of carbon stored or estimating the annual carbon sequestration rate (Iverson *et al*., 1993). The studies on carbon stock and sequestration have been focusing on and expressing the sequestration in terms of biomass and carbon stock. The design and evaluation of global scale carbon models require field estimates of forest biomass such as aboveground, belowground, litterfall and dead wood, which are involved in the regulation of atmospheric carbon concentration (Fearnside, 1997; Baishya *et al*., 2009).

Moreover, the process of decomposition of forest litter is very important for forest ecosystem functioning. The process includes leaching, break up by soil fauna, transformation of organic matter by microorganisms and transfer of organic and mineral compounds to the soil (Yong *et al*., 2007). Much of the energy originating from primary production is released during decomposition (Phillipson *et al.,* 1975). During this release, plant nutrients become available for recycling within the ecosystem. Thus, decomposition and nutrient release processes are particularly important in tropical ecosystems, where soils can be naturally low in fertility and nutrient status (Okeke and Omaliko, 1992).

Decomposition is mostly biological but influenced by abiotic factors through their effects on soil fauna. Climate, soil characteristics, quality of decomposing organic matter and soil organisms are the most important factors regulating litter decomposition (Swift *et al*., 1979). With respect to the relationship of climatic condition and decomposition of litter, climatic factors including temperature and rainfall are the strongest determinant factors on the mass loss of litter (Liu *et al*., 2005). Consequently, litter decomposition rates increase with increasing temperature and exponentially decrease with higher altitude and lowering temperature (Vitousek *et al.,* 1994). Moreover, climatic seasonality characterized by alternating wet and dry periods also plays a vital role in regulating the rate of decomposition (Tripathi and Singh, 1992).

In the forest floor, decomposition acts as an input and output system of nutrients (Das and Ramakrishnan, 1985) and the rates at which forest litter falls and subsequently decays regulate energy flow, primary productivity and nutrient cycling in forest ecosystems (Olson, 1963; Waring and Schlesinger, 1985). Changes in amount of organic carbon in the soil are the result of differences between additions and losses. As an insulating layer, litter protects the soil from extreme changes in moisture and temperature, intercepts through-fall, and improves infiltration (Das and Ramakrishnan 1985, Bekele Lemma *et al*., 2007). It is also a principal source of energy for the saprobiota of the forest floor and soil. In the natural forest and plantations, litter-fall exhibits periodic characteristics for some species (Proctor, 1983). In sites with severe dry seasons, deposition of litter on the forest floor increases and decomposition rate decreases resulting in an accumulation of litter at the soil surface, but, decomposition and mass reduction starts at the onset of moist conditions during the wet season (Proctor, 1983, Swift and Anderson, 1989).

Mean annual decomposition rate constants (k) for temperate and tropical forests have been estimated at k = 0.9 and k = 1.8, respectively (Torreta and Takeda, 1999). Within the tropics, there is some evidence of regionality in decay rates coefficient with k > 2 (high) for most African forests and k = 1-2 (medium to high) for forests in Southeast Asia and Neotropics (Anderson and Swift, 1983). However, decay rates coefficient can be low (k < 1) even in tropical areas depending on litter type, season and altitude (Verhoef and Guandi, 2001). The higher decomposition rates facilitate more rapid nutrient cycling in ecosystems (Sariyildiz *et al*., 2005).

1.2. Problem Justification

The current forest cover of Ethiopia is about 12.3 million ha or 11.0 % (FAO, 2010; MEFCC, 2015). It seemed to have shown a certain increase compared to the figure (~4.0%) of forest cover in Ethiopia in the past due to mainly by the change in forest definiton (von Breitenbach, 1961, 1962). However, the natural forests in different parts of the country are still under degradation threat. Deforestation and forest degradation of the country is caused by the anthropogenic pressures resulting in the change of forest structure and forest loss leading to the total absence of primary forests. For instance, the changes in forest cover between 1990 and 2010 showed that Ethiopia has lost an average of about 140,900 ha (0.93%) per year. Due to the above and other related factors, the country has been experiencing the climate change effects as evidenced by the increase in average temperature, a change in rainfall patterns and heavy flooding, recurrent drought and food insecurity. These climate change impacts will affect everyone, particularly, the Sub-Saharan Africa, where countries are already vulnerable to climate variability and have the least capacity to respond (BERSMP, 2010). However, countries can respond and counteract to the climate change effect, partly, by designing climate change mitigation strategies. Ethiopia, as one of the countries in Sub-Saharan Africa, has a climate change mitigation potential i.e. greening the environment by planting trees and regenerating the natural forests disturbed by man. As an evidence, the rough estimate of Ethiopia's forests are shown to contain 219 million metric tons of carbon in living forest biomass (FAO, 2011), almost equivalent to about 67% of global annual carbon emission per year.

With regard to the Ethiopian forest vegetation, many studies have been conducted, particularly, in relation to classification, floristic composition and species diversity (White, 1983; Friis, 1992; Tamrat Bekele, 1993, 1994; Sebsebe Demissew *et al*., 1996; Zerihun Woldu, 1999; Friis and Sebsebe Demissew, 2001, Tadesse Woldemariam, 2003; Sebsebe Demissew *et al*., 2004; Teshome Soromessa *et al.,* 2004; Feyera Senbeta, 2006; Ensermu Kelbessa and Teshome Soromessa, 2008). These studies have also included various proximate and underlying causes of deforestation and forest degradation in the country to determine the pattern of forest structure and the plant diversity of natural forests. However, similar studies are lacking in Biteyu (dry evergreen afromontane forest) and Boter-Becho forest (moist evergreen afromontane forest) which are the subject of the present study.

Comparatively, the country is lacking periodic inventory data of carbon stocks for national carbon accounting and for the purpose of REDD+ initiatives. Due to this and other factors, developing sustainable forest management planning which attracts carbon markets has become a great challenge for the country.

Furthermore, the litterfall production, leaf litter decomposition, and the nutrient dynamics are very important for determining the functional processes of forest ecosystem and hence are indicators of ecosystem stability or homeostasis. The processes of litter decomposition, mainly, play an important role in terrestrial carbon and nutrient cycling (Rapport *et al*., 1985). Even if the above facts are important for determining the functioning of ecosystem processes and hence the forest health status, such studies are very limited in the natural forests of Ethiopia.

Not to disregard, there are a very few fragmented, small scale studies conducted in some parts of the country in relation to the forest carbon stock, litterfall production and litter decomposition in forest ecosystems. For instance, Humbo Community-Based Natural Regeneration project in SNNPR by World Vision Ethiopia and Australia are the first forest carbon project (CCB-AR-PDD, 2009). An afforestation/reforestation covering an area of 2,728 ha to restore indigenous forest species to the land was done. This project was registered under the CDM (clean development mechanisms) of the Kyoto Protocol in 2009 and the World Bank Bio Carbon Fund has purchased the emission reductions generated by the project (CCB-AR-PDD, 2009). Bale REDD+ project supported by Bale Eco-Region in Oromiya region by Farm Africa and SOS Sahel (BERSMP, 2010) are working in the project level. In addition, structure, biomass and net primary production in a dry tropical afromontane forest in Ethiopia (Getachew Tesfaye, 2007), Estimating and Mapping of Carbon Stocks based on Remote Sensing, GIS and Ground Survey in the Menagesha Suba State Forest, Ethiopia (Mesfin Sahle, 2011); Estimation of Carbon Stock in Church Forests; Implications for managing Church forests for carbon emission Reduction (Tulu Tolla, 2011) are some of the studies to mention. However, these studies are limited to make inferences about the national forest carbon accounting warranting further studies.

On the other hand, a very few studies have been available on the litterfall production and decomposition dynamics of some of plantation and natural forest species in Ethiopia. The litter production and decomposition dynamics of the detritus material for exotic species were studied by Lisanework Nigatu and Michelsen (1994) at Menagesha State Forest on the gentle slope of Mt. Wuchacha in the central highlands of Ethiopia. According to the above authors, the total annual fine litter production was found to be in the order of *Cupressus lusitanica* < *Eucalyptus globulus* < *Pinus patula* < *Juniperus procera*. Bekele Lemma *et al*. (2007) found that the soil organic carbon under *Cupressus lusitanica* was larger than under *Pinus patula* and *Eucalyptus grandis* due to higher total litter input at Belete Forest, located in the southwestern highlands of Ethiopia. Moreover, Ambachew Demissie (2012) in Gambo District, Southern Ethiopia, suggested that the decomposition rate in *Pinus patula* was relatively lower than the other species and the litter production under *Eucalyptus* was comparatively higher than that in coniferous species. With regard to the nutrient dynamics in the forest, Jiregna Gindaba *et al*. (2004) determined the nutrient composition and short-term release from *Croton macrostachyus* and *Milletia ferruginea* leaves in 12 weeks study period around Wondo Genet, Ethiopia.

The above-mentioned authors have tried to determine the variation in litterfall production, decomposition rate and nutrient release of some of the species in plantation and natural forests of the country to explain partly the nutrient dynamics in forest ecosystems. Since these studies only depended on species level, this warrants additional studies on the forest level (Boter-Becho forest). Therefore, it is imperative to determine forest structure, total carbon stocks in biomass, soils and forest litter layers, which are the known potential pools for organic carbon in both forests, and the litterfall production, decomposition, and nutrient release pattern of Boter-Becho forest. Thus, from this point of view, the following research questions were developed.

1.3. Research Questions

- What is the status of plant diversity in Biteyu Forest?
- What are the environmental variables that determine the pattern of species distribution and community formation in Biteyu forest and the carbon sequestration potentials of both forests?
- Do the terrain variables have an effect on the forest carbon pools?
- What is the amount of carbon stored in different carbon pools of both Biteyu and Boter-Becho forests?
- What is/are the rates of leaf litter decomposition, patterns of litterfall and nutrient release, and the effect of temporal/seasonal variation on litter decomposition in Boter-Becho Forest?

Scientific information regarding the above issues were lacking in both forests. Such information is critically important to design and implement appropriate and suitable conservation measures. Therefore, based on the above research questions, the following study objectives were derived.

1.4. Objectives of the study

1.4.1. General Objective

- To determine forest structure and provisions of ecosystem functions in a changing environment of the Western Escarpment of the Central Rift Valley and the Gibe Watershed, Ethiopia.

1.4.2. Specific Objectives

- To determine species diversity of the Biteyu forest.
- To identify major environmental variables, which determine the pattern of species distribution and community formation in Biteyu forest.
- To analyze the forest structure of Biteyu and Boter-Becho forests.
- To estimate carbon stocks in soil, AGB, BGB, forest litter and quantify the distribution of carbon stock in different tree size and height classes of the two forests.
- To estimate the carbon storage and sequestration potential of the two forests.
- To examine the effect of terrain variables (altitude and slope) on the carbon pools of both forests.
- To investigate litterfall, leaf litter decomposition, nutrient release pattern and temporal variation in litterfall production in the Boter-Bocho forest.

Note: As I tried to mention above in the objectives, there are parts included and considered in both (Biteyu and Boter-Becho) forests, and parts only considered for Biteyu or Boter-Becho forest in this study. These include the following major points.

- Floristic composition was considered only for the Biteyu forest but not for the Boter-Becho forest. This is because another student is studying the floristics of Boter-Becho forest at the same time in the same department by the same supervisors.
- Forest structure was analyzed for both forests considering woody species into account.
- Total Carbon stocks were estimated for both forests.
- Litterfall and litter decomposition dynamics, and nutrient release patterns were studied only for Boter-Becho forest. This was because of the fact that it was first planned to study in both Biteyu and Boter-Becho forests but failed to do so in Biteyu forest due to intensive cattle interference, illegal logging, grazing and browsing activities, which prohibited littertrap and litterbag experiment. Thus, to fulfil this gap, two sites were selected from Boter-Becho forest based on disturbance level determined subjectively by field observation.

CHAPTER TWO

2. LITERATURE REVIEW

2.1. Forest structure of Afromontane forests

Understanding the pattern of forest structure and composition has considerably important implication on population dynamics (Clark, 1991). However, Cabin *et al*. (2002) noted that fragmentation and habitat loss could influence the structure and regeneration of afromontane forests. Accordingly, regeneration of woody species in the forest is strongly influenced by human-induced disturbances, which in turn determine the forest structure and floristic composition (Cotler and Ortega-Larrocea, 2006). Moreover, it often leads to altered environmental conditions, which influence the process that can both augment and erode species diversity in the tropical forest community (Kennard *et al.,* 2002; Sapkota *et al.,* 2010).

With regard to Ethiopia, historical records show that forests used to cover large areas in the Ethiopian highlands. However, there are controversies about the extent of the former forest cover in Ethiopia. Accordingly, the natural forests are assumed to have covered 37% of the Afromontane region of Ethiopia in the past, which declined to only 4.4% by 1960 (EFAP, 1994; von Breitenbach, 1962). Destruction of the remnant high forests continues at an estimated rate of 150,000 - 200,000 ha per year (EFAP, 1994; Reusing, 1998). According to the estimate by Earth trends in 2000 cited in the report by Gatzweiler (2007), Ethiopia had 4,344 million ha of natural forest area, 4% of the country's landmass. In between 1990–2010, 2.91 million ha of the forest cover in Ethiopia was deforested with an average of 140,900 ha or 0.93% per year (FAO, 2010). However, according to Forest Sector Management in MEFCC (2015) the forest cover of Ethiopia was estimated to be 12.3 million ha (11%) with the new definition of forest to include dense woodlands found in Gambella and Benishangul Gumuz Regional States. Here, forest is defined as land spaninng atleast 0.5 ha covered by trees and bamboos with a height of higher than atleast 2 m and a canopy cover of at least 20% or trees with the potential to reach these thresholds in situ. However**,** the above forest definition differs from the definition used for international reporting to the Global Forest Resources Assessment (FAO) and from the forest definition used in the National Forest Inventory which both applied the FAO forest definition with the thresholds of 10% canopy cover, a 0.5 ha area and a 5 m height (FAO, 2010).

Currently, the remnants of the original Afromontane forest species are largely restricted to churchyards and other sacred groves in a matrix of cropland and semiarid degraded savanna in the highlands of northern Ethiopia (Aerts, 2006). This is because highlands of Ethiopia, in contrast to most mountain systems outside Africa, are very suitable for human settlement. This population pressure on the highlands accompanied by sedentary agriculture, extensive cattle herding activities and political factors has resulted in heavy deforestation, forest fragmentation, and loss of biodiversity and impoverishment of ecosystems in general (Eshetu Yirdaw, 2002) which in turn will affect the forest structure.

Despite their economic and environmental value, the remaining forests in Ethiopia are under threat. The growing population requires more fuel wood and more agricultural production, in turn creating needs for new farmland and timber. These factors currently result in deforestation and forest degradation by affecting the structure of the existing forest in the country. Accordingly, the forest degradation level due to firewood consumption is expected to increase in the same proportions as the Ethiopian population (2-3% per year until 2030) in a business as usual‘scenario (MOA and EPA, 2013). Therefore, Ethiopia has chosen REDD+ as a climate change mitigation mechanism for the forest sector (Federal Democratic Republic of Ethiopia, MOA and EPA, 2013) among other things. However, there is very few scientific works so far known to have focused on the potential role of afromontane forests of Ethiopia in the global carbon cycle in general and, plant biomass and productivity in particular. Improvement in our understanding of the carbon pool and dynamics of such forests would be useful for developing better policy decisions related to forest utilization/conservation and more importantly in the management of climate change.

2.2. Multivariate Analysis in Community Ecology

According to Green (1979), multivariate analysis of community data is very challenging due to various reasons. Four of the reasons as per the author are described as follows. a/ community data is very complex due to the fact that it involves noise, redundancy, relationship and outliers; b/ community data are so large in which even the modest study with 100 samples and 100 species has a data matrix of 10,000 entries which is quite difficult, c/ the investigator may have a particular question in mind up on which a community data bear informatively and but indirectly. Often the best method for obtaining the desired information is rather to derive the desired information by means of multivariate analysis. This alternative is motivated and encouraged because of the advantages of community samples, including objectivity, speed and low cost, effective correlation with other factors that are more elusive to observe directly, relevance to a variety of present or potential research interests, compatibility and comparatively with other related studies based on community samples, d/ the variety of dataset, research purposes, and required formats for presentation of results demand both a variety of multivariate methods for choosing an appropriate method or methods. To facilitate communication among ecologists, there is a need for the selection of only a modest number of preferred methods for the common usage (Pielou, 1977). In order to meet these challenges, ecologists have developed, applied and tested a number of multivariate methods among which the two are classification and ordination.

2.2.1. Classification

Classification can be defined as an act of putting things in groups. In community ecology, more often, the "things" are samples or communities. It involves grouping similar entities together mainly in the form of clusters (Poore, 1962; Sokal, 1974). In community ecology, there are three types of classification namely table arrangement (Braun-Blanquet, 1932), non-hierarchical classification (Gauch, 1982) and hierarchical classification (Gauch and Whittaker, 1981).

Table arrangement orders the samples by species data matrix by placing samples and species into the order that best reveals the intrinsic structure of the data (Braun-Blanquet, 1932; Mueller-Dombois and Ellenberg, 1974). Similar samples in composition and similar species in distribution are brought close together. It was described that the non-zero data matrix entries are thereby concentrated into blocks, and lines may be drawn in the matrix to mark off sample and species clusters (Hill, 1979). According to Gauch (1981), non-hierarchical classification merely puts similar samples or species into the clusters and the researcher may control the number of clusters produced. However, hierarchical classifications put similar samples or species into groups and further arrange the groups into a hierarchical tree like structure called dendrogram. Hence, dendrogram indicates the relationships among the groups (Everitt, 1978; Gauch and Whittaker, 1981).

Hierarchical classifications are of two kinds: divisive and agglomerative. A divisive hierarchical classification constructs the classification from the top to the bottom. It starts with the entire set of samples, and gradually divides it into smaller and smaller groups. An agglomerative hierarchical classification bottom-up approaches which produces clusters by successively merging pairs of clusters that are closest to each other (Zerihun Woldu, 2016; Gauch, 1982). It starts with small groups of samples, and gradually groups them into larger and larger clusters, until the entire data set is sampled (Pielou, 1984). In general, classification is important for four aspects of data in community ecology. These are a/it reduces noise by combining the samples of a cluster into a single average or composite sample (Gauch, 1982), b/it is obviously effective in summarizing redundancy in the data, c/table arrangements in classification reveal relationships among samples and species, non-hierarchical classifications indicate clusters of similar samples but do not indicate the larger picture of relationships among clusters, d/ it can easily detect outliers simply by nothing samples that fail to cluster with other samples at a given, fairly, low level of similarity (Gauch, 1982).

2.2.2. Ordination

Goodall (1954) proposed the term 'ordination' and he introduced the Principal component analysis (PCA) into ecology. Ordination is a research tool for the interpretation of field data on the plant and animal assemblages and their environment (ter Baak, 1994). Hill

(1973) defined ordination as arrangement of samples or sites along gradients on the basis of their species composition or environmental attributes. According to many other authors (Everitt, 1998; Pielou, 1984; Peet, 1980), ordination helps to summarize community data (such as species abundance data) by producing a low-dimensional ordination space (usually 2-dimesnsional) in which similar species and samples are plotted close together, and dissimilar species and samples are placed far apart. The data set reduced in the ordination is most useful for investigating possible structure in the observations. More importantly, the graphical representations from most ordination techniques lead to intuitive interpretations of species-environment relationships in addition to pattern detection and hypothesis generation (Everitt, 1998).

It is known that ordination methods are used to describe relationships between species composition patterns and the underlying environmental gradients, which influence these patterns by asking the question "*what factors really structure the community?'*. For instance, if an ecologist wanted to examine the distribution patterns of vegetation in a given study area, ordination could be used to determine which species are commonly found associated with one another, and how the species composition of the community changes with increase in environmental gradients (See Gauch, 1982). Ordination methods can be divided in two main groups, direct and indirect methods (figure 1). Based on the underlying model they use for the species responses along environmental gradients, both groups of methods can be subdivided into linear or non-linear, with unimodal response model being a case of particular ecological interest and the form in which they use the species data (Jongman *et al*., 1987; ter braak, 1987, 1994; Clarke, 1993).

Direct Gradient Analysis methods, the so-called constrained ordination method, uses species and environment data in a single, integrated analysis (ter Braak, 1994; Oksanen, 2012). The method studies only the variation that can be explained by the available environmental variables. It also tries to explain differences in species composition between sites by differences in environmental variables and hence, examine the relationship between environmental variables and species composition. The method does not use similarity indices; rather the ordination is based on the raw data matrix. Direct gradient analysis or constrained ordination methods include constrained (or canonical) correspondence analysis (CCA), redundancy analysis (RDA) and distance-based redundancy analysis (db-RDA) (Gauch, 1982). In these methods, constraining, means that ordination only shows the community variation, which can be explained by external environmental variables or constraints (Jongman *et al*., 1987; ter Braak, 1994). Indirect Gradient Analysis or the so-called unconstrained ordination method constructed in such a way that they "best" explain the species data only. The methods are based only on similarity matrices calculated from the species abundances (Palmer, 1993). In indirect gradient analysis, environmental variables associated with each stand can be overlaid onto the ordination plot. The inputs of environmental data, which might or might not be relevant to the species distributions, do not influence the ordination itself (ter Braak, 1994). This method is viewed as like regression analysis but with the major difference in ordination is that the explanatory variables are not known environmental variables but they are called theoretical '*latent'* variables (ter Braak, 1994; Gauch, 1982). Unconstrained ordination include principal component analysis (PCA), correspondence analysis (CA), Deterended correspondence analysis (DCA), Non metric multidimensional scaling (NMDS), principal coordinate analysis (PCoA) or metric multidimensional scaling and polar ordination (Bray- Curtis ordination) (Gauch, 1982).

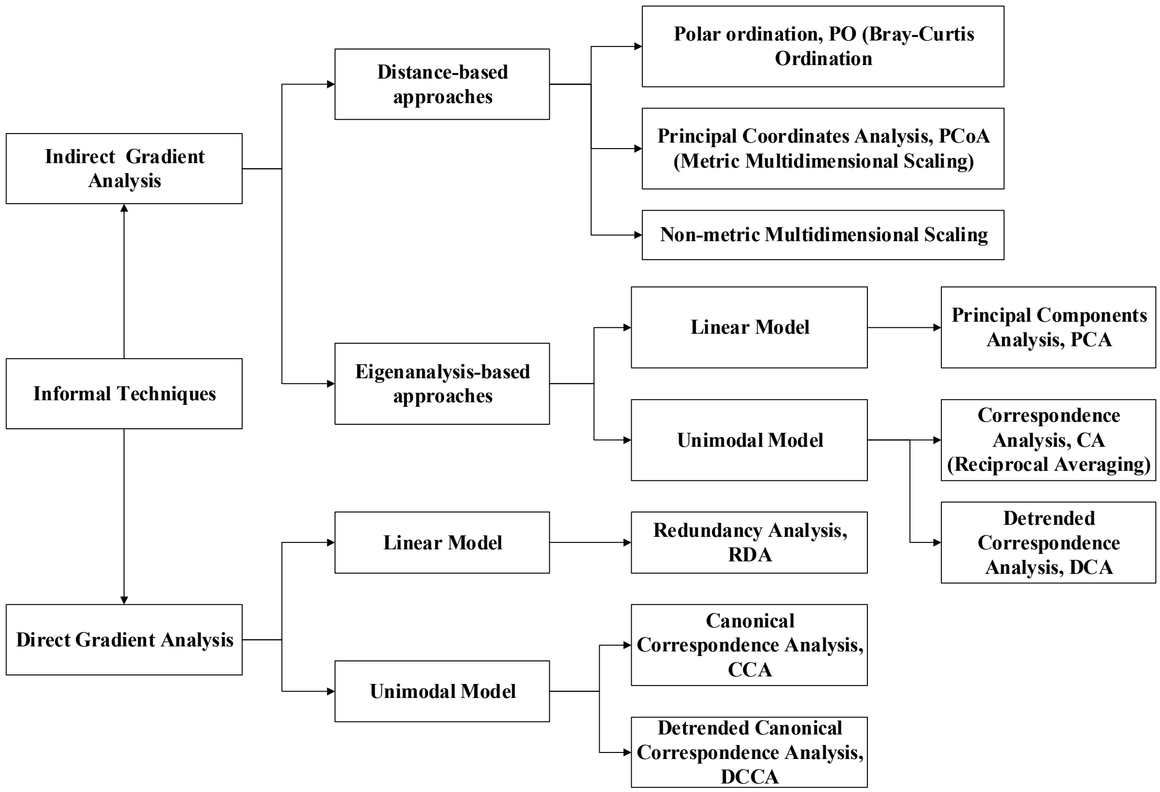


Figure 1. Different groups of Ordination techniques (source: Gauch, 1982)

2.2.3. Measures of species diversity

According to Paul (1993) and Whittaker (1972), species diversity can be interpreted in terms of species richness and evenness. Widely known measures of species diversity in studies of ecology include alpha, beta, and gamma types (Whittaker, 1972). The number of species per community or per qudrat refers to alpha diversity (species richness) whereas the difference in species diversity between communities or areas refers to beta diversity. Beta diversity also represents differences in species composition between environments and sometimes called habitat diversity. A measure of diversity of a landscape, or geographic area is called total or gamma diversity and it is a product of the alpha diversity of its communities and the degree of beta differentiation among them (Whittaker, 1972). There are a number of diversity indices, which have been used to measure species diversity in ecology and other related fields of study.

According to Magurran (2004), most methods used two major components in measuring diversity. These are species richness and relative abundance or evenness of species within a given sample or community. According to Whittaker (1972), a strong measure of species diversity is species richness. The widely used indices of diversity in ecology are the Shannon Wiener index and Simpson's index of diversity. These indices combine species richness with relative abundance in measuring species diversity (Kent and Coker, 1992). The Shannon index explains the relative abundance or equitability of species while Simpson's index gives weight to dominant species (Simpson, 1949; Whittaker, 1972). According to Magurran (2004), the diversity indices are biased on either species richness or species evenness making it difficult to obtain one robust index of diversity measurement. The Shannon index of species diversity is insensitive to rare species (Sanjit and Bhatt, 2005).

2.3. Forests' Role in Climate change Mitigation

Anthropogenic emissions of the greenhouse gas, carbon dioxide, into the earth's atmosphere continue to rise. Recently, understanding the role of forests in carbon cycles and predicting whether they will be carbon sinks or sources in the future are important to ongoing international dialogue on the subject of climate change (IPCC, 2007). Sixty percent of the earth's terrestrial carbon stocks are contained by forests of the world

(McKinley *et al.*, 2011) and account for about 80% of carbon exchange between terrestrial ecosystems and the atmosphere (Pearson *et al.,* 2005). It stores 1.1 ± 0.8 gigaton carbon annually in biomass, soils, and organic matter (Pan *et al*., 2011). It has been studied that mature rain forests contain about 350-500 t ha^{-1} only in the AGB with secondary forests attaining about 75% of these values before further logging (Brown and Lugo, 1984; Palm *et al*., 1986). Moreover, Brown (1997) stated that trees in tropical forests constituted up to 97% of AGB while the understory vegetation constitutes less than 3%. Sierra *et al*. (2007) estimated a mean total carbon stock of 228.2 ± 13.1 t C ha^{-1} for tropical secondary forests, of which 84% was in the soil organic carbon pool, 5% in the total belowground biomass, 9% in the total AGB and 1% in the total standing litter.

Tropical forests have great potential for the mitigation of CO_2 if appropriate conservation and management takes place (FAO, 1997). For instance, Lewis *et al*. (2009) estimated that tropical forests of Africa store 24-39 Gt of carbon and yield 0.34 Gt C yr^{-1}. In addition, Lewis *et al.* (2013) stated that African tropical forests are characterized by relatively high AGB at 395.7-ton ha^{-1}. The majority of the areal extent of African closed-canopy forest is located in Central Africa which is having higher (429) ton dry mass $ha^{-1}$and statistically indistinguishable from the high AGB stocks of the forests of Borneo at approximately 445 ton ha^{-1}. These African and Asian values are significantly higher than the forest AGB reported from a synthesis across Amazonia at 289 ton ha^{-1}. These results show that there is a difference between generally higher AGB palaeo-tropical forest versus generally lower AGB neo-tropical forest, which supports recent studies showing neo-versus palaeo-tropical differences in stem allometry, basal area (Banin *et al.,* 2012) and AGB (Feldpausch *et al.*, 2012) based on more limited African data.

Moreover, the study conducted by Baccini *et al*. (2008) showed that the average above-ground biomass estimates for tropical forests of Africa range from 85 ton ha ^{1}for closed deciduous forests to 251 ton ha ^{1}for swamp forests. Lianas are other important components of the AGB in tropical forests, as they possess up to five times the leaf mass of trees of the same diameter at breast height (Vargas *et al*., 2008). The amount of C stored in the surface litter in moist broad-leaved evergreen tropical forests is small and ranges between 100 and 500 g C m ^{2} (Raich *et al.,* 2006). Despite its less diversity compared with South America and Asia, tropical African forest stores about 96% of carbon and the remaining 4% is in the grassland and shrub savanna biomass (Gaston *et al*., 1998).

Houghton and Hackler (2006), for instance, estimated about 15% of the global net flux of carbon from the land use change is in Sub-Saharan Africa in the 1990s. As described in Hulme *et al.* (2005), the African continent has been warming at the rate of about 0.5 ºC per century during the 20th century, with larger warming in June August, and it is predicted that Africa will be between 2 6 ºC warmer in 100 years. This phenomenon might relatively imply the vulnerability of ecosystems in Africa. Compared with other continents, African people has quite close dependencies on natural resources and a future with substantial industrial, agricultural and social development. This means that more impacts from human beings will be imposed on African forests, and the corresponding response from forest. Both of the natural and anthropogenic factors emphasize that the researches of carbon dynamics on African forests and a need to predict the warming impact on carbon storage and fluxes in African forests are required urgently.

Since tropical forests have been subjected to various natural and anthropogenic disturbances such as agricultural practices and deforestation, various stages of forest development exist in tropical forested regions of Africa. Secondary forests recovering from disturbances assimilate carbon in tree tissues and soils by attaining a positive net balance between photosynthesis and respiration and are capable of rapidly transporting carbon from the atmosphere to the biosphere (Lugo and Brown, 1992; Dixon *et al.,* 1994).

Several authors (Ringius, 2002; Vagen *et al.,* 2004; Williams*, et al.,* 2008) argue that Africa has great potential for mitigating climate change due to the fact that 16-17 % of global forest area is located on the continent and forest carbon stocks can rise up to 300 ton C ha^{-1} (FAO, 2006; 2011). Managing this forest resource for the sake of carbon conservation and management is quite important to slow down the increase of CO_2 in atmosphere (Malhi *et al*., 2002) and get revenue from carbon trade. However, they are not getting the important concern to implement the necessary actions for their effective participation from the developed countries. This is mainly because of the absence of scientific data on forest carbon stocks, which largely limited their capacity to assert their interest in the international negotiations on the issue of climate change.

2.3.1 The Global Carbon cycle in Forest Ecosystems

Globally, the carbon cycle plays a key role in regulating the Earth's climate by controlling the concentration of carbon dioxide in the atmosphere (IPCC, 2007; Houghton, 2007). Of the C present in the world's biota, 99.9% is contributed by vegetation and microbial biomass regardless of a negligible C reservoir of animals. Carbon dioxide is important because it contributes to the *greenhouse effect*, in which certain gasses trap heat generated from sunlight at the Earth's surface, and prevented from escaping through the atmosphere. The greenhouse effect itself is a perfectly natural phenomenon and, without it, the earth would be a much colder place. But as is often the case, too much of a good thing can have negative consequences, and an unnatural build-up of greenhouse gasses can lead to a planet that gets unnaturally hot (Houghton *et al*., 2009).

Global carbon is partitioned into the major carbon reservoirs (Figure 1), which are estimated to be oceanic (38,000 Pg); geologic (5,500 Pg); pedologic, or soils-based (2,700 Pg). Of the soil-based carbon, 1,650 Pg in SOC and 1,050 Pg in SIC form (www.globe.gov/projects/carbon). Moreover, the atmospheric carbon (750 Pg), is increasing at the rate of approximately 4.5 Pg yr^{-1}; and biotic C (560 pg) (Lal, 2004; IPCC, 2006, 2007; Houghton *et al*., 2009; Morgan *et al.,* 2010). The exchanges of carbon between the atmosphere and terrestrial ecosystems (soil and vegetation) is critical to the patterns of carbon dioxide concentration in the atmosphere (Luo and Zhou, 2006; Houghton, 2007; IPCC, 2007). Carbon is returned to the atmosphere via three of the largest fluxes, oceanic release (~90 Pg C yr ^{1}), plant respiration (~59 Pg C yr ^{1}), and soil respiration (~58 Pg C yr ^{1}) (Schlesinger, 1997; IPCC, 2000; Luo and Zhou, 2006; Houghton, 2007). These fluxes are mainly countered by plant photosynthesis, which absorbs about 120 Pg C yr ^{1} from the atmosphere and ocean uptake about 92 Pg C yr ^{1}. The earth's carbon reservoirs naturally act as both sources, adding carbon to the atmosphere, and sinks, removing carbon from the atmosphere (Lal, 2006).

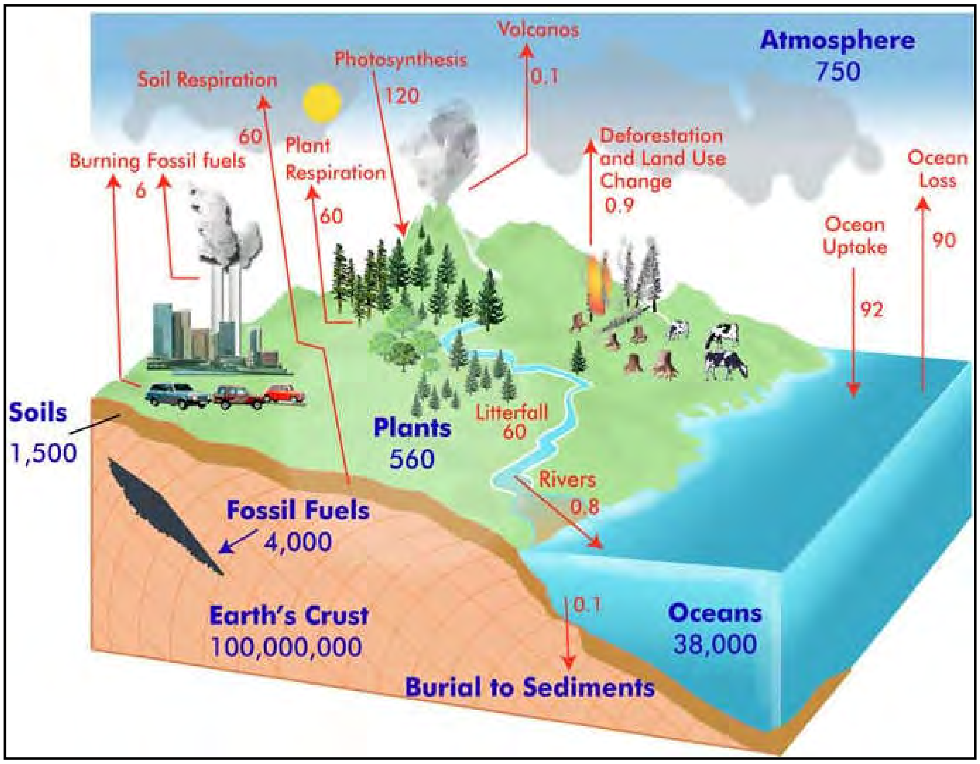


Figure 2. A simplified diagram of the global carbon cycle. Pool sizes, shown in blue, are given in gigatons of carbon. Fluxes, shown in red, are in Pg (petagram) per year. (*www.globe.gov/projects/carbon*)

2.3.2. The International Discussions on Climate change

Very recently, climate change is becoming already a reality in many localities of Africa including Ethiopia. The impacts of climate change such as seasonal shift, starvation and catastrophic social and economic disruption are currently happening in Africa as described by Henry (2010). According to Karsenty *et al*. (2003) spread of desertification, poverty of much of the population, which depends largely on the use of natural resources, the weakness of its economy and institutions, lack of government's capacity to shape the problems have been a worrying situation. In order to reduce the climate effect, international discussions and attempts have been made by various international stakeholders. For instance, the UNFCCC launched a call to promote actions in the African continent during the COP12 in Nairobi. However, it remains largely unimplemented. There were about 2.6% of the CDM (clean development mechanisms) projects in Africa and only one forestry project from Ethiopia was found to be successfully implemented (UNFCCC, 2007).

A set of guidelines for estimating greenhouse gas inventories at different tiers of quality has also been produced at the global level by IPCC; ranging from Tier 1 (simplest to use; globally available data) up to Tier 3 (high-resolution methods specific for each country and repeated through time) (Penman *et al.,* 2003; IPCC, 2006). Accordingly, ground-based measurements of tree diameters and height can be combined with predictive relationships to estimate forest carbon stocks (Tiers 2 and 3). Moreover, remote-sensing instruments mounted on satellites can estimate tree volume and other proxies that can also be converted using statistical relationships with ground-based forest carbon measurements (Tiers 2 and 3) (Gibbs *et al*., 2007).

As an alternative to forest degradation in developing countries, carbon credit programs have been proposed to industrialized countries. The United Nations Framework Convention on Climate Change (UNFCCC, 2011) promotes carbon-offset programs by limiting forest based GHG emissions to curb global warming. Negotiations being made under the UNFCCC have helped raise the profile of forests and forests' contribution to offsetting GHG emissions. Reducing Emissions from Deforestation and forest Degradation (REDD+) is a carbon offset mechanism developed in 2005 that provides financial incentives to countries preserving tracts of forests, thereby reducing emissions through avoided deforestation (UN-REDD, 2011). REDD+ is considered not only to enable developing countries to contribute to a reduction in emissions under future arrangements to the UNFCCC, but also to strengthen SFM at local and national levels. Accordingly, if a country proves a reduction in GHG emissions through avoided deforestation, they can sell the avoided carbon emissions as carbon credits on an international market. However, quantifying forest carbon stocks, a piece of information critical to emission reduction is a major scientific challenge and a setback to REDD implementation (Gibbs *et al*. 2007; Kohl *et al*., 2009; Saatchi *et al*., 2011). Ethiopia has been in progress to implement the REDD+ by including it in the country's Climate Resilient Green Economy Strategy (2011) in addition with other sustainable forest management mechanisms to reduce the climate change effect.

Paris Agreement in COP 21st added Article 5.1 and puts in place an expectation that Parties should take action to 'conserve' and 'enhance' sinks and reservoirs of greenhouse gases such as biomass, forests and oceans as well as other terrestrial, coastal and marine ecosystems. Accordingly, the land sector and forests, and its long-term goals are dependent on sequestration, which will be achieved mostly through natural processes via forests and oceans. There is no question that taking better care of the world's remaining forests and natural ecosystems is good for people and for the climate and will contribute to meeting multiple sustainable development goals. In addition, Paris Agreement of COP 21st also included Article 5.2 to encourage the implementation and support of REDD+, and it provides international endorsement of both REDD+ and of a joint mitigation and adaptation approach to the integral and sustainable management of forests. Moreover, it reaffirms the importance of non-carbon benefits, which confirms the broader scope of REDD+ to be a market and a non-market mechanism as much applicable to adaptation actions as it is to mitigation.

2.4. Forest Carbon Pools

At the global level, 19% of the carbon in the earth's biosphere is stored in plants and 81% in the soil. In tropical, temperate and boreal forests together, approximately 31% of the carbon is stored in the biomass and 69% in soil (IPCC, 2000). Tropical forests worldwide, stored about 50% of the total carbon in AGB and 50% in the top 1 m of the soil despite the marked differences among different sites (IPCC, 2000). For instance, moist tropical forests in Africa had more than three times as much carbon in AGB as in soil to 1 m depth (Djomo *et al*., 2010), but a montane forests of Peru had twice as much carbon in soil as in AGB (Gibbon *et al*., 2010).

The amount of carbon sequestered or lost by a forest can be carbon sink or source and forest carbon can be estimated from the biomass accumulation since approximately half of forest dry biomass weight constitutes carbon (IPCC, 2006; FAO, 2005). According to the IPCC (2006), five carbon pools of terrestrial ecosystem involving biomass are the aboveground biomass, belowground biomass, the dead mass of litter, woody debris and soil organic matter. Of the carbon pools, soil and vegetation on the earth's land surface store three times the carbon present in the Earth's atmosphere (IPCC, 2001). According to Malhi *et al.* (2009) and Sierra *et al*. (2007), carbon is stored in forests mainly in live biomass and soils, with smaller amounts in coarse woody debris and forest litter. The variation in carbon storage among tropical forests is caused by the variation in a number of factors such as climate, disturbance, species composition, successional stage, and soil fertility (Wright, 2005).

Any changes in the land use such as forest degradation and deforestation have a direct impact on this component of the carbon pool. The belowground biomass, which constitutes all the live roots (IPCC, 2006) plays an important role in the carbon cycle by transferring and storing carbon in the soil. The dead mass of litter and woody debris contribute merely a small fraction to the carbon stocks of forests (Ravindranath and Ostwald, 2008). Soil organic matter is also a chief contributor to the carbon stocks of forests (Lal, 2005), next only to the AGB. Soils are assumed to have been major source of carbon emissions following deforestation (Ravindranath and Ostwald, 2008; Page *et al*., 2002) eventhough it varies from continent to continent.

2.4.1. Estimation of Forest Carbon Pools

2.4.1.1 Aboveground biomass (AGB)

The most comprehensive method of accounting the carbon pool of AGB is destructive sampling, whereby vegetation is harvested, dried to a constant mass and the dry-to-wet biomass ratio determined (Chave *et al*., 2014; Lu, 2005; Devi and Yadava, 2007). The destructive method or the harvest method, is the most direct method for estimation of AGB and the carbon stocks stored in the forest ecosystems (Gibbs *et al*., 2007). This method of biomass estimation is limited to a small area or small tree sample sizes. Despite the accurate determination of the AGB for a particular area, this method is time and resource consuming, strenuous, destructive and expensive, and it is not feasible for a large scale analysis and needs special permission. Moreover, it is not environmentally friendly to cut large trees to develop allometric models at this time in the study forests where the environmental issues are becoming more challenging. This method is also not applicable for degraded forests containing threatened species (Montès *et al*., 2000).

According to various authors (Chave *et al.,* 2005; Segura and Kanninen, 2005; Navar, 2009) this method is used for developing biomass equation to be applied for assessing biomass on a larger-scale.

There is also tree biomass estimation using the non-destructive method. This method estimates the biomass of a tree without felling by the help of allometric equations, which use only the indicator parameter obtained from the forest inventories. The non-destructive method of biomass estimation is applicable for those ecosystems with rare or protected tree species where harvesting of such species is not very practical or feasible as it is the case for the present study. For instance, Montes *et al*. (2000) developed a non-destructive method for AGB estimation of *Juniperus thurifera* woodlands in the High Central Atlas, South of Morocco. According to this author, the biomass of the individual tree was estimated by taking into account the tree shape (by taking two photographs of the tree at orthogonal angles), physical samples of different components of the trees such as branches and leaves and dendrometric measurements, volume and bulk density of the different components. Although, it is a non-destructive method, to validate the estimated biomass, the trees had to be harvested and weighted. Similarly, another way of estimating the aboveground biomass is non-destructive method, which are climbing the tree to measure the various parts (Aboal *et al*., 2005), measuring the diameter at breast height, height of the tree, volume of the tree and wood density (Ravindranath and Ostwald, 2008) and calculate the biomass using allometric equations (Brown *et al*., 1989; Hughes *et al*., 1999; Chave *et al.,* 2005). Even though, these methods do not involve felling of tree species, it is not easy to validate the reliability and can involve a lot of labour, time and climbing. Moreover, for estimating the biomass density of trees, two approaches have been more commonly applied. The first one directly estimates biomass density through allometric equations and the second one converts wood volume estimates to biomass density using biomass expansion factors as described in Brown (1997).

In forest ecosystems, the stock of tree AGB held in vegetation are usually inferred from ground census data and the most widely used method for estimating biomass of forest is through allometric equations. The allometric equations have been developed and applied to forest inventory data to assess the biomass and carbon stocks of forests (Brown and Lugo, 1982; Brown *et al.,* 1989; Brown, 1997; Chave *et al*., 2005; 2014). Accordingly, tree biometric measurements are converted into biomass values using an empirical allometric model (Brown *et al.,* 1989; Brown, 1997) though the quality of these allometric models have some limitations in assessing AGB stocks (Chave *et al*., 2004, 2005, 2014; Baccini and Asner, 2013).

Many researchers have developed generalized biomass prediction equations for different types of forest and tree species (Brown *et al*., 1989; Brown, 1997; Nelson *et al*., 1999; Montes *et al*., 2000; Ketterings *et al.,* 2001; Chave *et al*., 2005; Basuki *et al*., 2009; Navar, 2009; Chave *et al*., 2014).

The allometric models developed by Brown (1997) have been proposed to be used depending on vegetation type and on the availability of total tree height information. As a compromise between environmental variation and data availability at the time, Brown (1997) proposed a classification of tropical forests into three forest types; dry, moist, and wet, following the Holdridge life zone system (Holdridge, 1967). This estimate of Brown's biomass equation takes into account diameter at breast height, total height and wood density and Holdridge life zone (Table 1). Moreover, Nelson *et al*. (1999) conducted a study to develop species-specific and mixed-species allometric relationships for estimating total aboveground dry weight using eight abundant secondary forest tree species in the Amazon.

The total AGB of a moist tropical forest in South-western Cameroon was estimated by Djomo *et al*. (2011) using a locally developed mixed species allometric equation. Since the forest biomass estimates vary with age of the forest, site class and stand density the choice of allometric equations has a significant effect on the biomass calculations of the forest ecosystem. Hence, the generalized allometric equations available for large landscape scales should be used with caution as the site greatly influences allometric relationships (Montagu *et al.,* 2005). On top of this, Ryan *et al*. (2011) carried out a study to quantify the forest carbon stock in Miombo woodland in Mozambique and developed a new site-specific allometric equation, between stem diameter and tree stem based on destructive harvest of 29 trees. Furthermore, Kenzo *et al*. (2009) harvested 136 trees from 23 species to measure the above-ground biomass in various tropical secondary forest trees in Sarawak, Malaysia. They also developed allometric relationships between the stem diameter at breast height, stem diameter at ground and leaf, stem and total root biomass. Their study also showed a relatively high correlation of allometric relationships between the tree height and plant-biomass. According to the studies conducted by Vielledent *et al*. (2012), when biomass allometric models are not available for a given forest site, a simple height-diameter allometry is required to estimate the biomass and carbon stocks accurately from plot inventories.

Recently, the Chave *et al's* model should be especially valuable in Africa, considering that the continent's tropical forests represent 30% of the global total and that almost no allometric models are available for Africa except very few species specific models (Henry *et al.*, 2010). Because of this, Africa is considered to be the least known in terms of C stocks and rates of conversion (Baccini *et al.*, 2008) though it is the second largest block of rainforest in the world. The forest carbon stocks are widely estimated from the allometric equations for forest biomass and carbon concentration of the different parts of a tree is assumed to be 47-50% of the biomass (Brown, 2002; IPCC, 2003, 2006). Therefore, the forest biomass estimation can be worked out using any of the methods or a combination of the methods mentioned. At the same time, while choosing a method for biomass estimation one should keep in mind the applicability or the suitability of that method for the area or forest type or tree species used. Some of the workable allometric equations selected for different forest types are presented in Table 2 below.

Table 1. Various Allometric Equations used for forest Biomass Estimation

Equation	**Eco-Climatic Zone**	**References**
$AGB = 0.136\ (DBH)^{2.320}$	Global Dry forest	Brown (1997)
$AGB = 0.118\ (DBH)^{2.530}$	Global moist forest	Brown (1997)
$AGB = 21.297\text{-}6.530DBH + 0.740DBH^2$	Global wet forest	Brown (1997)
$AGB = 42.69\text{-}12.800(DBH) + 1.242(DBH^2)$	General	Brown (1997)
$AGB = 0.044\ (DBH^* H ^*\rho)^{0.972}$	Pantropical	Brown *et al.*(1989)
$Y = 34.4703 - 8.0671\ (DBH) +0.6589(DBH^2)$	Dry (RF < 1500 mm)	Brown *et al.*(1989)
$AGB = 0.0559^*(\rho DBH^{2^*}H)$	General	Chave *et al*. (2014)
$AGB = 0.112\ (DBH^2{}^* H ^*\rho)^{0.916}$	Global dry forest	Chave *et al.*(2005)
$AGB=0.135DBH^{2.420}$	Neo-tropical	Chave *et al.* (2001)
$AGB=0.125DBH^{2.562}$	Pantropical moist	Djomo *et al*. (2010)
$AGB= 0.051(DBH^2H)^{0.930}$	Western Kenya	Henry *et al.*(2009)
$AGB = \exp(\ \ 1.997 + 2.413 \times \ln(DBH))$	C. Amazonia	Nelson *et al*. (1999)

Nowadays, Chave *et al.* (2014) improved their equation developed in 2005 by greatly increasing the sampling effort in both dry and wet vegetation types and the recent equation is being proposed by IPCC to be used in the REDD projects. Equations developed by Chave *et al*. (2005) were with data from natural forests growing in tropical climates, excluding plantations or managed forests. Chave *et al*. (2001) equation was developed from datasets spanning moist to wet tropical forests while equations by Brown (1997) were developed for broadleaf forests from a database that included trees with DBH 5-148 cm.

The challenge to estimate C stocks and emission factors in complex vegetations of Sub-Saharan Africa is due to lack of biomass equations. In particular, Sub-Saharan Africa faces important gaps related to the understanding of the contribution of the African ecosystems to the C cycle. White (1981) identified 80 different ecological zones with different vegetation structures and floristic compositions and hence different C stocks. The use of biomass allometric equations or stand tables is the starting point for estimating the biomass or the C stock of a tree or a stand. Very few tree biomass allometric equations were reported for Sub-Saharan Africa (AGO, 2002; Ponce-Hernandez, 2004; IPCC, 2007). Only 1% of the necessary tree allometric equations are currently available in this region (Henry *et al*., 2011). While inventories of allometric equations were performed for Europe (Zianis *et al.,* 2005) and for South-America (Návar, 2009), no inventory exists for Sub-Saharan Africa. According to Chave *et al.* (2005), there are to date no destructively sampled trees to develop allometries for forests of Sub-Saharan Africa. The national communication for the Forest Resource Assessment of the FAO analyzed that 13 countries used biomass coefficient for the national biomass accounting and only seven of them used national specific data while the others used data from FAO, IPCC and Millington *et al.* (1994). Even the default values proposed by the IPCC do not cover all the ecological zones; for instance, no data were reported for tropical deserts and subtropical humid forests, which comprise 6,837 and 5,035 ha respectively.

In contrast, some countries in Sub-Saharan African such as Cameroon, Zambia established national forest inventories that can be used to estimate C stocks in their REDD projects. However, C stock assessment remains limited by the availability of data on wood density and biomass allometric equations in the region. Regarding countries that did not undertake national forest inventories, particularly the countries of the Congo Basin and Ethiopia, it is possible to assess biomass based on conventional forest inventories as recommended by various authors (Brown *et al.,* 1989; Brown, 1997; Chave *et al*., 2005). This approach is still limited by the fact that conventional national forest inventories were performed in specific ecosystems that are not representative to large areas, to commercial trees with diameter over the minimum diameter to be felled. In addition, estimating the total forest biomass using the commercial volume is a difficult exercise as the structure of the forest and the diameter distribution vary between forest ecosystems and very few biomass expansion factors were reported for natural forests in Sub Saharan Africa. Because of this, very few fragmented studies have been done only by using general allometric equations to predict the forest biomass and forest carbon in the region.

2.4.1.2. Belowground Biomass (BGB)

Belowground biomass is defined as the entire biomass of all live roots, although fine roots less than 2 mm in diameter are often excluded because these cannot easily be distinguished empirically from soil organic matter. Belowground biomass is an important carbon pool for many vegetation types and land-use systems and accounts for about 20% (Santantonio *et al.* 1997) to 26% (Cairns *et al*., 1997) of the total biomass mainly in the tropics. Belowground biomass accumulation is linked to the dynamics of aboveground biomass in the forest ecosystem. The greatest proportion of root biomass occurs in the top 30 cm of the soil surface (Bohm, 1979; Jackson *et al*., 1996). Re-afforestation of degraded land leads to continual accumulation of BGB whereas any disturbance to topsoil leads to loss of belowground biomass. According to Cairns *et al.* (1997), as BGB could account for 20–26% of the total biomass, it is important to estimate this pool for most carbon mitigation as well as other land-based projects. Stock change estimates in BGB are also necessary for GHG inventory at national level for different land-use categories such as forestlands, cropland and grassland (IPCC, 2006). Cairns *et al.* (1997) developed allometric equations for estimating belowground or root biomass of natural forests as indicated in the table 2 below.

Table 2. Allometric equations for estimating belowground or root biomass of Natural forests

Conditions and independent variables	Allometric Equation	Sample size	R^2
All forests, AGB	Y = exp[−1.085 + 0.9256 *ln (ABD)]	151	0.83
All forests, AGB and AGE	Y = exp[−1.3267+0.8877*ln (ABD)+0.1045*ln (AGE)	109	0.84
Tropical forests, AGB	Y = exp[−1.0587 + 0.8836 * ln (ABD)]	151	0.84
Temperate forests, AGB	Y = exp[−1.0587 + 0.8836 * ln (ABD) + 0.2840]	151	0.84
Boreal forests, AGB	Y = exp[−1.0587 + 0.8836 * ln (ABD) + 0.1874]	151	0.84
Tropical forests	$0.489 * AGB^{0.890}$	-	-

Note: *Y= root biomass in t ha^{-1} of dry matter, ln = natural logarithm exp = "e to the power of" AGB = AGB in t ha^{-1} of dry matter AGE = age of the forest, years (Source: Cairns et al., 1997; Mokany et al., 2006)*

2.4.1.3. Carbon in litter and SOM

The litter pool includes dead organic surface materials less than 10 cm diameter and the herbaceous pool (Ravindranath and Ostwald, 2008). Generally, it has been known that forest soils are an important carbon sink (Goodale *et al*., 2002). As tree species alter the SOM storage (Oostra *et al*., 2006; Vesterdal *et al*., 2008), the sink function of the forest soil may be increased by the appropriate choice of tree species. The decomposition of plant materials such as root or leaf litter can be the source of SOM in the soil (Schulze, 2002). On a global basis, the soil carbon pool is greater than that of the vegetation pool and there are considerable differences in the allocation of carbon between living biomass and soil pools, both within and between biomes (Schlesinger, 1997). For instance, Brown and Lugo (1982) found that while tropical forests accounted for 46% of the world's living terrestrial C pool, tropical soils accounted for only 11% of the world's soil C pool. Despite the differences of estimates by various researchers, the global estimate of soil organic carbon ranges from 684 to 724 gigaton of C in the upper 0.3 m, 1462 to 1548 gigaton of C in the upper 1 m, and 2376 to 2456 giaton of C in the upper 2 m (Batjes, 1996).

The soil organic matter and dead organic matter composed of intact litter (>2 mm) constitute about 50% or more of the total carbon pool in the tropical forests (Brown and Lugo, 1984; Vogt *et al.,* 1986). According to Theng *et al.* (1989), soil organic matter can be separated into cellular material "light fraction" and non-cellular humus material, "heavy fraction", which makes up 79-90% of total carbon in most soils. The light fraction is a minor proportion of soil organic matter by mass but includes microbial biomass and is an important pool in N and P dynamics in soils (Parton *et al*., 1987). In tropical forests, the decomposition rate of litter/light fraction is rapid, usually within a few months to a year or two depending up on chemical composition (Anderson and Swift, 1983). Therefore, the balance between litterfall inputs, decomposition, determines soil C densities and leaching losses; ecosystem processes that are largely controlled by the same environmental factors controlling biomass production (Brown and Lugo, 1982; Post *et al*., 1982). Soil organic carbon (SOC) dynamics and carbon flux from the soil can be significantly influenced by land-use and soil-management practices. Spatially distributed estimates of SOC pools and flux are important to understand the role of soils in the global carbon and its response to the climate change or variation (Post *et al*., 1990; Schimel *et al.,* 2000).

2.5. Deforestation and Forest degradation on the carbon stock

Deforestation and forest degradation influences the amount of biomass and carbon stored in vegetation (IPCC, 2001; Pearson *et al,* 2005; IPCC, 2007) and are major threats to biodiversity mainly in the tropics (Laurance *et al*., 1998; 2011). This is the case for the montane forests in Ethiopia in general and the study area in particular. Land use change due to deforestation and forest degradation contributes 0.3 to 3.0 Pg C yr^{-1} of the total fossil fuel emissions of the globe which is about 0.6 to 6 times the Kyoto protocol target for 2012 (Houghton, 2003, 2007; Sohngen and Brown, 2008) and this represents the most significant proportion of total emissions from many developing countries.

According to Kapos *et al.* (1997), forest degradation is known to alter the microclimate because of a higher insolation and wind penetration, which increases temperature and decreases humidity along forest edges. Moreover, forest degradation elevates the rates of tree mortality, which influence forest biodiversity and species composition (Laurance *et al.,* 1998). By altering the ecosystem structure, forest degradation can lead to changes in ecological processes such as litter production (Werneck *et al.,* 2001) and nutrient cycling (Laurance, 1999; 2008). Aboveground live biomass increases with succession in forest fragments (Vitousek and Reiners, 1975) because of the recruitment of old growth species characterized by a larger diameter and height (Clark and Clark, 1996) and canopy development. Thus, litter production is expected to increase with forest development (Songwe *et al*., 1988; Clark and Clark, 1996).

According to IPCC (2003), African forests contain large C stocks in biomass, up to 255 t C ha^{-1} in tropical rainforests. Compared to the other continents, Africa contributes less than 4% to the global anthropogenic fossil fuel emissions (Canadell *et al.,* 2009). However, about 40% of fire emissions have been attributed to the African continent significantly affecting the atmospheric chemistry (Kituyi, *et al.,* 2005; van der Werf *et al.,* 2006). Many authors (Cao *et al.,* 2001; Baker *et al.,* 2004; Williams *et al.*, 2007) estimated that about 50% of inter-annual variability of global atmospheric CO_2 is attributed to the variability of the African C balance. In Africa, the forestry and agriculture sectors including conversion to croplands and shifting cultivation together account for 75% of the total emissions from the region (Canadell *et al.,* 2009). Similarly, land clearing and degradation turn the valuable carbon sink into a major source of greenhouse gas emissions. For instance, as land continues to degrade, livelihood options for at least 485 million Africans also dwindle with it (www.TerrAfrica.org). Consequently, about five million hectares of forest may have been lost annually in Africa from 2005-2015, releasing nearly 2 billion tons of CO_2eq each year, or 13% of annual global emissions from forestry and agriculture combined (IPCC, 2007; Sohngen and Brown, 2008).

There is 316 billion tons of CO_2eq stored in the top soils of Africa (Sohngen and Brown, 2008). However, with two-third of Sub-Saharan Africa's cropland, rangeland, and woodland already degraded, this stored carbon is being returned to the atmosphere. What is worth mentioning here is that due to the climate change effect by 2020, 75-250 million people across Africa could face water shortage and rain-fed agricultural yields, which could drop by 50% in some African countries. Loss of biodiversity, rising sea level, more extreme and intense weather events, the retreat of forests and the spread of vector borne diseases will have additional negative impacts on lives and livelihoods in the region

(IPCC, 2007; Sohngen and Brown, 2008).

According to Ethiopia's Climate Resilient Green Economy Strategy (2011), Ethiopia's current contribution to the global increase in GHG emissions since the industrial revolution has been practically negligible. Even after years of rapid economic expansion, today's per capita emissions of less than 2 t CO_2eq are modest compared with the more than 10 ton per capita on average in the EU and more than 20 t per capita in the US and Australia. Overall, Ethiopia's total emissions of around 150 t CO_2eq represent less than 0.3% of global emissions in the year 2010. Of the 150 t CO_2eq, more than 85% of GHG emissions come from the agricultural and forestry sectors. Power, transport, industry and buildings also contributed 3% each. Carbon emissions from forest degradation are difficult to assess because of a lack of consistent data in the country. There is a growing concern on the future of such C stocks both from possible climate perturbation inducing changes in ecosystems and from direct human induced processes such as deforestation and forest degradation.

2.6. The effect of altitude and slope on forest Carbon pool

Korner (2007) noted that altitude interacts through the effects of local climate on ecological processes but not as an ecological factor affecting plant distribution by itself. The forest carbon pattern will depend on variation in the density of stems and canopy height along the elevational gradient (Baker *et al.,* 2004). While comparing sites with the same climate, the species-biomass relationship appears to be unimodal in many systems with diversity peaking at intermediate levels of biomass (Gross *et al*., 2000; Bhattarai *et al.,* 2004). According to Dossa *et al*. (2013), the highest AGB was 24.24 kg m^{-2} at 1200 m and the lowest was 9.22 kg m^{-2} at 2200 m.

Carbon stocks at a given altitude varied significantly between and within continents though carbon stocks decreased with altitude in some of the various studies conducted in tropical Asia (Kitayama and Aiba, 2002; Culmsee *et al*., 2010), South America (Alves *et al*., 2010; Girardin *et al.*, 2010; Moser *et al.,* 2011) and East Africa (Marshall *et al*., 2012) tropical montane forests. The existing information indicates that the pattern in AGB varies significantly between different tropical mountain ranges (Kitayama and Aiba, 2002; Wang *et al*., 2003; Moser *et al*., 2008).

Rahbek (1995) and Rahbek (2005) critically reviewed floristic diversity along altitudinal variation. He argued that species richness has a mid-altitude peak, which leads to the variation in AGB. He also suggested that the differences among studies may be partly due to sampling regime and the influence of the size of the area sampled. According to Korner (2007), increasing elevation may affect tree growth rates and stand structure because of reduced air and soil temperatures, often-increased rainfall, and alterations in nutrient availability and soil chemistry. In the East African montane forest, Mwakisunga and Majule (2012) estimated the carbon stock along altitudinal gradient in rungwe forest, southern highland of Tanzania. They found out that aboveground carbon content increased with altitude ranging from 9.2 t ha^{-1} at 2031 m to 561.7 t ha^{-1} at 2312 m above sea level due to less forest disturbance at high altitudes. In contrast, from the same country, Swai *et al.* (2014) reported a trend that carbon stock in AGB decreases along altitudinal gradient in Hanang forest. Accordingly, it was highest at low altitude (71.5 ± 17.0 t ha^{-1}), followed by mid altitude (49.83 ± 9.94 t ha^{-1}) and lowest at high altitude (12.98 ± 7.7 t ha^{-1}). Moreover, Mwampamba (2009) studied forest recovery and carbon sequestration under shifting cultivation in the Eastern Arc Mountains, Tanzania. This study was only to understand the effect of elevation on C_{AGB} and C_{soil} on the primary forest plots. He found that C in AGB increased at a rate of 0.25 t ha^{-1} for each 1 meter increase in elevation (N = 13, R^2 = 0.39, p = 0.0236). C in soil increased at a rate of 0.023 t ha^{-1} per 1 m increase in elevation (R^2 = 0.48, p = 0.0088). A strong and positive correlation between elevation, and C_{AGB} and C_{soil} was maintained in lowland plots but disappeared in submontane plots.

In a study conducted in Southern Appalachian spruce-fir forest, soil carbon did not show a clear trend with altitude, likewise the carbon dynamics did not show a consistent pattern with altitude (Tewksbury and Miegroet, 2007). Hamere Yohannes *et al*. (2015) reported that altitude has inverse relation with AGB, belowground biomass, deadwood carbon and total carbon density estimated in the Gedo forest, West Shewa Zone, Ethiopia. Soil organic carbon and litter biomass carbon showed no relationship with altitude in the same study. Similarly, it has been reported that biomass carbon storage decreases with altitude increases (Moser *et al.,* 2007; Sheikh *et al*., 2009). In contrast, many studies identified the trend that live biomass carbon increase with altitude (Zhu *et al*., 2010; Gairola *et al.,* 2011; Adugna Feyissa *et al*., 2013).

According to Bohra *et al.* (2014), SOC stock was found to be in decreasing pattern with increasing altitude from 193.6 to 166.4, 146.4 to 137.6 and 159.2 to 141.6 t C ha^{-1} in Oak, Pine and Sal forests, respectively. It is an indicator of higher biological activity or anthropogenic disturbance associated with top layers of these forest areas. Higher SOC was recorded in Sal forest compared to Oak. In Sal forest, high tree density leads to higher accumulation of SOC compared to conifers while it was low in wide spread Pine forest, resulting in less storage of carbon stock in turn. However, much less is known about the biomass of tropical montane rain forests and changes in tree biomass carbon pools along elevational gradients. Therefore, a deeper understanding of variation in species diversity, species composition and biomass with elevation may serve to elucidate the factors affecting each of these components.

Slope is also the terrain attributes known to influence the distribution of carbon pool across a landscape (Bolstad and Vose, 1998). Tibebu Yelemfrhat and Teshome Soromessa (2015) obtained the trend that carbon stock of the forest was increased as the degree of slope gradient decreased. Slope has significant effect on leaf litter biomass carbon (Hamere Yohannis *et al*., 2015). Moreover, Adugna Feyissa and Teshome Soromessa (2017) also found that the carbon density of different pools resulted differently in different slope gradients in the forest.

2.7. The Afromontane forests in Ethiopia

2.7.1. The Description of Afromontane forests in Ethiopia

The Afromontane forests of Ethiopia are part of Eastern Afromontane Biodiversity Hotspots, one of the 36 regions globally important for biodiversity conservation (http: //www.cepf.net/resources/hotspots/Pages/default.aspx). Most Afromontane communities are found above 2000 m above sea level, but they can occur as low as 1200m above sea level in some places (White, 1983). The Ethiopian highlands (land areas above 1500 m above sea level with the associated valleys) comprise over 50% of the Eastern Afromontane Hotspot and over 40% of the Horn of Africa Hotspot (Yalden, 1983). Moreover, the Ethiopian afromontane forests belong to the least protected eco-regions in

Africa (Kuper *et al.,* 2004; Scholes *et al*., 2006; Burgess *et al*., 2007; Tadesse Woldemariam *et al.,* 2008). This is amazing given the fact that these areas in Ethiopia have been recognized as global biodiversity conservation priority areas and centres for plant diversity (WWF and IUCN, 1994; Barthlott *et al.,* 1999) but also for other groups of organisms such as Endemic Bird Areas (ICBP, 1992). Despite its importance, the Afromontane forests of Ethiopia are being cleared and degraded at an alarming rate due to several social, economic, and political factors (Feyerea Senbeta and Manfred, 2006).

Many tree species under this group exhibit a wide range of growth forms making it difficult to classify the forest types. Rainfall received by the afromontane forest (AF) also varies from 800 mm to considerably more than 2500 mm per year (White, 1983). The forests on the Ethiopian highlands can be broadly divided into dry montane forests and moist montane forests. The dry montane forests are dominated by hard leaved evergreens, while the moist montane forests are characterized by large broad leaved and soft leaved species (Tamrat Bekele, 1993; Sebsebe Demissew *et al*., 1996). The dry montane forests are dominated by *Juniperus procera*, *Podocarpus falcatus*, and *Olea europaea* subsp.*cuspidata*. The moist montane forests consist of species like *Pouteria adolfi-friederici*, *Olea welwitschii*, O. *hochstetter*, and *Croton macrostachyus*. Mountain cane (*Arundinaria alpina*) stands are also found at humid highland elevation areas (2,500–3,400 m.) as scattered, but large and compact concentration. Nevertheless, the distinction between 'wet' and 'dry' types by earlier workers is difficult to apply as to the wide tolerance of many dominant species (Greenway, 1973). Thus, it has been recommended to further refer the Friis (1992) and Friis *et al* (2011) classification for the description of afromontane forests of the country.

2.7.2. Carbon stock Estimates of Afromontane forests in Ethiopia

There are very few but fragmented studies on the carbon storage of afromontane forests of Ethiopia with a great variation in estimating mainly AGB. The biomass stock of the country is largely concentrated in the afromontane forests of south and southwest (Figure 2). The afromontane forests in particular are important carbon stock to sequester the CO_2 gases in the atmosphere and, therefore, a major 'Carbon sink' reducing the greenhouse gases. Estimates have shown that these forests sequester nearly 27,579 Gg of CO_2 per year from the atmoshpere (Million Bekele, 2001) and at present Ethiopia is a net sinker of GHG owing to its natural forest resources. Brown (1997) reported a carbon density of 101 tons ha^{-1} for high forests in Ethiopia (containing both DAF and MAF) though under estimated. However, some case studies show even higher carbon density values of close to 200 tons ha^{-1} than the estimates based on WBISPP for high forests in Bale Mountains (Tsegaye Tadesse, 2010).

The national carbon stock presented in Table 3 was estimated based on WBISPP data. It is clear from the table that the largest store of carbon in the country is found in the woodlands (45.7%) and the shrublands (34.4%) due to the higher land area they contain. High forests contain large amount of biomass but smaller land area so that they contain aboveground carbon density smaller than woodland and shrubland (WBISPP, 2005).

Table 3. Mean aboveground carbon and total carbon stocks (t ha^{-1}) in major forest categories

Forest category	Free-Bole Biomass	BEF	AGB-C	Area(10^6ha)	Total C-stock
High forest	131.5	2.74	106.68	4.07	434.19
Woodland	21.0	6.9	42.75	29.55	1,263.13
Plantation	178.8	2.33	123.0	0.50	61.52
Lowland bamboo	26.0	6.19	47.5	1.07	50.80
Highland Bamboo	83.0	3.44	84.23	0.03	2.53
Shrubland	14.9	8.20	36.04	26.40	951.54
Total C					2,763.70

Source: WBISPP (2005), Assuming the carbon content of green wood is ~ 50% of the biomass. C is calculated based on the formula developed by Brown (1997).

Moreover, the estimated AGB over Ethiopia by using a pan-tropical biomass map based on coarse resolution MODIS data is shown in Figure 3 (FAO, 2010). This map provide a first glance of a country's AGB stocks eventhough they are very coarse and are no substitute for national forest inventory (NFI) data in a higher-tier MRV(monitoring, reporting and verification) system. Accordingly, Ethiopia's biomass stocks are largely concentrated in the montane forests of the south and southwest of the country as clearly indicated in the map i.e. in the south-central Oromia (Bale Mountains region), northwest SNNPR (Kafa-Sheka zones) and western Oromia (region surrounding Yayu Biosphere Reserve). As indicated in the map, the lower AGB stocks are found in Gambella, Beni-Shangul Gumuz, Amhara, and Tigray regions, and significantly lower AGB stocks in the drier regions of Afar and Somali.

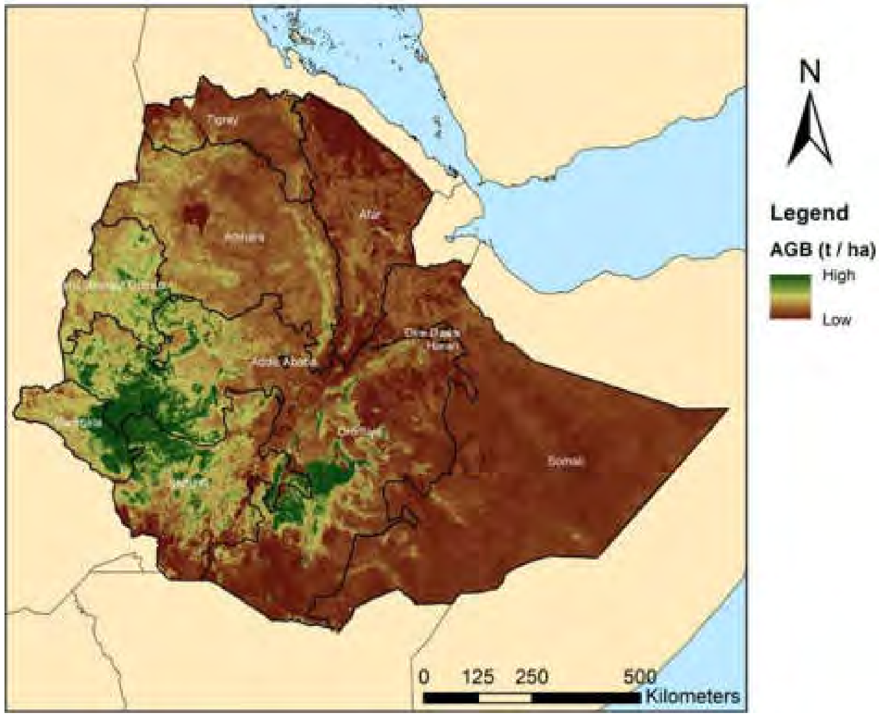


Figure 3.Coarse-Resolution Above-ground carbon stocks in Ethiopia (green regions indicate areas of high biomass and dark brown regions indicate areas of low biomass. (Source: FAO, 2010).

There are also some fragmented carbon stock estimates done in different forest types of the country using different methods particularly for the above biomass estimation. Muluken Nega *et al.* (2015) estimated the carbon stock of Adaba-Dodola community forest in Oromia regional state, Ethiopia. They found that the mean total carbon density of 507.29 t ha^{-1} with the biomass carbon density of 319.43 t ha^{-1}. The soil organic carbon of up to 30 cm depth layer is 186.40 t ha^{-1}, undergrowth shrubs (0.4 t ha^{-1}), litter, herb and grasses carbon density accounts for 1.06 t ha^{-1}. The allometric equation they used for estimating above ground biomass carbon is Chave *et al.* (2005); $AGB = _{Exp} (-2.187+0.916 \ln (D^2H\rho)$ for trees greater than 5 cm diameter. Tibebu Yelemfrhat *et al.* (2014) also estimated the mean total carbon stock of about 568.314 t ha^{-1} in lowland area of Siemen Mountains National Park. The mean above and belowground biomass carbon stocks were

270.89 ± 154.50 and 54.18 ± 30.81 t ha^{-1} respectively. The mean carbon in dead wood, litter, herb and grasses and soil carbon were 0.7258±1.0479, 0.019±0.008 and 242.51±46.42 t ha^{-1}, respectively. For their study, Tibebu Yalemfrhat *et al.* (2014) used the Brown *et al*. (1989) allometric equation; AGB= 34.4703 - 8.0671 (DBH) + 0.6589 (DBH^2) to estimate the tree AGB with a DBH ≥5 cm.

Furthermore, Abel Girma *et al.* (2014) estimated the carbon stocks in the woody plants of Mount Zequalla monastery forest located 74 km east of Addis Ababa, Ethiopia. The forest is part of the dry evergreen afromontane forest in the western edge of the central Rift valley. Accordingly, the mean total carbon stock of mount Zequalla forest is 348.8 t ha^{-1}. The mean carbon stock per hectare in different pools are 237.2, 47.6, 6.5 and 57.6 ton for AGB, below ground biomass, litter biomass and soil. The allometric equation that Abel Girma *et al*. (2014) used for the estimation of forest aboveground carbon was Brown *et al., (*1989) equation; AGB = 34.4703-8.0671 (DBH) + 0.6589 (DBH^2). Moreover, Tullu Tolla (2011) estimated the mean total carbon stock of the selected church forests in Addis Ababa, Ethiopia, by using the same biomass equation as above. He found out that the mean aboveground and belowground carbon stock of 129.86 and 25.97 t ha^{-1} and the soil organic carbon of 135.94 t ha^{-1}. Moreover, Getachew Tesfaye (2007) reported that the AGB of dry tropical afromontane forest in Ethiopia in the range of 403.08 - 754.32 t ha^{-1} by using Brown *et al.,* (1989) allometric equation. Hamere Yohannes *et al*. (2015) estimated the total mean carbon stock of 523.64 ± 29 ton ha^{-1} with AGB (281 ± 23.34 t C ha^{-1}) and belowground biomass 56.1 ± 4.66 t C ha^{-1}), litter biomass (0.41 ± 0.008 t C ha^{-1}), deadwood biomass (2.37 ± 1.33 t C ha^{-1}) and soil organic carbon (183.69 ± 6.17 t C ha^{-1}) in Gedo forest, Ethiopia.

2.8. Litterfall, decomposition and nutrient dynamics in the forest ecosystem

Litter dynamics, including production, decomposition, and accumulation are vital links between plant and soil nutrient storage and cycling (Xu and Hirata, 2005) especially in humid tropical forests where nutrient availability is intimately tied to litter inputs and decomposition (Diaz, 2004). According to Bernhard-Reversat and Loumeto (2002) and Montagnini and Jordan (2002) litterfall provides three main functions in the forest ecosystem. These are energy input for soil microflora and fauna, nutrient input for plant nutrition, and material input for soil organic matter building up. The first two functions are completed through decomposition and mineralization, and the third one through decomposition and humification (figure 4). Those functions are related to the main soil processes, such as biological activity, nutrient cycling and soil structure.

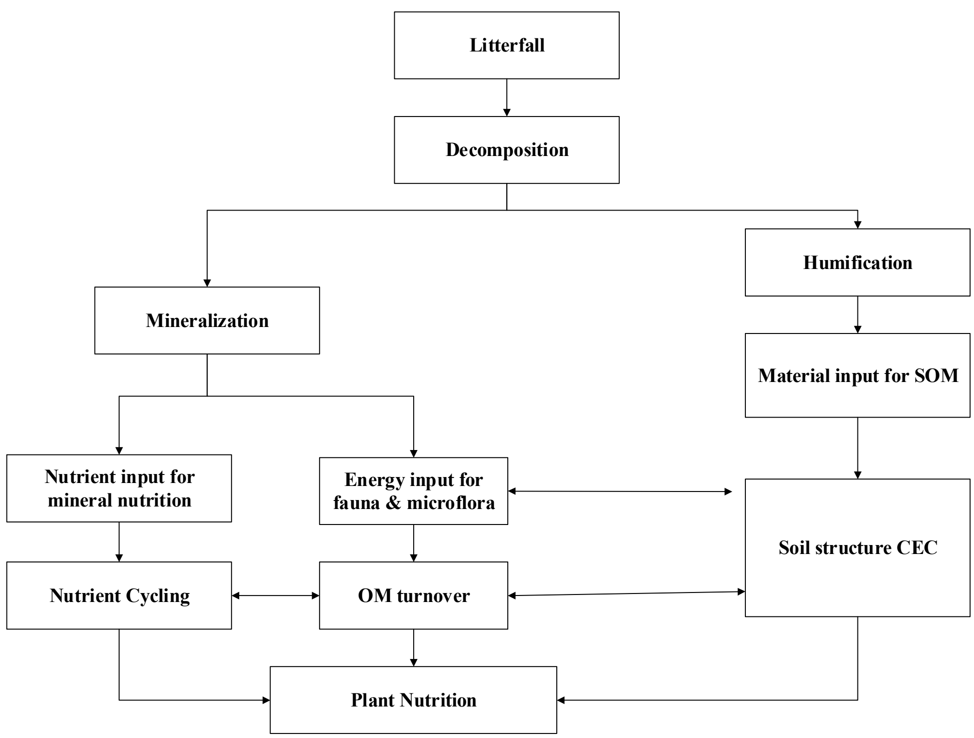


Figure 4. Schematic representation of litterfall functions in the Ecosystem (Bernhard-Reversat and Loumeto, 2002)

These ecosystem processes are of key importance in tropical forests, where the vegetation is generally sustained by soils with low fertility. Forest productivity depends on efficient nutrient cycling mechanisms that ensure rapid turnover of litter nutrients (Lavelle *et al*., 1993; Montagnini and Jordan, 2002). In tropical montane forests, nutrient cycling mechanisms are especially relevant, since their leaf litter is composed of lower nutrient concentrations than those from lowland forests (Bruijnzeel and Veneklaas, 1998) and the lower temperature reduces the decomposition rate and nutrient release (Röderstein *et al.,* 2005), making their soils extremely poor in nutrients (Bruijnzeel and Veneklaas, 1998).

2. 8.1. Litterfall production and Climatic factors

Forest litter is an important stage in habitat conservation providing nutrient return and organic matter replenishment (Ashton, 1975). Distribution of litter components, amounts of litterfall, concentration of nutrients in litter components, and amount of nutrients returned in litters are all very important aspects of nutrient recycling in forest ecosystems (Berg, 2000). It acts as an input-output system of nutrients. The rates at which forest litterfalls and subsequently, decays contribute to the regulation of nutrient cycling as well as to soil fertility and primary productivity in forest ecosystems (Bubb *et al*., 1998; Jamaludheen and Kumar, 1999; Berg, 2000; Rangers *et al.,* 2003). Agren and Bosatta (1996) described litter as 'the bridge between plant and soil.' It also represents an energy source of heterotrophic organisms, a nutrient reservoir for cycling and a factor influencing hydrology (Christensen, 1975; Kimmins, 1987). It is critical to understand the nutrient dynamics of litter in the forest ecosystems because it is the imperative link between the autotrophs and heterotrophs, reduces bulk density, increase water holding and cation - exchange capacity of soil and serves as reserve store of plant nutrients

(Hoyle, 1973).

Leaves account for about 70 to 80 % of litterfall (Conner, 1994) and non-woody litter, including flowers, fruits and seeds, typically account for less than 10 % of litterfall total (Megonigal and Day, 1988). Litter production follows a seasonal pattern with lowest values during wet seasons and highest values in dry seasons (Nsabimana, 2009). According to Nsabimana (2009), the litter production rate was comparable to the values observed in other tropical forests, while the biomass increment of the same study site was larger than the values from other tropical forests. According to Sousa Neto *et al*. (2011), the litterfall also decreased with altitude from 8.40 to 5.50-ton ha^{-1}. The same study also showed that the carbon and nitrogen stocks in this zone also decreased with altitude. Litterfall patterns are often linked to seasonality of rainfall, even though correlations show large variations among different studies. Most tropical forests have seasonal variation in litter fall and in decomposition resulting in large variation in the amount of litter on the soil, for instance, in the Sempervirent Banco forest of Ivory Coast, the amount of litter ranges from 0.7 t/ha during the wet season to 3.5 t ha^{-1} during the dry season in the same site.

Rainfall may have a two-fold influence on litter production and nutrient input. On the one hand water limitation in dry periods provokes increased shedding of senescent leaves, on the other hand heavy rains and storms force the shedding of non-senescent leaves, providing a nutrient pulse through higher qualities of leaf litter (Cuevas and Lugo, 1998). The nutrient concentrations in the leaf litter correlate with precipitation (Wood *et al.,* 2005). Both positive and negative relations are reported, which probably can be related to shifts in nutrient availability, uptake and loss (Wood *et al*., 2005). Hence, temporal variations of litterfall and litter quality are rather linked to regional climate, than to local factors like nutrient status of soils (Wood *et al.* 2006). Qiu *et al.* (1998) suggested that rainfall, temperature and light might play an important role in leaf fall and flushing among dominant canopy species in the forest. In the same study, seasonal litter production is highest in dry months, which is consistent with results from tropical forests in India (Arunachlam *et al.,* 1998; Sundarapandian and Swamy, 1999).

On the other hand, disturbances influence the composition and structure of tropical forests (Lodge *et al*., 1991; Dezzeo *et al*., 2004) and therefore change the nutrient cycling patterns, changing the equilibrium between production of biomass, accumulation of organic matter and decomposition and absorption of nutrients (Barnes *et al*., 1998). Longer litter turnover times, recorded near edges, lead to biomass and nutrient accumulation on the forest floor and are related to the effect of increased litterfall, which indirectly reduces decomposition rates by alteration of litter quality and decomposer community characteristics (Dirham, 1998; Johansson, 1995).

2.8.2. Amounts of Litterfall in different forests

The amount of litterfall varies widely among forests of the world. Oziegbe *et al*. (2011) studied litterfall in secondary lowland forests of Nigeria and found a total litterfall of 9.3 t ha ^{1} per year for the period of November 2001 to October 2002. They considered the value to be well within the range of 7.0–14.1 t ha ^{1}y^{-1} reported for other West African tropical lowland rainforests (Muoghalu *et al.,* 1993; Songwe *et al.,* 1995; Odiwe and Muoghalu, 2003). The value also fell within 5.8–12.0 t ha ^{1} year^{-1} reported by Anderson and Swift (1983) and Proctor (1987) for lowland tropical rain forests.

Moreover, Chave *et al.* (2010) showed that across old-growth tropical rainforests, litterfall averages 8.61 t ha ^{1} per year. Secondary forests have a lower annual litterfall than old-growth tropical forests with a mean of 8.01 t ha ^{1} per year. The outlying secondary forest at the edge of the Mata de Piedade site, the Atlantic rain forest of Brazil, produced 14.74 t ha ^{1} per year litter. On the other hand, montane forests and low forests had lower mean annual litterfall (7.06 t ha ^{1} per year and 3.01 t ha ^{1} per year, respectively). Gonzalez (2012) reported a wide range, from 1.99 to 9.16 t ha ^{1} per year, of total litterfall in the floodplain forests of a large Mediterranean river. Liu *et al.* (2005) observed that litterfall in evergreen broadleaf forests in the subtropical region of China ranged from 7.1 to 11.0 t ha ^{1} per year with a mean value of 9.0 t ha ^{1} per year; but the average annual litterfall in the pine forest was only 2.67 t ha ^{1} per year. Annual rates of litter production varied between 8 and 10.5 t ha ^{1} per year in a tropical semi-deciduous forest of the southern Amazon Basin (Sanches *et al*., 2008). Litterfall in some evergreen and deciduous forests in Chile ranged from 3.5 to 5.8 t ha ^{1}yr^{-1} and was temporarily lower in the managed than in the unmanaged deciduous stand (Staelens *et al*., 2011). The amount of litterfall estimated in some of the tropical forests is given in Table 4.

Table 4. Amounts of Litterfall (t ha^{-1}) in Tropical Forests

Forests	**Location**	**Litterfall**	**References**
Natural tropical forests of	Ghana	10.7	Vitousek and Sanford
Variable species diversity	Zaire	12.4	(1986)
Natural Forest	Indonesia	13.67	Triadiati *et al.* (2011)
Broad-leaved forest	China	3.56	Zhou *et al*. (2007)
Mixed pine and Broad-leaved forest	China	8.61	
Evergreen broad-leaved forest	China	8.16	
broad-leaved plantation species and natural forest	Ethiopia	9.7 to 12.6	Ambachew Demessie *et al*. (2012)
Coniferous species	Ethiopia	4.9 to 6.6	Ambachew Demessie *et al*. (2012)

2.8.3. Leaf litter decomposition and Nutrient Release Pattern

2.8.3.1. Leaf litter decomposition

Berg and McClaugherty (2008) referred decomposition as any changes in biochemistry, appearance and weight. Moreover, Wood (1974) defined decomposition in plants as weight losses due to a number of factors, including the removal and/or consumption of tissues by leaf-feeding invertebrates, leaching, and biochemical degradation by microorganisms during passage through the guts of invertebrates. Furthermore, Coûteaux *et al.,* (1995) defined it in such a way that it involves the mineralization and humification of lignin and cellulose by microorganisms simultaneously with the leaching of soluble compounds. The process allows for carbon and other minerals to be progressively immobilized or mineralized into the soil and essential for the regeneration of organically bound nutrients (Berg and McClaugherty, 2008).

Decomposition of plant materials occurs through initial fragmentation by soil macrofauna (earthworm, millipedes, termites, etc) and then by microbial activity via enzyme production for further transformations (Stevenson, 1986; Coûteaux *et al.,* 1995; Tian *et al.,* 1995). Each stage involves the partial conversion of C to CO_2 and the synthesis of microbial tissue (Stevenson, 1986). In addition, the constituents of organic residues have been established to decompose at different rates. For example, simple sugars, amino acids, most proteins and cellulose decompose rapidly mainly by the action of bacteria whereas lignins and some microbial melanins decay slowly mostly through the action of actinomycetes and fungi (Stevenson, 1986; Berg and McClaugherty, 2008). Decomposition is an important indicator of ecosystem stability involving the interaction of vegetation, soil nutrient availability, micro-and macro-fauna, and microbial populations (Knoepp *et al.,* 2000). If decomposition is impaired, the ecosystem is functionally impaired (Schaeffer *et al*., 1988). More importantly, forest litter decomposition is a major pathway for providing organic and inorganic elements for the nutrient cycling processes (Mudrick *et al*., 1994).

Ecosystem functional processes (e.g., productivity and decomposition) are generally known as important indicators of ecosystem stability or homeostasis (van Voris *et al*., 1980; Rapport *et al.,* 1985). The growth and productivity of forest ecosystems mainly depend on the amount, the nature and the rate of decomposition of forest litter. Moreover, litter decomposition plays an important role in carbon cycling in terrestrial ecosystems (Aerts *et al*., 2006; Field *et al*., 1998; Shiels, 2006; Prescott, 2005). It also occurs largely on the forest floor, where bacteria and fungi apparently are among the earliest colonizers of fresh litter (Berg and McClaugherty, 2008).

The decomposition process, because of the activities of soil organisms, releases nutrients contained in the litter into the soil solution. The released nutrients are then taken up by plants and microorganisms to satisfy their growth requirements. Consequently, the rate of litter decomposition is a strong determinant of the availability of nutrients in the soil for plant growth. As described by Liski *et al.* (2003), to predict the amount of carbon released through the litter decomposition, this variability should be accountable and well documented. It is not entirely clear how the litter decomposition rate distributes at the large spatial scale, and which factors are critical in controlling the litter decomposition globally (Zhang *et al*., 2008). The major factors that regulate litter decomposition rate include; mean annual temperature (MAT), Mean annual precipitation (MAP) and annual actual evapo-transpiration (AET) (Aerts, 1997; Berg *et al.,* 2000; O'Neill *et al.,* 2003). Moreover, litter quality such as nitrogen content, carbon to nitrogen ratio, lignin content, lignin to nitrogen ratio (Berg *et al.,* 2000)**, v**egetation and litter type also determine litter decomposition rate (Prescott *et al.,* 2000). In addition, geographical variables such as latitude and altitude (Aerts, 1997) and aspect, slope position (Mudrick *et al*., 1994), acidity (Berger and Glatzel, 1994), soil fertility (Klemmedson, 1987), soil temperature, aeration, and moisture availability (Cortez, 1998; Coulis *et al*., 2013) are among the factors contributing to litter decomposition.

Climatic seasonality characterized by alternating wet and dry periods also plays a vital role in regulating the rate of decomposition (Tripathi and Singh, 1992). In general, climate markedly modifies the nature and rapidity of decomposition of plant remains on soil surface and thus exerts an important influence upon the nature and abundance of the organic matter.

Soil moisture and temperature are amongst the most crucial variables (Singh, 1969; Brinson, 1977) because they affect both the development of plant cover and the activities of microorganisms, which are highly critical factors in soil formation. Kononova (1975) concluded that the highest intensity of organic matter decomposition was observed under conditions of moderate temperature (about 30°C) and soil moisture content of about 60-80 percent of its maximum water-holding capacity. Simultaneous increase or decrease of temperature and moisture beyond the optimal levels brought about a decline in the rate of organic matter decomposition. As temperature increases, soil moisture assumes an increasingly important role for maintaining high rates of microbial activity (Peterjohn *et al*., 1994). As a result, rates of fresh litter decomposition increase with both increasing temperature and precipitation (Meentemeyer, 1978). Rate of litter decomposition are often estimated using a regression approach. There are different approaches to using the most common equations, one of which is the first order negative exponential decay equation proposed by Jenny *et al.* (1949), and elaborated by Olson (1963)(see section 3.7.4). The decomposition rate constant, k, can be calculated by fitting the exponential decay model to a scatter plot of t vs X_t/X_o (Harmon *et al*., 1999).

2.8.3.2. Nutrient Release pattern in Decomposing litter

Nutrient release pattern from decomposing leaves in tropical forests involves three major phases (Berg and McClaugherty, 2008; Swift *et al*., 1979). The phases as indicated in Figure 5 include: (1) an initial phase where leaching and nutrient release dominate; (2) a net immobilization phase where nutrients are imported into the residues by microbes; and (3) a net release phase where the nutrient mass decreases. It has been reported that not all of these stages are seen in practical experiments. For instance, the accumulation phase could be missing, particularly in litter with high N concentrations (Berg and Laskowski, 2006).

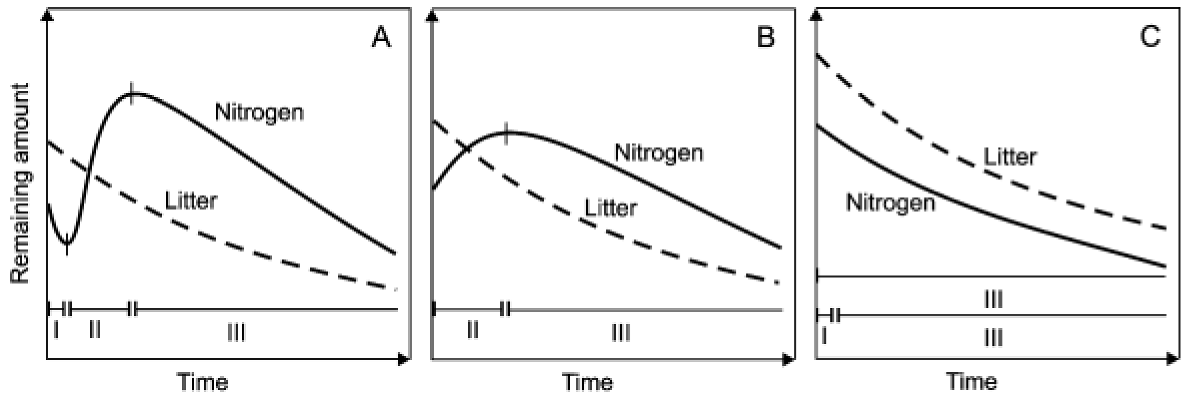


Figure 5. Three separate phases may be distinguished for the change in amount of litter N over time. A) A leaching phase (I) is followed by an accumulation (II) and release phase (III). B) An accumulation (phase II) is followed by a release (Phase III). C) Only a release is seen (Phase III or Phase I + Phase III) (Source: Berg and Laskowski, 2006)

More specifically, N and P dynamics can vary from this model, being characterized by early immobilization (net accumulation) and followed by net release (Vitousek & Sanford, 1986). The initial N and P contents in leaf litter are good indicators of the decomposition rate (Wang *et al*., 2008). Moreover, P availability in many tropical sites is very low compared to N. Thus, the relationships between decay rates and litter P concentrations have been reported for sites where P availability is low due to either edaphic factors or N deposition (Vitousek *et al.,* 1994). This is mainly attributed to a low P supply through weathering of parent material and tight absorption of orthophosphate in the widespread oxisols and ultisols. Consequently, it might be expected that variability in P supply is an important control of litter decomposability in the tropics (Vitousek and Sanford, 1986). However, many authors (Lisanework Nigatu and Michelsen, 1994; Kwabiah *et al.,* 2001; Tesfaye Teklay, 2007) stated that significant portion of P in leaves are inorganic forms and leaching might explain a major part of the P release from the leaf residues during decomposition.

Carbon and nitrogen dynamics are related to the relative availability of C and N to the microbial population. If C: N is less than 20, N will be released by enhancing the decomposition of materials. When C: N is higher than 20, there will be N immobilization until decomposition lowers the C: N ratio, though many authors mentioned different critical values of C: N for different climate zones (Swift *et al.* 1979; Bosatta and Staaf, 1982; Duchaufour, 1984; Torreta and Takeda, 1999; Heal *et al*., 1997). Moreover, indices that incorporate both C chemistry and nutrient content, such as C: N or lignin:N or C:P ratios, are often negatively correlated with early decay rates in various studies (Moore *et al.,* 2006; Thomas *et al*., 2014). In addition to the above facts, rapid mass loss was strongly correlated with the soluble labile C content during the early decomposition phase (Harmon *et al*., 1990; Berg and McClaugherty, 2008). This is due to the fact that the amount of carbon, and the chemical composition of the carbon containing molecules (e.g. starch, cellulose, lignin), are also strong determinants of decomposition rate. For example, starch is easier to break down and provides a greater energy yield than complex lignin molecules. Litter containing a high amount of starch is broken down faster than litter containing high amounts of lignin. However, no single parameter explains the C-mineralization process in litter decomposition in the soil which suggest that the combination of environmental and biological factors are involved in the process (Berg and Laskowski, 2006; Berg and McClaugherty, 2008). Potassium (K), calcium (Ca), and magnesium (Mg) are also essential macronutrients for energy metabolism, photosynthesis, and membrane transport of plants (Slovic, 1997) and rarely limit microbial processes and rapidly lost from the decomposing litter (Anderson *et al*., 1983). These nutrients are available to forest trees from the litterfall (Likens and Bormann, 1995).

Many authors have described the dynamics of K, Ca, and Mg during litter decomposition: K leached out quickly from decomposing litters, Ca decreased as carbon loss during litter decomposition, and Mg often showed an intermediate release pattern (Berg and Cortina 1995; Adams and Angradi, 1996; Hasegawa and Takeda, 1997; Osono and Takeda, 2004). K is not a structural material and it exists mainly in solution in plant cells. Mobile K thus leached out quickly from decomposing litters. The late phase of K dynamics was, on the other hand, characterized by seasonal changes despite the initial litter quality (Osono and Takeda, 2004) and K concentrations decreased rapidly throughout the two years study on three tree species and the net release of K by the end of two years was 91% of initial inputs (Blair, 1988).

CHAPTER THREE

3. Material and Methods

3.1. Materials

The materials used for the data collection constitute measuring tape for measuring DBH, rope for making a quadrat, Global-positioning system (GPS), slide calliper for measuring DBH of small stems, balance for weighing litter and herb layer biomass. Moreover, plastic bags for collecting harvested herbs/ litter biomass for dry weight estimation, nylon cloth with mesh size of 2 mm for litter trap and litter bag study, plastic bags for soil sampling, square wooden frame for sampling herb layer biomass and litter trap construction, pressing materials, compass, clinometer and data recording sheet.

3.2. Description of the Study Area

The study area include Biteyu forest in Gurage Zone (SNNPR), from the Western Escarpment of the Central Rift Valley and Boter-Becho forest in Jimma Zone, Oromia Regional state, from the Gibe watershed. Boter-Becho forest makes part of watershed for the main Gibe River where cascading hydropower dams are being constructed. In addition, Boter-Becho is also one of the forest priority areas of the country.

Biteyu dry evergreen montane forest of Gurage Mountain chains serve as a watershed for the *Wegeram* river that eventually forms the *Meki* river. This forest has been subjected to deforestation and forest degradation leading to the loss of forest biodiversity. Moreover, the high population density that largely depend on subsistence agriculture for their livelihoods exist and no published works have been documented for the Biteyu forest.

3.2.1. The Biteyu Forest

3.2.1.1. Location

The Gurage Mountain Chain harbours patches of intact vegetation cover in a mosaic of various land use and land cover matrices. It is located at the edge of the central rift valley in Ethiopia, which is one of the environmentally susceptible areas. Biteyu Forest is one of the remnant forest patches in the Gurage mountain chain (Figure 6). The total area of the Biteyu forest is estimated to be 547 ha (SUNARMA, 2015). SUNARMA is a local NGO working on the area of Natural Resource Management. This forest patch is a remnant Dry Evergreen Afromontane Forest that was once widespread on the Central plateau of Shewa (Mekonnen Biru, 2003). This area lies between 8^0 13-$8^0$14' N and 38^0 19' - 38^0 20' at the edge of the western escarpment of the Central Rift Valley which is about 20 km NW of Butajira town of the Gurage zone. Biteyu forest is approximately about 130 km south of Addis Ababa. The topography of the forest is very rugged with an elevation ranging from 2200 m to 3400 m a.s.l and the slope of the forest ranges from 20 to 55 degrees. This forest covers mountain chains such as *Kechemochi*, *Gerodeaebi* and *Beiri.* Consequently, Biteyu forest drains more than forty seasonal small streams, and two perennial rivers, which merge together to form *Wegeram* River that eventually forms the *Meki* River in the western escarpment of the Central Rift Valley. Logan (1946) described that Central Ethiopia contains the majority of the soils made of volcanic origin. Two principal soil types originating from the disintegration of volcanic substrates mixed with sand and limestone, black and red soils described. The black soils appear on the plateau and in the bottom of valleys whereas the red soils on rugged and sloppy mountains (von Breitenbach, 1961). Generally, the type of the soil is luvisol (FAO, 1998).

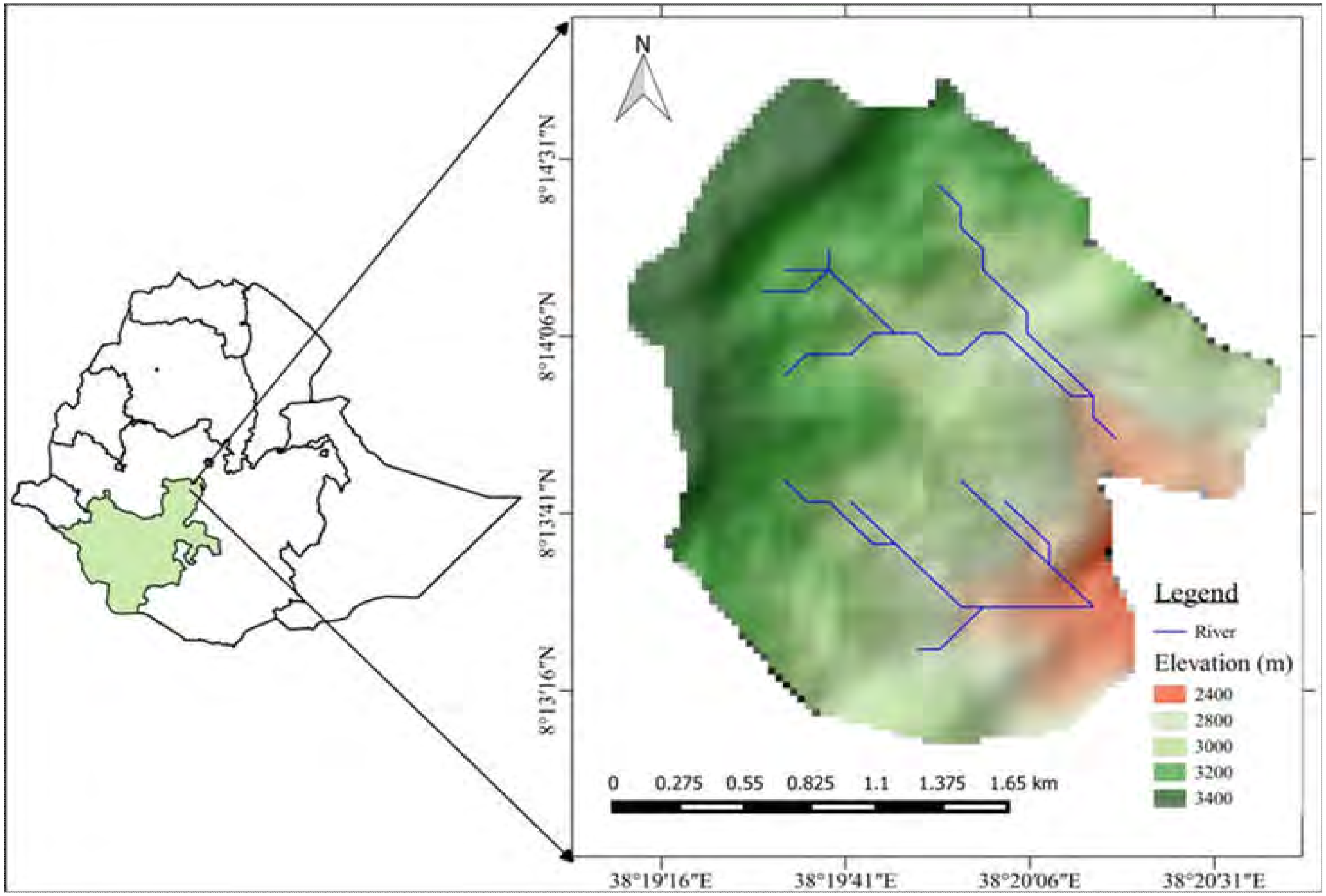


Figure 6. Location map of Biteyu forest in SNNPR of Ethiopia

3.2.1.2. Climate

In order to construct the climate diagram of Biteyu forest (Figure 7), 21 (1994-2015) years climate data of Butajira station was obtained from the Ethiopian Metrological Service Agency (NMSA, 2016). The mean monthly minimum temperature of the coldest month is 9.9 ^{0}C and mean monthly maximum temperature of the warmest month is 27.1 ^{0}C. Moreover, the mean annual temperature and mean annual rainfall of this area is 18.8 ^{0}C and 1128 mm respectively. The area possess unimodal rainfall pattern by which it gets small rainfall in March and April and the highest rainfall in between May and September.

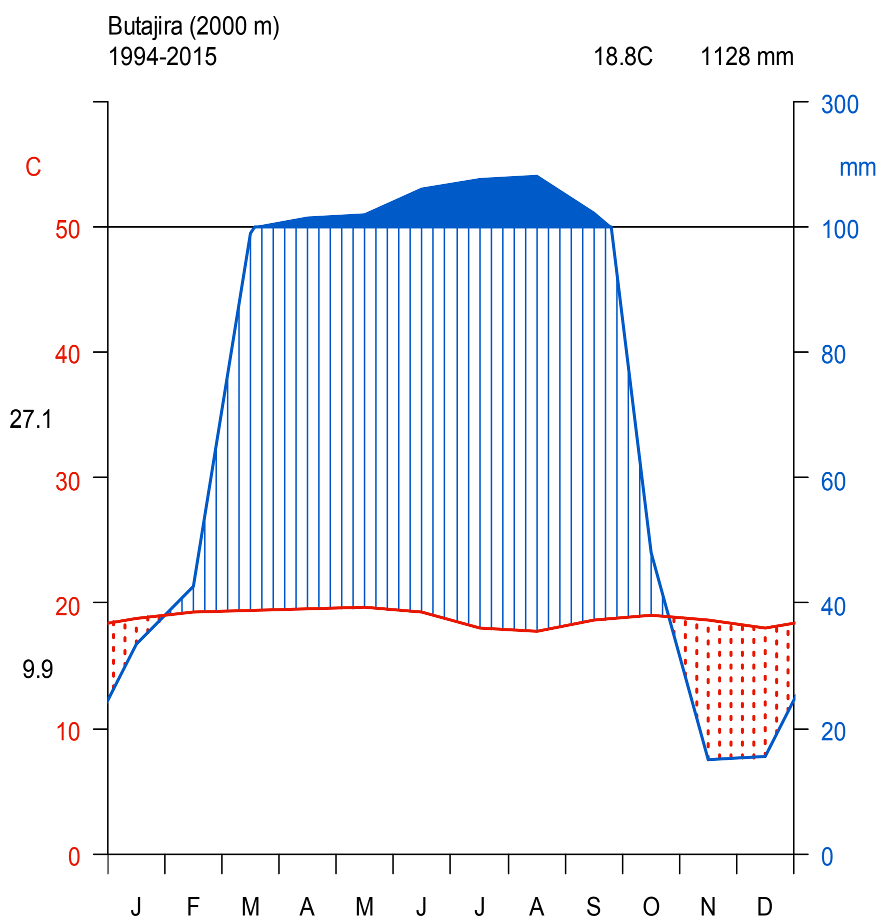


Figure 7.Climadiagram of the nearest Metreological station (Butajira town)

3.2.1.3. Vegetation

Biteyu forest is classified as Dry Afromontane forest; the vegetation of the forest is characterized by indigenous species such as *Juniperus procera, Olea europaea* subsp.*cuspidata, Podocarpus falcatus* and associated woody species including *Maytenus addat, Myrsine melanophloeos*, *Olinia rochetiana, Maesa lanceolata* and others. Logging for construction purpose, fuelwood collection and agricultural expansion are the major causes for the dwindling of the vegetation. Severe trampling and heavy livestock grazing pressure were observed as the major causes of forest degradation and deforestation (Figure 8).

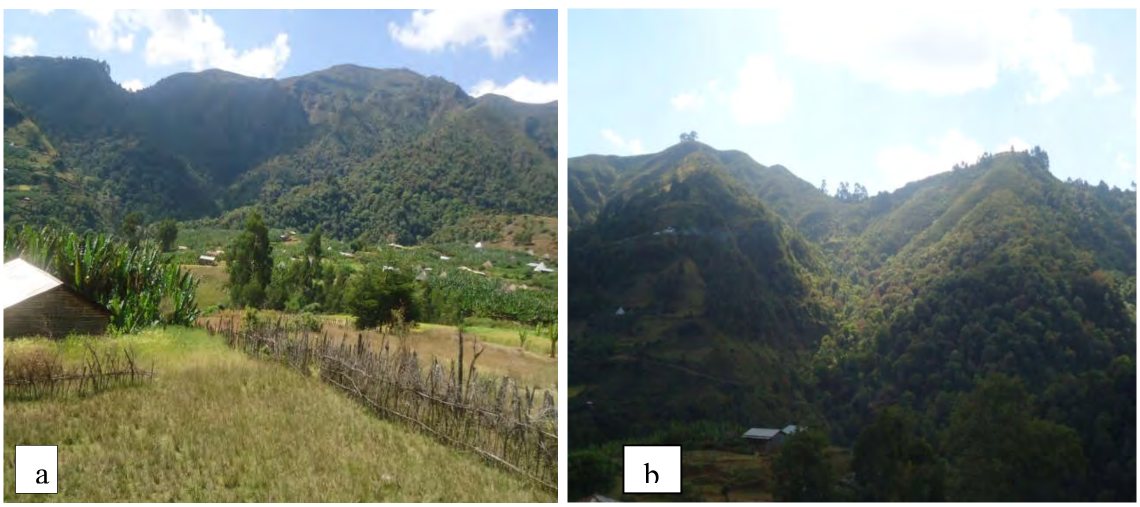


Figure 8.a/The view of Biteyu Forest and b/Agricultural expansion to the forest

3.2.2. Boter-Becho Forest

3.2.2.1. Location

Boter-Becho forest is one of the national forest priority areas of Ethipia. It lies in between *Tiro-Afeta* and *Chora-Boter* districts of Jimma zone, Oromia region (Figure 9). Approximately, the forest is located between $08^0 10'$- $08^0 30'$N and 037^0 8' - 037^0 21' E coordinates. The forest is located at about 223 km southwest of Addis Ababa, Capital city of Ethiopia. The estimated total area of the Boter-Becho Forest is about 31,600 ha. The forest is dominated by Acacia woodland in the lower altitude, high montane forest on slopes and in the valleys up to around 2500 m. However, the Moist Evergreen Forest (MAF) part was considered for this study. The area lies along a volcanic mountain ridge, running almost north to south, and rising to a series of small peaks, the highest of which is 3100 m above sea level. The slope of the forest ranges from 2 to 45 degrees. The soil type is eutric nitosol, which is the characteristic soil type in Ethiopian highlands. The Soil beneath the forest is reddish-brown and well drained, and typical of the forest soil derived from Ethiopian plateaus. The rock dominating the southern part of the ridge is volcanic and the southeast principally comprises Olivine and basaltic, and Kaolinite, halloysite and iron oxides dominate their clay mineralogy. The parent rocks in the area also include ignimbrite and agglomerate (FAO, 1998).

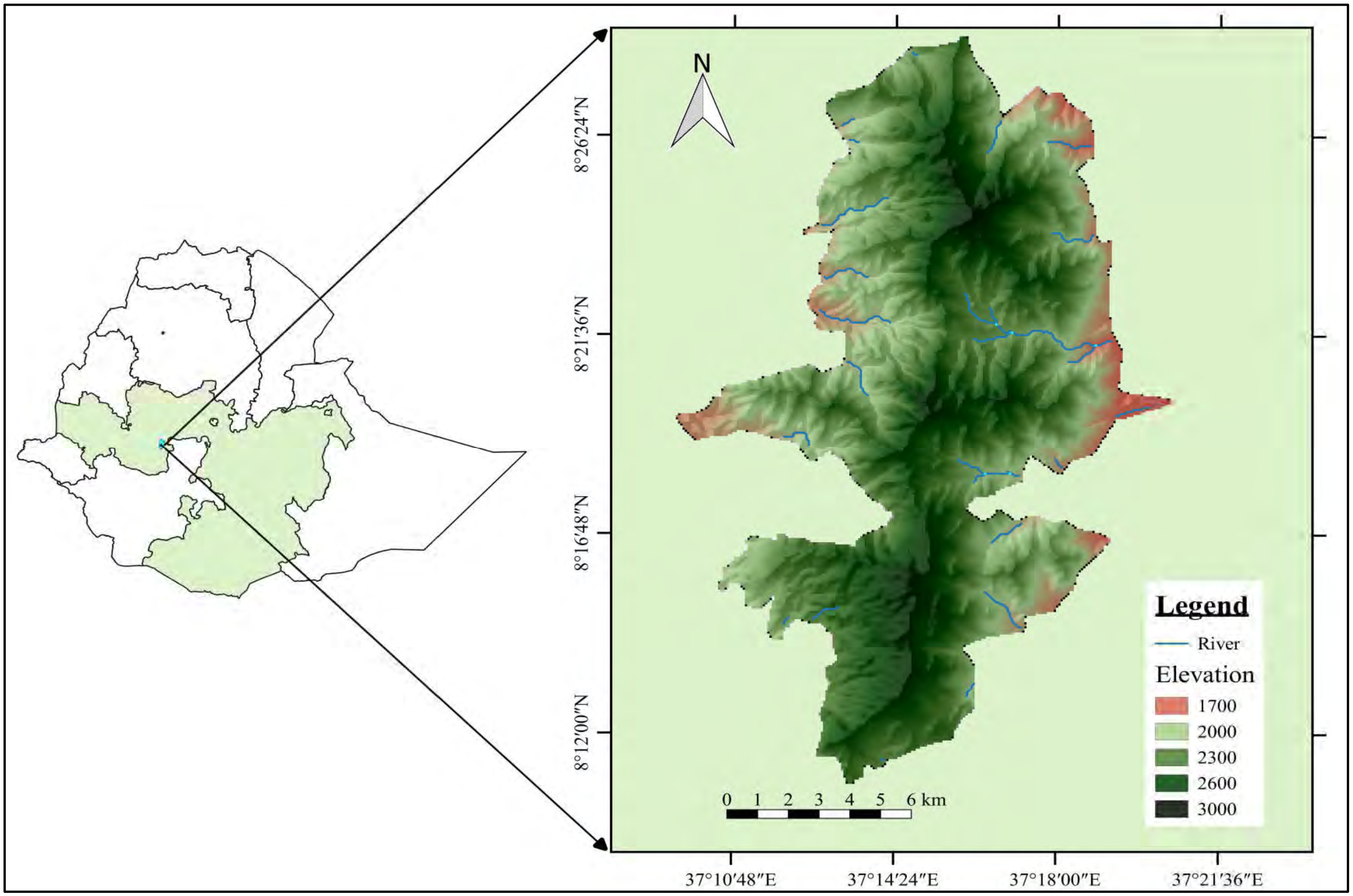


Figure 9. Location map of Boter-Becho forest in Oromia Regional State of Ethiopia

3.2.2.2. Vegetation

The forest (Figure 10) is composed of a mixture of *Juniperus procera*, *Hagenia abyssinica*, *Olinia rochetiana*, *Olea capensis* subsp. *Macrocarpa, Syzygium guineense* Subsp. *afromontanum* and other small trees that grade into an open *Erica arborea* zone around 3100 (the highest peak). Moreover, it contains a number of medium sized trees and large shrubs, a mixture of *Podocarpus falcatus* and broad-leaved species as emergent trees in the canopy including *Pouteria adolfi-friederici*. There are some patches of *Arundinaria alpina* in wet, sheltered valleys.

Several streams forming rivers in mountains of *Botor-Becho* and are drained by the Gilgel Gibe to the west, which forms a wide valley to the lower parts of the forest, and the main Gibe River to the north and east

Figure 10.The view of Botor-Becho forest

3.2.2.3. Climate

The 15 years climate data (2001-2015) for the Boter-Becho station was obtained from Ethiopian Metreological Service Agency (NMSA, 2016) and used to develop the climadiagram of the Boter-Becho forest. The climate in Boter-Becho is warm and temperate. Boter-Becho has mean annual rainfall of 1434 mm and mean annual temperature of 14.6 ^{0}C (Figure 11). Boter-Becho has unimodal rainfall pattern with long rainy seasons from March to September of which the highest rainfall is recorded in June and August. The area posseses short dry season from October to February where it gets small rainfall. The mean monthly minimum temperature of the coldest month is 1^{0}C and mean monthly maximum temperature of the warmest month is 27.0 ^{0}C. The mean annual temperature is 14.9 and 14.2 ^{0}C in wet and dry season, respectively.

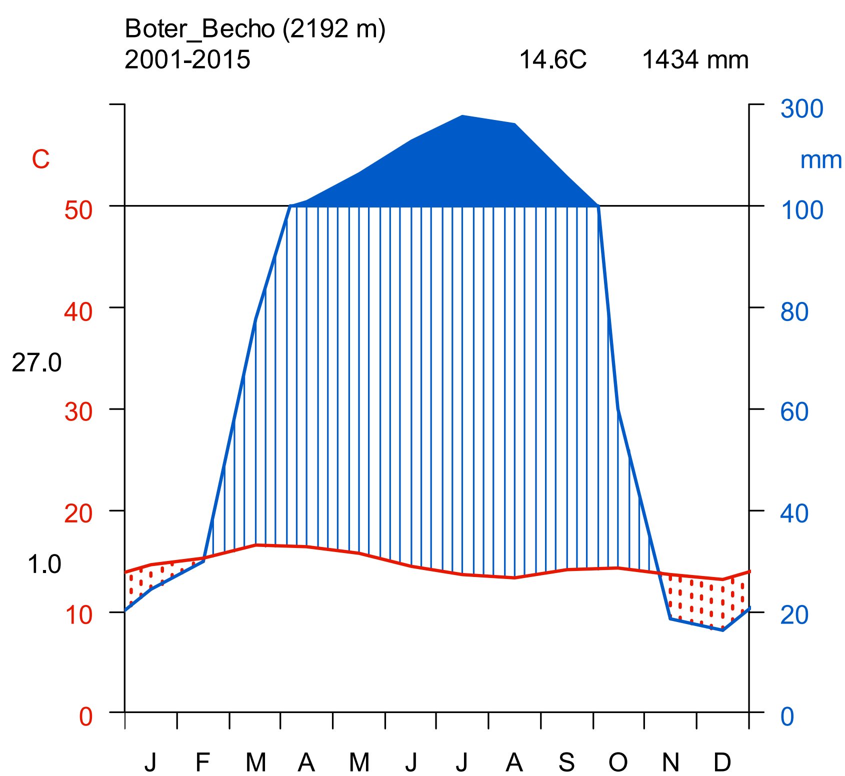


Figure 11.Climadiagram of Botor-Becho Forest

3.3. METHODS

3.3.1. Sampling Technique

Systematic sampling technique was used for vegetation data and environmental data collection following Muller-Dombois and Ellenberg (1974) and Kent and Coker (1992). A total of 22 line transects (6 in Biteyu and 16 in Boter-Becho forest) were laid along elevational gradients. The transects were 500 m and 300 m apart in Boter-Becho and Biteyu forests, respectively. The sampling plots were 300 m and 200 m apart in Boter-Becho and Biteyu forests, respectively. Each of the line transects had different number of sampling plots depending on the length of the transect.

3.3.2. Plot shape and Sample size

A square or a rectangle is the most commonly adopted shape for sampling plots for vegetation types including forests, plantations, agroforestry, shelterbelt, grassland and cropland (Pearson *et al.,* 2005). Square plots were used in this study. Sampling should be carried out with statistical rigour to come up with good results. Accordingly, the first step is identifying the number of plots required to reach the desired precision in the results. Therefore, preliminary data were necessary in order to calculate the required number of plots for the desired level of precision. Preliminary data from 6 and 10 randomly selected plots were collected from Biteyu forest and Boter-Becho forests, respectively, following Pearson *et al*. (2005). Total carbon stock from four carbon pools (AGB, BGB, soil and litter) was determined from each plot and the mean carbon stock fom all the plots in each forest was used for sample size determination. Since both forests are seemed to be homogenous, each forest was considered as a single stratum. Thus, preliminary data for a single stratum was used to determine sample size using the formula: $n = \frac{(NxS)2}{\frac{N2xE2}{t2} + NxS2}$

where, n= number of sampling units in the population, N = number of sampling units, E = allowable error, S = standard deviation, t = sample statistics from t-distribution at 95%.

From the preliminary data of Biteyu forest, mean of carbon stock (t ha^{-1}) analyzed = 235.82, Standard deviation = 66.6, E = mean of carbon stock×desired precision (10%, in this case) = 235.82 X 0.1 = 23.582, plot size = 0.09 ha Similarly, from the preliminary data of Boter-Becho, mean of carbon stock (t ha^{-1}) analyzed = 115.3, Standard deviation = 52.19, plot size = 0.09 ha, E = 11.53. The number of plots calculated by inserting these values in the above equation were 30 for Biteyu forest and 71 for the Boter-Becho forest.

3.3.3. Vegetation Data collection

A square sample plot of 900 m^2 was used to collect vegetation data with a DBH ≥ 2.5 cm and a height of 2 m and above. All trees/shrubs with a diameter of 2.5 cm or greater in each plot were counted and recorded. To sample herbaceous vegetation in the forest floor, five smaller subplots of 1 m x 1 m = 1 m^2 (four at the corner and one at the centre of the main plot) were established (Figure 12). The diameter of trees was measured at breast height (DBH, 1.3 m height from the ground) using measuring tape. Moreover, the height of a tree greater than or equal to 2 m were measured using clinometer. Heights were estimated visually in place where the topography or crown structure made it difficult to use the height meter. In cases where trees exhibit buttressed boles, measurements were taken at the points just above the buttresses. In cases where the trees branched at about breast height, the diameter was measured separately for the branches and their average values were considered.

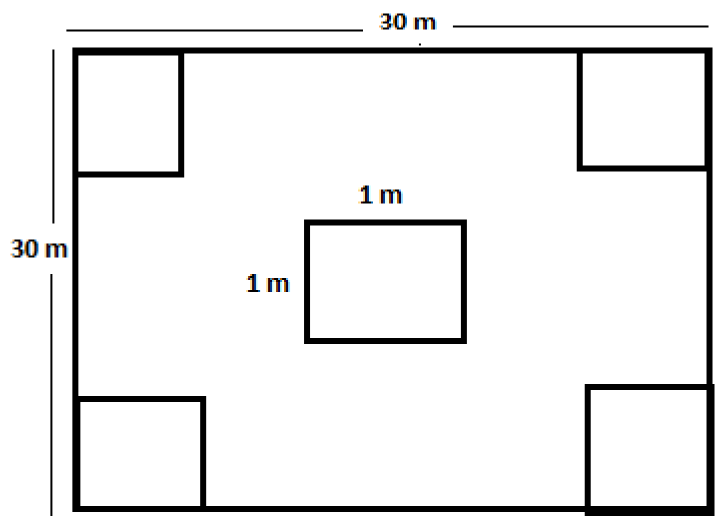


Figure 12. Plot layout for collecting vegetation data

Individuals per plant species were counted in each sample plot. Plant specimens occurring outside the plots were also recorded for enriching the diversity of the flora particularly for the Biteyu forest. Moreover, percentage of canopy cover of each plant species was estimated during data collection in the field. The percentage cover values estimated in each sample plot were converted into cover abundance values using 1-9 modified Braun-Blanquet Scale (Van der Maarel, 1979) and indicated in Table 5. Voucher specimens were collected, pressed, and brought to the National Herbarium (ETH), Addis Ababa University for identification. The specimens were identified using published books of the Flora of Ethiopia and Eritrea, and other taxonomic works and comparing with specimens already deposited in the National Herbarium.

Table 5. Modified Braun-Blanquet Scale for cover abundance values

Scale	Cover abundance values
1	Rare, generally one individual
2	Occasional, less than 5 % cover of the total
3	Abundant, with less than 5% cover of the total
4	Very abundant, with less than 5% cover of the total
5	cover 5-12.5% of the total area
6	cover 12.5- 25% of the total area
7	cover 25-50% of the total area
8	cover 50-75% of the total area
9	cover >75% of the total area

(**Source**: Van der Maarel, 1979)

3.3.4. Above ground Biomass (AGB)

To estimate the above ground biomass, all woody species in sample plots with DBH ≥ 2.5 cm were identified and recorded. For woody species with multiple stems at 1.3 m height, all stems were measured and the averages were taken during data collection. Even though height is an important explanatory variable for evaluating the AGB, many of the allometric equations tend to omit this variable including Brown equations. However, Chave *et al*. (2005) tested explanatory variables such as DBH, height and ρ (specific wood density) for 20 sites in the tropical forests of the world and recommend the inclusion of height in allometric equations. They recommended the following allometric equations for AGB estimation in the dry, moist and wet tropical forests:

$(AGB)_{dry} = \exp[-2.187+0.196*\ln(\rho D^2H)] = 0.112*(\rho D^2H)$, areas with rainfall <1500 mm $year^{-1}$ and a dry season of several months---Eq.1

$(AGB)_{moist} = \exp[-2.997+\ln(\rho D^2H)] = 0.0509*(\rho D^2H)$, moist for areas with 1500-3500 mm $year^{-1}$ and a short dry season.---Eq.2

$(AGB)_{wet} = \exp[-2.557+0.94*\ln(\rho D^2H)] = 0.0776* (\rho D^2H)$, wet for high rainfall area with >3500 mm $year^{-1}$ (standardized to sea level conditions) and no seasonality------Eq.3

Very recently, Chave *et al*. (2014) have improved their previous equations and propose a single allometric equation to estimate tree AGB across vegetation types when wood specific gravity, trunk diameter, and total tree height are available. Most of the variation was found within vegetation types, and the apparent variation among vegetation types appears to reflect small sample sizes. This interpretation is supported by the fact that the form factor (ratio of AGB divided by ρD^2H) varies weakly across vegetation types. Accordingly,

$AGB = a*(\rho D^2 H)^b$ ---Eq.4

Where, coefficients a = 0.0673 and b = 0.976 and parameters, AGB (kg), ρ (g/cm3), D (cm) and H (m).

This model performed well across forest types and bioclimatic conditions. The alternative model for the above, where the exponent was constrained into one is

$AGB = 0.0559*(\rho D^2H)$ ---Eq.5

In order to improve their equation developed in 2005, Chave *et al.* (2014) greatly increased the sampling effort in both dry and wet vegetation types. In the 2005 analysis, only three dry forest sites where tree height was measured were included to form the basis of the dry forest equation such as Australia (46 trees), Yucatan (175 trees) and India (23 trees). In comparison, in their present study, they included 22 dry vegetation sites, and 1891 trees. Likewise, Chave *et al.* (2005) included only three wet forest sites having height data including New Guinea (42 tree species), Puerto Rico (30 tree species) and

Cambodia (72 tree species) but now included 12 wet forest sites and 681 trees in their present analysis.

Thus, according to Chave *et al*. (2014), it should be possible to estimate tree AGB to a good accuracy without felling the tree with additional wood specific gravity measurements. Similarly, they found out that separate regression parameters for the dry, moist, and wet vegetation types do not improve the statistical performance of the model based on their data. They suggested that it is more parsimonious to retain a single allometric model to use for the three vegetation types. They also note that the new model holds for both old growth and secondary vegetation types. Since there are no specific allometric models that have been derived for both of my study areas as well as regions, it was decided to use a generic allometric model in equation (5) for AGB and carbon stock estimation. This is because this allometric equation uses the tree parameters such as DBH, height, and specific wood density by giving more precise result than other allometric equations, which uses DBH only (Djomo *et al*., 2010).

Specific wood density values were obtained from different studies of tropical African forests (Reyes *et al*., 1992; Zanne *et al.*, 2009; IPCC, 2006; Chave *et al*., 2009). In cases where the wood density for a species was not listed, the average specific wood density for its genus was used. when this was not possible, an average value of 0.5 was used as recommended by Chave *et al.* (2005) for trees from tropical areas. It is generally known that tropical African forests have a specific wood density value which ranges from 0.3-0.9 gcm^{-3} (Brown, 1997).

In each plot, AGB for each indvidual tree using Eq.5 was estimated and extrapolated to a per-hectare (ton ha^{-1}). The carbon content of the AGB was estimated by multiplying the values of AGB by default IPCC carbon fraction value of 0.50 (IPCC, 2006) as the dry mass contains 50% organic carbon (Pearson *et al.,* 2005; Penman *et al*., 2003; Brown & Lugo, 1982; Cannel *et al.*, 1995; Dixon *et al.,* 1994; Ravindranath *et al.,* 1997; Richter *et al.,* 1995; Schroeder, 1992).

3.3.5. BGB Estimation

Measurements of belowground biomass are indeed highly uncertain, and the lack of guidelines for measuring carbon stocks in forests empirical values for this type of biomass has for decades been a major weakness in ecosystem models (Pearson *et al*., 2005, 2007). Therefore, it is more parsimonious to apply regression model to determine belowground biomass from knowledge of AGB (Cairns *et al.,* 1997). Applying these equations provide relatively good estimate of below-ground biomass. This is the most practical and cost-effective method of determining biomass of roots. BGB was estimated from AGB using the relationship derived for the tropics by Cairns *et al*. (1997):

$$BGB = \exp(-1.0587 + 0.8836 \ln ABG) \text{ -- Eq. 7}$$

3.3.6. Soil sampling

Composite soil samples were collected from four systematically designated points at the corner and one at the centre of each plot. The depth interval of sampling was fixed to 30 cm. Soil samples were taken from soil depths of 0-30 cm after collecting litter samples from the soil surface. Undisturbed soil samples for bulk density measurement were taken from the same depth intervals using a soil-sampling cylinder with a volume of ($25\ cm^2 \times 5\ cm$) or $125\ cm^3$. The soils were then taken to Holeta Agricultural Research Laboratory, Ethiopia for the following analyses.

3.3.6.1. Soil organic carbon

Soil carbon concentration was determined by *Potassium Dichromate method* (Walkey and Black, 1934) after the soil has been grounded and passed through a 2-mm sieve. Bulk density was determined by weighing a soil sample of known volume (125 cm^3) after drying at 105^0C. The bulk density of soil (BD) was calculated by:

BD = Sw/ V, where S_w is the oven-dry weight of the soil sample, and V is the sample volume.

The soil organic carbon density for a sampling plot was estimated (Tian *et al*., 2009) as:

SOC (ton/ha) = OC/100 * BD * D * 10,000 m^2/ha-------------------------------------Eq.8

Where, OC (%) is the concentration/percent of organic carbon, BD (Mg/m^3) is the bulk density of soil sample, D (m) is the given soil depth. The unit metric tons per hectare were used for soil organic carbon as well as biomass carbon measurements in this study due to the fact that megagram (Mg) is equal to metric tons. The carbon stock density was converted into tons of CO_2 equivalent by multiplying it with 44/12 or 3.67, which is molecular ratio of CO_2 to carbon to understand the climate change mitigation potential of the study area in particular (Pearson *et al.,* 2005).

3.3.6.2. Other Soil Parameters

Moreover, other soil parameters were analyzed using standard methods following Allen (1974). Soil physical properties (soil texture and soil moisture), and chemical properties (PH, EC, total nitrogen, available phosphorus and available potassium) were analyzed for the soil data collected from the Biteyu forest. Moreover, the soil organic carbon, soil pH and soil moisture of the sample collected from the Boter-Becho forest were analyzed. The soil samples were air dried, grounded and passed through a 2-mm sieve before analysis. Roots and bigger litter were then removed from the soil samples. Soil texture was measured by a hydrometer method, pH in a 1:2.5 (v/v) soil: water suspension, electrical conductivity in a 1:2 (v/v) soil: water suspension, available P by the Bray II method and measuring absorbance by the Spectrophotometer, total N by the Kjeldahl method and available potassium in the extract by the flame photometer.

3.3.8. Litter Biomass

The forest litter, herbs and grasses on top of the soil were included in the litter pool in this study. This has been estimated using simple harvesting technique (IPCC, 2003). A small sample plot of $1m^2$ which was established for soil sampling in four corners and the centre of the bigger plot was used to collect all materials inside the plot including herbs, grasses and litter. A well-mixed subsample was collected to determine oven dry-to-wet weight ratio to convert the total wet mass to oven dry mass, which follows the following equation:

$$Cl = \frac{W\,field}{A}\ X\ \ W\frac{subsample(dry)}{subsample(fresh)}\quad \frac{1}{10000} \text{ ---Eq. 9}$$

Where: C_L= biomass of litter, herb and grasses (t ha^{-1})

W_{field} = weight of fresh sample of destructively sampled within an area of size $1m^2$ (g);

W subsample (dry) = weight of the oven-dry sub-sample, and

W subsample (fresh) = weight of the fresh sub-sample taken to the laboratory (g).

The carbon content in forest litter was calculated by multiplying it with the carbon fraction analyzed in the laboratory.

3.4. Environmental Data

Environmental data that are assumed to affect the occurence and distribution of vegetation in both study forest were collected. These data include altitude, slope, aspect and soil of each sample plot. Aspect was determined by using Zerihun Woldu *et al*. (1989) with a modified scale that refers to the total amount of solar energy received, based on N = 0, NE = 1, E = 2, SE =3, S = 4, SW = 3.3, W = 2.5, NW = 1 .3, Ridge top = 4. The disturbance level in each plot of Biteyu forest was subjectively evaluated using ordinal scales from 0 to 3 (where 0 represents the absence of the influence and 3 the highest influence). The disturbance scores were based on the indicators of grazing and browsing, presence of fresh or old cut wood stumps, broken branches and debris, and the occurrence of trampled seedlings in each plot. The scale of 3 represents all the 5 disturbances, 2 where three of the disturbances, 1 where only one disturbance and 0 where none of the disturbances was observed. This scoring was done only for the Biteyu forest but not for the Boter-Becho forest.

The environmenal data collected from the Biteyu forest was only used to see its effect on the plant community identified but the environmental data of the Boter-Becho forest was not included in this study because of the fact that it is being studied by another PhD student at the same time. Moreover, threats to the forest biodiversity in Biteyu was determined by indicating cattle interference and cut wood stumps. The cattle interference, here refers to the number of livestock (cows, oxen, donkeys, sheep and goats) and if not cattle present in each plot, the number of cattle trail and or the number of cattle dung piles recorded in each plot. On the other hand, the number of cut stump (dead or alive) or the number of half looped or 100% looped trees were counted in each plot to determine the human disturbance on the forest ecosystem.

3.5. Litter-trap and litterbag study from Boter-Becho forest

First, the study was proposed to comparatively work on the Biteyu (DAF) and Boter-Becho forest (MAF) which were clearly distinct. However, due to the intensive cattle interference, illegal logging, grazing and browsing activities, which took place in the Biteyu forest, it was impossible to conduct the litter-trap and litterbag experiment. Therefore, the two sites in the Boter-Becho forest based on the local disturbance level was selected namely high-disturbed sites (HD) and low disturbed sites (LD). The local disturbance level was subjectively characterized due to the information obtained from the forest scouts and researcher's observation in the field. The HD site had signs of human trampling, browsing and grazing by domestic animals, and illegal cutting of large trees likely for honeybee harvesting, and fuel-wood collections. The LD site had not suffered of logging, browsing or grazing and only disturbed by human trampling. The forest canopy of HD site was more open than the LD site due to intensity of disturbance. The density (no of stems ha^{-1}), mean of Height ± sd (standard deviation) and total basal area (m^2 ha^{-1}) of dominant tree species in the study plots of LD and HD sites are given in Table 6 and Table 7 below.

Table 6. The tree species in LD site

	LD Site		
	Density	Height	Total BA
Allophylus abyssinicus	311.11	10.4 ± 2.9	29.26
Croton macrostachyus	140.74	18.9 ± 8.0	11.94
Chionanthus mildbraedii	296.31	3.68 ± 1.13	0.7
Ficus sur	85.15	16 ± 3.2	12.43
Macaranga capensis	88.91	18 ± 4.23	21.02
Milletia ferruginea Subsp. *darassana*	177.78	13.15 ± 4.11	12.8
Olea capensis subsp. *macrocarpa*	503.7	21.73 ± 3.49	32.13
Olinia rochetiana	214.81	10.9 ± 2.82	16.89
Podocarpus falcatus	311.11	16.13 ± 3.53	12.98
Polyscias fulva	133.33	11.32 ± 2.45	6.87
Pouteria adolfi-friedricii	162.96	19.2 ± 2.76	18.34
Schefflera volkensii	166.67	12 ± 3.72	14.87
Syzygium guineense Subsp. *afromontanum*	466.67	21 ± 4.22	33.56

Table 7. The tree species in HD site

			HD sites
Scientific Name	Density	Height	Total BA
Albizia gummifera	200.00	15.3 ± 9.6	16.8
Apodytes dimidiata	77.70	15.4 ± 5.5	29
Brucea antidysentrica	125.90	8 ± 3.4	8.6
Calpurnia aurea	281.48	5.75 ± 2.11	2.1
Celtis africana	129.62	6.97 ± 3.7	2.86
Chionanthus mildbraedii	292.60	5.5 ± 2.3	0.81
Clausena anisata	251.85	3.7 ± 2.1	1.08
Croton macrostachyus	77.80	14.67 ± 3.76	4.88
Ehretia cymosa	92.59	4.3 ± 1.8	4.62
Milletia ferruginea Subsp. *darassana*	225.92	13.6±3.27	6.46

Oxyanthus speciosus	88.89	4.5±1.25	0.76
Podocarpus falcatus	59.26	21±4.3	6.62
Pouteria adolfi-friedricii	85.18	18±4.89	5.63
Teclea nobilis	125.90	5.3±2.3	1.2

Unknown proportions of the mixture of the above tree species for each site were used for litter trap and litterbag study to represent forest ecosystem functions.

3.6. Litterfall, Leaf Litter Decomposition and Nutrient Dynamics

3.6.1. Litterfall

It is possible to use a mixture of litters within a bag to match the overall composition of the forest ecosystem in Boter-Becho as described in Blair *et al*. (1990). For the litterfall collection, litter trap (1m^2) made of wooden frame was constructed (Figure 13 and 14). The 1m^2 traps were supported at each corner by wooden pegs to suspend the frame at 80 cm above the ground (Figure 14). Nylon clothes with a mesh size of 2 mm were suspended on the wooden frame prepared allowing for the rapid drainage of rainwater.

Figure 13. Square wooden frame for litter trap preparation

Figure 14. Litter trap for litter collection in the forest

Litterfall sampling and litter decomposition were time-consuming and expensive. Hence, only five larger plots (30 m x 30 m) each were selected from LD and HD sites of the forest for litter trap and litter bag study. The five plots in each site were selected based on altitudinal gradient spanning from 2000 to 2800 m a.s.l to represent Boter-Becho forest. In each larger plot, two litter traps were deployed with a total of ten in each site of the forest; which means that a total of 20 litter traps were deployed in both sites of the forest. Litter traps were installed on 4th Feb.2014 on all sites of the forest. The litterfall was collected at monthly intervals for 12 months (4th March 2014 to 3rd February, 2015). A total of 240 litterfall collections were made throughout the whole sampling period (Table 8). However, the average of two replicates in each plot was taken in each sampling period for the data analysis.

Table 8. Total litter traps for litterfall collections

Sites	No litter traps	Sampling period	Total
LD sites	5 plots × 2 traps each =10	12 (Once in a month)	120
HD sites	5 plots×2 traps each= 10	12 (Once in a month)	120
Total	20	12	240

The collected litterfall at each sampling time was oven-dried at 80 ^{0}C for 24 hrs to constant weight. The collected forest litter was sorted into leaf, wood/twigs < 2.5 cm in diameter, reproductive parts: flowers, fruits, and seeds, and finest litter/trash (any material passing through a 2-mm sieve, named *'other'* throughout this thesis). These fractions were oven dried at 80 °C to a constant weight and weighed. The collections in each month were combined to obtain litterfall data per year. To calculate litterfall flux per plot, all the oven-dry weights of the litters from a plot was added up to get the total litter mass (gm^{-2}) and converted into t ha^{-1} yr^{-1} (note: 1 g m^{-2} = 0.01 t ha^{-1}).

3.6.2. Leaf Litter Decomposition and Nurient Release

Litter decomposition was studied using the litterbag method. Freshly fallen composite leaf litter from the forest floor was collected in February 2014 and air-dried for one week. The 20 g of leaf litter was added into 2 mm mesh size litter bags made of nylon cloth, 25 x 25 cm in size, which were then tied, and closed according to Salinas *et al*. (2011). In order to tie, the end of the litterbag was folded over and sealed with staples (a non-reactive alloy) using 5 to 6 staples per one bag. The electronic balance (DIAL-O-GRAM, OHAUS, USA) with a capacity of 2610 g and precision of ± 0.01 were used for the weight measurement of leaf litter. Ten plots of size 30 m x 30 m in both LD and HD sites, five in each site were established for litter trap. To determine the mass-loss during leaf litter decomposition, the same five plots selected for the litter trap experiment in each site were simultaneously used.

Fifteen litterbags were buried in the top soil in the centre of each plot with a sub-total of 75 (15 × 5) in each site, with 150 litterbags in both LD and HD sites of Boter-Becho forest. Of 150 litterbags, only 120 were used for the data analysis and the remaining 30 were deliberately included to replace in cases when the litterbags missed. The distance between each litterbag was 50 cm. Of the total, ten litterbags, one from each plot, were randomly retrieved at monthly intervals for 12 decay periods (starting from Mar. 2014 to Feb. 2015). Similarly, to determine the rate of nutrient release pattern during decomposition, litter bag samples retrieved from the three purposefully selected plots (1^{st}, 3^{rd} and 5^{th}) in each site, a total of 72 (3 plots × 12 litter-bags × 2 sites) were used and analyzed for chemical contents. A total of 120, 60 litterbags for each sites were retrieved for decomposition study (Table 9).

Table 9. Total litterbags for leaf litter decomposition

Sites	Litterbags	Sampling period	Total collections
LD sites	5 plots×1 litterbag	12 (Once in a month)	60
HD sites	5 plots×1 litterbag	12 (Once in a month)	60
Total	10	12	120

The content of each bag was emptied, taking care not to lose any material. Extraneous materials such as visible animal materials and fine roots were removed. The rest was oven-dried at 80 °C for 24 hrs to a constant weight and then weighed. Sub-samples of the air-dry forest leaf litter of each sites was oven-dried at 80°C to constant weight to obtain a conversion factor for calculating the oven-dry weight of forest litter for each bag as the air-dry weight will vary from lab to lab and from year to year as the humidity of the room changes. Thus, the air-dry weights were converted to the oven-dry weights based on conversion factor. The conversion factor (CF) was calculated from each subsample as: $CF \ = \frac{\text{Oven−dry weight(g)}}{\text{Air−Dry Weight(g)}}$

To get the oven-dry weight for each litterbag, the air-dry weight was multiplied by the corresponding CF:

$$\text{Oven} \quad \text{dry weight(g)} = \text{Air\_dry weight} \times \text{CF (x)}$$

The dry weight of the initial leaf litters from each plot of both sites was determined and a subsample was ground and analyzed for C, N, P, and K to examine the pattern of nutrient release in the forest system (See section 3.6.3). Mass loss over time was computed using the negative exponential decay model (Olson, 1963). Net nutrient fluxes were derived from nutrient concentration and mass-loss data. Percent nutrient remaining at time t was calculated as the product of percent mass remaining and nutrient concentration in the residual material at time divided by the initial nutrient concentration of that litter type (Blair, 1988; Bockheim *et al.,* 1991). More explanation was given in data analysis section 3.7.4.

3.6.3. Chemical Analysis of Leaf Litter

In order to estimate initial forest leaf litter chemistry, carbon (C), nitrogen (N), phosphorous (P) and potassium (K) concentration were determined in mixture of subsamples taken from each plot before the experiment. Carbon and the same nutrients were analyzed at each time a litter bag was taken out for measurement of remaining mass. Forest leaf samples taken before and during the experiment period were ground in a

Wiley mill for chemical analysis according to the following method. The organic carbon concentration was determined using Walkley-Black Method (Anderson and Ingram, 1993), total nitrogen concentration with sulphuric acid digestion followed by distillation and titration (Anderson and Ingram, 1993). Available P concentration was determined using sulphuric acid digestion followed by colorimetric determination (Anderson and Ingram, 1993), available potassium concentration (K) using flame photometry (Allen, 1974). This analysis was done in the Plant Laboratory of Hawassa University, College of Agriculture, Ethiopia.

3.7. DATA ANALYSIS

3.7.1. Vegetation Data of Biteyu Forest

Shannon Wiener (1949) index of species diversity was applied to quantify species diversity and richness. The diversity and evenness were often calculated using Shannon diversity index (Kent and Coker, 1992);

$$H' = -\sum_{i=1}^{S} Pi \ln pi$$

Where: $\sum$ = summation symbol; ln = $_{\log}$ base n, and P_i= the proportion of individuals or the abundance of the species i^{th} expressed as a proportion of total cover in the sample. Measuring species diversity is for the following major reasons, i.e. to measure stability to determine if an environment is degrading and compare two or more environments.

Evenness or equitability (E) is used to quantify the unique representation of a given species against a hypothetical community in which all species are equally common, such that when all species have equal abundance in the community and hence, evenness is maximal. One of the simplest means of analyzing floristic vegetation data is to look at the degree of association between species and level of similarity between samples (Kent and Coker, 1992). Evenness index (J) was calculated using the formula**:**

$$J = \frac{H'}{H'_{max}} = \frac{-\sum_{i=1}^{s} pi \ln pi}{\ln S}$$

Where, J = evenness; H' = Shannon-Wiener Diversity Index; S = total number of species in the sample. The value of evenness index falls between 0 and 1. The higher the value of evenness index, the more even the species is in their distribution within the given area (Kent and Coker, 1992).

Sample plots of the Biteyu forest of the Gurage mountain chain were grouped into clusters with the help of Multivariate analysis using R statistical software (R Core Team, 2013). For cluster analysis, the Relative Euclidean Distance measure by using Ward's method (linkage) was applied. The Euclidean Distance was used due to the reason that it eliminates the differences in total abundance among sample units. Similarly, Ward's method was used as it minimizes the total within-group mean of squares or residual sum of squares.

The community types distinguished were further refined in a synoptic table where each column represents a community type and species occurrences were summarized by Synoptic cover-abundance values. Here, the synoptic values indicate the average cover-abundance values. Finally, the types were named after two dominant and/or characteristic species in the community. The description of plant community also involves the analysis of species diversity, evenness and similarity (Whittaker, 1975).

Species richness of each community was compared. The floristic similarity of the vegetation in Biteyu forest was compared with the vegetation of other forests in the country in terms of species composition using the Sorensen's coefficient of similarity (Ss) (Kent and Coker, 1992). Sorensen's coefficient of similarity value ranges from 0 (complete dissimilarity) to 1 (total similarity). Floristic similarity of forests were calculated by using the formula;

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

where, Ss = Sorenson Similarity coefficient; a = the number of species common to both forests; b = the number of species in one of the forests to be compared; and c = the number of species present in the other forest.

Cattle intereference and wood stump was recorded as indicator of disturbance in Biteyu forest. The relationship of these two factors with the gradient of altitude was analyzed by using simple linear regression. Moreover, the community types identified from the cluster analysis and the environmental variables were drawn in ordination. Data on species distribution in the forest were analyzed by canonical correspondence analysis (CCA). To test for the significance of the environmental variables determining patterns of communnity formation and species distribution in Biteyu forest, one way ANOVA with a Post-Hoc comparison test of Tukey's honestly Significant Difference (Tukey's HSD) was used. Tukey's test compares all possible pairs of means. It identifies any difference between two means that is greater than the expected standard error.

3.7.2. Structural Analysis of Biteyu and Boter-Becho Forests

The structure of the forest was analyzed in terms of tree density, girth diameter, height, important value and basal area per hectare for both Biteyu and Botor-Becho forest. Tree density and basal area were computed from the plots in a hectare basis. The DBH classified into nine DBH-size classes and the percentage distribution of woody species in each class were computed. Tree height was also classified into seven height classes and the percentage distribution of woody species in each height class was calculated. Structural comparisons were made in between this study areas and other forests of the country studied by other scholars.

Basal area is the cross-sectional area of tree stems at breast height. It accounts a measure of dominance. Here, dominance refers to the degree of coverage of species as an expression of the space it occupies (Barbour *et al*., 1987). Basal Area can be calculated as; $\mathrm{BA} = \frac{\pi d^2}{4}$

Where: BA= Basal Area in m^2 per hectare; d = diameter of tree stem at breast height.

Importance value Index (IVI) was computed for all woody species based on their relative density (RD), relative dominance (RDom) and relative frequency (RF). This index is used to determine the overall importance of each species in the forest system. In calculating this index, the percentage values of the relative density, dominance, and frequency are summed up together and this value is designated as the Importance Value Index or IVI of the species (Curtis, 1959; Kent and Coker, 1992). IVI can be calculated as follows:

IVI = Relative density (RD) + Relative dominance (RDO) + Relative frequency (RF), where

$$RD = \frac{\text{No. of individuals of the species}}{\text{No. of individual of all the species}} \text{ X } 100$$

$$RDO = \frac{\text{Total basal area of the species}}{\text{Total basal area of all the species}} \text{ X } 100$$

$$RF = \frac{\text{No. of occurences of the species}}{\text{No. of occurences of all the species}} \text{ X } 100$$

3.7.3. Forest Carbon Estimation

The total carbon stocks of the forest were calculated by summing up the carbon stocks of the individual pool in each study forest using the following formula.

C density = C_{ABG} + SOC+C_{BGB} + C_L

Where, C density = Carbon stock density for all pools for each study sample (t C ha^{-1})

C_{ABG} = Carbon stock in above ground tree biomass (t C ha^{-1})

SOC= soil organic carbon (t C ha^{-1})

C_{BGB} = Carbon stock in below ground biomass (t C ha^{-1})

C_L = Carbon stock in litter, herb and grasses (t C ha^{-1})

The estimated carbon stock density was multiplied by 44/12 or 3.67 to convert into CO_2 equivalent so as to estimate carbon sequestration potential. The individual carbon pool and total carbon stocks were compared between the two forests though they are quite different from each other in terms of species composition, forest type, disturbance and forest size. Distribution of AGB in a range of DBH size classes was considered to assess the potential of the forests across their size classes. In addition, the relationship between carbon of AGB and plot level species diversity of Biteyu forest was analyzed by using simple linear regression analysis. The same analysis method was used to examine the effect of terrain variables such as altitude and slope on the different carbon pool in both

Biteyu and Boter-Becho forests. The mean of four different carbon pools were compared and tested for the significant difference using one way ANOVA at $\alpha = 0.05$.

3.7.4. Litterfall, leaf Litter Decomposition rate and nutrient release

To calculate litter-fall fluxes per plot add up all of the oven-dry weights from all fractions in a plot to get the total litter mass in grams. Then, this value was converted into ton ha^{-1} (note: $g\ m^{-2} = 0.01\ t\ ha^{-1}$).

The leaf mass loss and decomposition rate constants (k) were computed using a single negative exponential decay function first proposed by Jenny *et al.* (1949) and described in detail by Olson (1963): $X_t = X_o e^{-Kt}$.

The exponential model was transformed to natural logarithm, $Kt = -\ln(Xt/X_o)$, where k is the decomposition constant ($year^{-1}$), t is decomposition time (in months), X_o is the initial litter mass at time zero (100%), X_t is the dry mass of the remaining litter for each sample in each sampling time. Mean residence time (R_t) of leaf-litter in each sample was estimated by the inverse of k calculated as: $R_t = 1/K$.

% Mass Loss = 100 * $(X_o - X_t)/X_o$ and **%** Mass Remaining in the litter-bag = 100 - % Mass Loss

As proposed by Harmon *et al*. (1986), the time at which 50% of the material decomposed was estimated by the equation; Half-life ($t_{0.5}$) = ln (0.5)/ (-K) = -0.693/ (-K) = 0.693/K and 95 percent ($t_{0.95}$) of the material are often reported as $t_{0.95}$=3/k. Nutrient content of the litter was calculated using the formula.

$$N = \frac{Ci}{Co} \times \frac{Mi}{M0} \times 100$$

Where, N is percentage of nutrient remaining in the litter. Co = is the concentration of element in the initial litter kept for decomposition; Ci = is the concentration of the element in litter at the time of sampling; M_0 = the initial dry mass of the leaf litter kept for decomposition; Mi = the mass of dry matter at the time of sampling (Sangha *et al.*, 2006; Bockheim *et al.,* 1991).

The percentage of nutrient released from the litter mass was calculated as **100-N**. Monthly dry weights of litterfall (t ha^{-1}) from each site was analyzed for mean variation using independent samples t-test at $\alpha = 0.05$. To analyze the decomposition through time, the mean litter dry mass of each sampled period was calculated and these values adjusted to an exponential equation above in which k is the coefficient of decomposition/decomposition constant. The monthly coefficient of decomposition was summed up to obtain the annual decomposition coefficient (t = 12 months) of the forest.

Stepwise multiple linear regression analysis was performed to determine how much of the total variability in litter decomposition can be explained by mean monthly rainfall, temperature, soil moisture and soil pH. Moreover**,** Pearson's product-moment correlation coefficient analysis was applied to see the correlation between dry mass remaining, and climate variables (mean monthly rainfall and temperature), soil pH and moisture during decomposition. The differences between dry mass, C, N, P and K concentrations remaining in decomposing leaf litter between LD and HD sites as a function of soil and climate variables were determined by simple linear regression. The rank order of the C and nutrient release pattern was determined from the percentage of the net release of each element at the end of the experiment period. The significance difference was evaluated by ANOVA at $P < 0.05$. R-software was used (R Core Team, 2013) for the data analysis.

CHAPTER FOUR

4. RESULTS

4.1. Floristic diversity of Biteyu forest

The floristic diversity and species composition of Biteyu forest in the Western Escarpment of the Central Rift Valley was assessed. A total of 190 species distributed among 154 genera and 73 families were recorded. Asteraceae was the family with the highest number of species (29 species, 15.26%) represented by 22 genera followed by Lamiaceae (14 species, 7.37%) represented by 11 genera and Rubiaceae (8 species, 4.21%) were represented by 7 genera. Moreover, Fabaceae, Apiaceae, Rosaceae and Solanaceae contribute 7 (3.68%), 6 (3.16%), 5 (2.63%), 5 (2.63%) species respectively to the total species composition (Apendix II). A small number of tree and shrub species dominate this forest vegetation. About 53.3% of the total abundance is contributed by only five species, namely *Olinia rochetiana, Maytenus addat, Ilex mitis, Maesa lanceolata* and *Vernonia rueppellii*. Similarly, a relatively small number of species were widespread in the forest with only nine species occurring in more than 80% of the plots sampled and more herbs than tree and shrubs were encountered in the forest.

4.2. Growth forms of the Biteyu forest

All plant species from the *Biteyu* forest were grouped in the following categories of the growth form (Table 10). Of the total species richness, herb accounted for (44.74%) 85 species. Shrub, tree, climber and fern accounted for 24.21% (46 spp), 15.79% (30 spp), 7.37% (14 spp) and 10 5.26% (10 spp) respectively. Altogether, herb, shrub, tree and climber accounted for 92.11% of the total species richness while fern, epiphyte and parasite contribute for the remaining proportion. The above classification was done based on the information obtained from fieldwork and Flora of Ethiopia and Eritrea.

Table 10. Growth form of all the species identified from the forest

Category	**Number**	**Percent**
Herb	85	44.74
Shrub	46	24.21
Tree	30	15.79
Climber	14	7.37
Fern	10	5.26
Parasite	3	1.58
Epiphyte	2	1.05
Total	190	100

4.3. Endemic Taxa identified from Biteyu forest and its distribution in Flora Area

Twenty species (10.53%) of the total 190 identified from Biteyu forest were found to be endemic taxa to the Flora Area (Appendix III). Family Asteraceae accounted for 8 species (40%) of the endemic taxa followed by two species (10%) contributed by family Lamiaceae and the remaining families in total contributed 50% to the endemic taxa of the study area. The endemic species and the level of their preliminary conservation status assessment results based on IUCN Red List Categories and criteria and available literature (Ensermu Kelbessa *et al*., 1992; Tesfaye Awas (EBI Report on Endemic plants of Ethiopia); Vivero *et al*., 2005) was presented in Appendix III.

4.4. Plant community types of the Biteyu forest

From the result of cluster analysis of agglomerative hierarchical classification using Euclidean distance based on Ward's method, three plant community types were identified. Vegetation clustering considers classification of plant communities by taking the dominant species as representatives of that particular community (Figure 15).

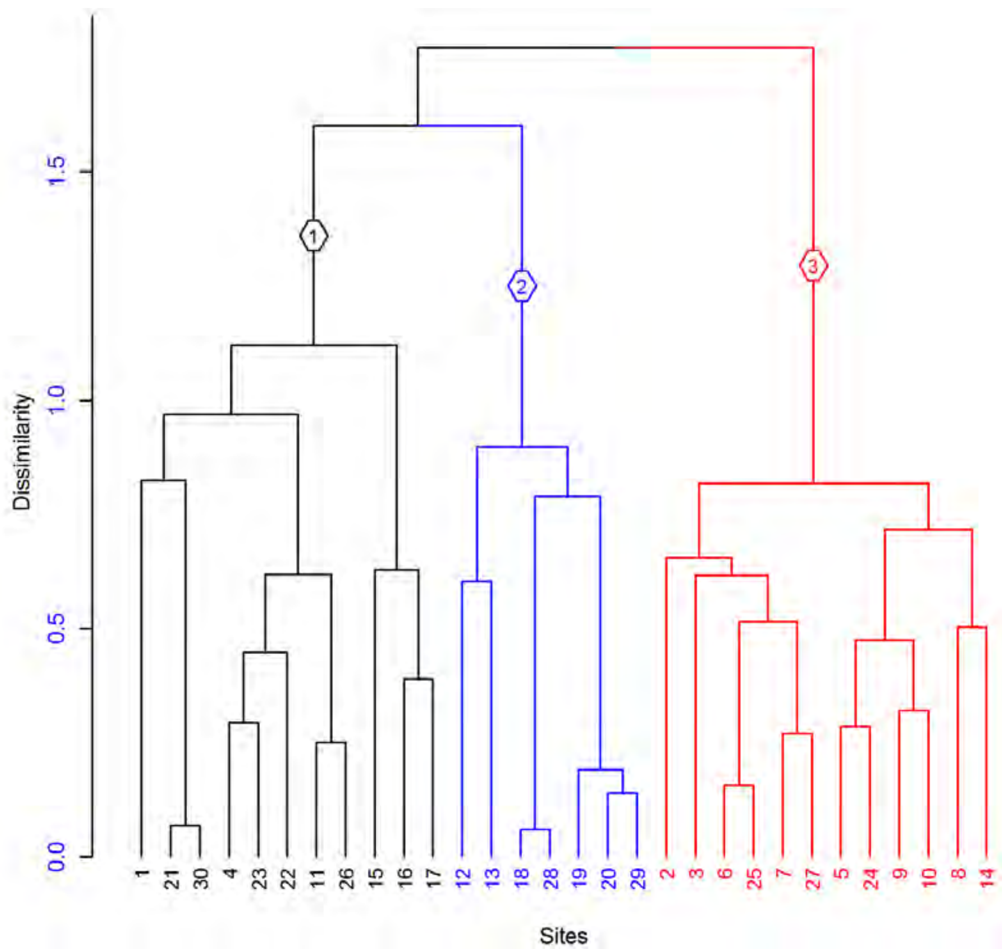


Figure 15.Dendrogram of sample plots in Biteyu forest

The community is named after the two dominant tree or shrub species in that specific community based on their average synoptic values (Table 11 and Appendix IV). The characteristics of the identified community types of Biteyu forest are described below.

4.4.1. *Olea europaea* subsp.*cuspidata-Hagenia abyssinica* community

This community consists of 11 of the total plots sampled and 49 species. The altitudinal range of this community is from 2725 to 2970 m above sea level. About 72.70% of the plots of this community are found above 2800 m above sea level. About 100% of the plots in this community exhibit slope greater than 30 and less than 53 degrees, consisting of an undulating and very rugged topography. The characteristic tree and shrub species more important to this community include *Olea europaea* Subsp. *cuspidata, Hagenia abyssinica, Myrica salicifolia, Erica arborea, Juniperus procera and Dombeya torrida.* The climber species representing this community include *Clerodendrum.myricoides.*

4.4.2. *Maesa lanceolata-Brucea antidysenterica* community

This community consists of 7 plots composed of 39 species. The community has an altitudinal range of 2726 to 2835 m above sea level. The plots of this community exhibit slope ranging from 31 to 40 degrees. The common tree and shrub species of this community include *Maesa lanceolata, Brucea antidysenterica, Ilex mitis, Galiniera saxifraga, Vernonia rueppellii, Maytenus addat* and *Olinia rochetiana.* Some of the understorey species of this community type include *Solanum benderianum, Satureja biflora, Pluchea discoridis, Solanum indicum, Solanecio mannii* and *Sparmannia ricinocarpa. Smilax aspera* constitutes the climber of this community among others. High disturbance, particularly high grazing intensity and high number of cut wood stumps were observed in the plots of this community.

4.4.3. *Olinia rochetiana-Maytenus addat* community type

This community consists of 12 plots composed of 75 species. The community has an altitudinal range of 2703 to 2897 m above sea level. The slope of this community ranges from 33 to 55 degrees. The common tree and shrub species of this community include *Olinia rochetiana, Maesa lanceolata, Maytenus addat, Schefflera volkensii, Ilex mitis, Myrsine melanophloeos, Podocarpus falcatus, Bersama abyssinica, Nuxia congesta, Discopodium penninervium, Schefflera abyssinica. Mikanopsis clematoides* is representative climber of this community type.

4.5. Species Diversity of Biteyu Forest

In order to measure species diversity in each community type, the Shannon diversity index was used because the Shannon index expresses the relative evenness or equitability of species. Accordingly, *Olinia rochetiana-Maytenus addat* community type had higher species richness (75) and diversity (H=3.40) (Table 11) than others. The high species diversity in this community type may be attributed to the higher disturbance in the plots sampled than the other two. Since species evenness shows the relative proportional abundance of species in a given community, the higher value indicates, the more even the species and the reverse indicate very few species dominating the community. Consequently, all the species had even distribution in the three community types having the evennenes values of 0.84, 0.79, and 0.78 for community types 1, 2 and 3 respectively. In general, the species diversity and evennes of the Biteyu forest is 3.18 and 0.8, respectively.

Table 11. Richness, Diversity and evenness of the three community types

Community	Richness	H	Evenness
I	49	3.25	0.84
II	39	2.90	0.79
III	75	3.40	0.78
Mean		3.18	0.80

The species recorded from the Biteyu forest was compared with some other similar forests of the country and similarity analysis was performed using Sørensen's similarity index as shown in the discussion section.

4.6. Importance value index of woody species in Biteyu forest

Importance value index is the most important parameter to compare the ecological significance of species in the given plant community. It is noted that species with the greatest importance value are the dominants of the forest. Accordingly, analysis of the importance value index (IVI) of woody species revealed that the forest is dominated by a very few species. Woody species with the highest IVI values in decreasing order were *Olinia rochetiana, Maesa lanceolata, Ilex mitis, Maytenus addat, Podocarpus falcatus, Schefflera volkensii* and *Olea europaea* subsp.*cuspidata* (Table 12). These were the most important species with high importance value index due to their ecological importance in the Biteyu forest. The above woody species contributed 51.76% of all the woody species involved in the IVI calculation while 48.24% were contributed by the remaining species.

Table 12. The IVI of woody species in the Biteyu forest

No.	Scientific Name	RD	RF	RDom	IVI
1	*Bersama abyssinica*	3.59	3.45	0.75	7.78
2	*Brucea antidysenterica*	3.69	3.13	0.75	7.58
3	*Buddleja polystachya*	0.21	0.63	3.55	4.39
4	*Canthium oligocarpum*	0.10	0.63	0.19	0.92
5	*Conyza hypoleuca*	0.21	0.63	0.45	1.28
6	*Croton macrostachyus*	0.42	2.19	4.59	7.20
7	*Discopodium penninervium*	2.03	3.76	0.67	6.46
8	*Dombeya torrida*	0.57	1.88	1.49	3.94
9	*Dregea schimperi*	0.26	1.25	0.18	1.70
10	*Ekebergia capensis*	0.16	0.94	6.26	7.36
11	*Embelia schimperi*	0.16	0.94	0.39	1.49
12	*Erica arborea*	2.60	1.57	0.05	4.21
13	*Galiniera saxifraga*	1.09	2.51	0.59	4.19
14	*Hagenia abyssinica*	3.43	3.13	2.45	9.02
15	*Ilex mitis*	4.11	7.84	8.46	20.41
16	*Juniperus procera*	1.04	1.88	4.56	7.48
17	*Maesa lanceolata*	16.70	8.46	1.95	27.12
18	*Maytenus addat*	5.52	6.90	5.65	18.22
19	*Myrica salicifolia*	1.30	2.51	2.72	6.48
20	*Myrsine africana*	0.21	0.63	0.08	0.92
21	*Myrsine melanophloeos*	3.28	4.70	2.48	10.46
22	*Nuxia congesta*	3.02	4.70	1.77	9.49
23	*Olea europaea* subsp. *cuspidata*	2.76	4.39	3.86	10.95
24	*Olinia rochetiana*	20.50	7.84	1.86	30.88
25	*Osyris quadripartita*	1.09	3.13	0.20	4.43
26	*Pittosporum abyssinicum*	0.26	0.94	1.17	2.37
27	*Podocarpus falcatus*	9.11	4.08	3.04	16.22
28	*Prunus africana*	0.21	0.63	4.19	5.02

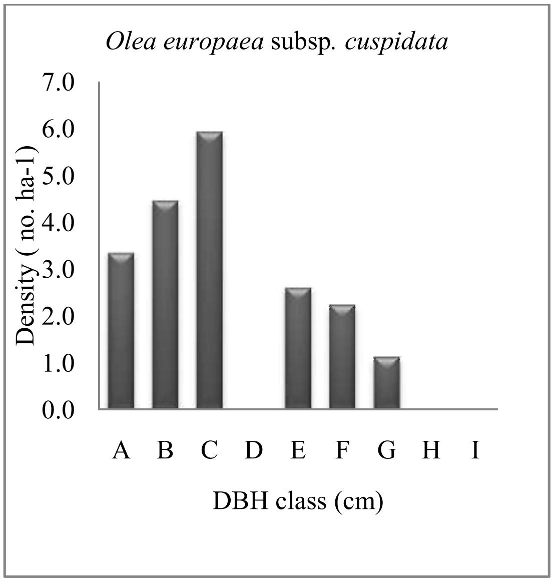
29	*Rhus glutinosa*	0.21	0.63	3.29	4.13
30	*Schefflera volkensii*	4.01	4.70	4.85	13.66
31	*Vernonia rueppellii*	6.71	2.82	0.31	9.84

4.7. Population Structure of Biteyu Forest

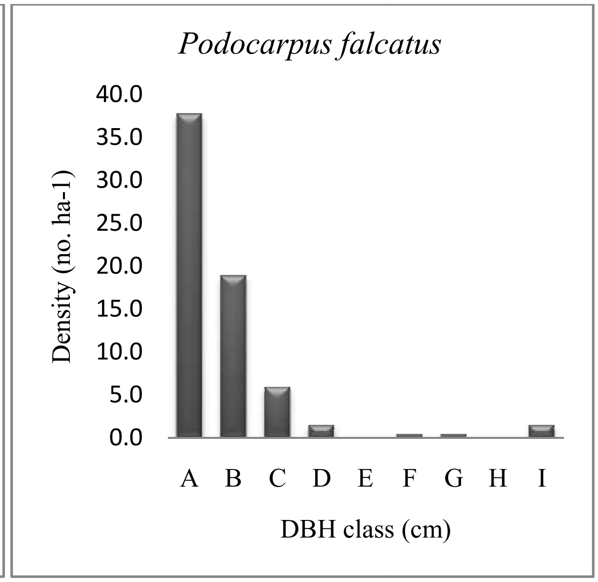
Population structure of the selected tree species in the *Biteyu* forest showed variable structural patterns (Figure 16). The first pattern is an inverted J-curve formed by *Olinia rochetiana, Maesa lanceolata* and *Nuxia congesta* (not shown). These are woody species represented with the highest number of individuals in the lower DBH classes and a gradual decline in number of individuals toward higher DBH classes. The second pattern was an interrupted inverted J curve type of species population structure. This pattern was represented by *Juniperus procera* and *Podocarpus falcatus.* There was no individual recorded in the 10-20 cm DBH class for *Juniperus procera* and in the 40-50 cm DBH class for *Podocarpus falcatus* due to selective logging.

The third pattern was represented by *Olea europaea* subsp.*cuspidata and Schefflera volkensi,* which showed a Gauss curve type of species population structure. However, there were no individuals recorded in the fourth DBH class for *Olea europaea* subsp. *cuspidata.*

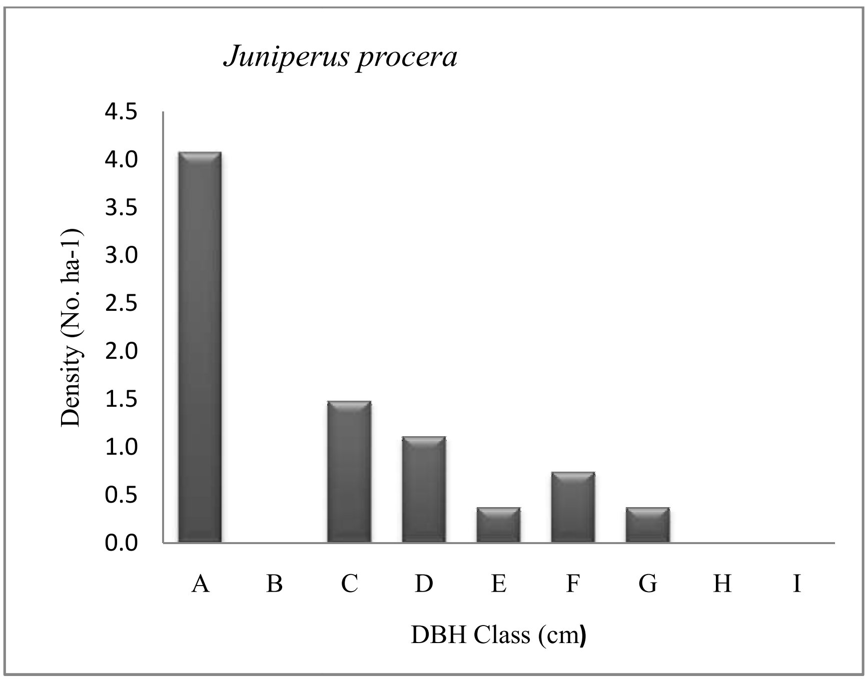
The fourth pattern was represented by *Ilex mitis* in which its density increases as DBH increases even if there was a decrease in the number of individuals in the middle of the DBH class distribution. An equal number of stems per ha (4.07) was recorded in the DBH classes of 10-20 cm, 20-30 cm and 50-70 cm for *Ilex mitis*.



Olea europaea subsp. cuspidata
7.0
6.0
5.0
4.0
3.0
2.0
1.0
0.0
Density (no. ha-1)
A B C D E F G H I
DBH class (cm)



Podocarpus falcatus
40.0
35.0
30.0
25.0
20.0
15.0
10.0
5.0
0.0
Density (no. ha-1)
A B C D E F G H I
DBH class (cm)



Juniperus procera
4.5
4.0
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
Density (No. ha-1)
A B C D E F G H I
DBH Class (cm)

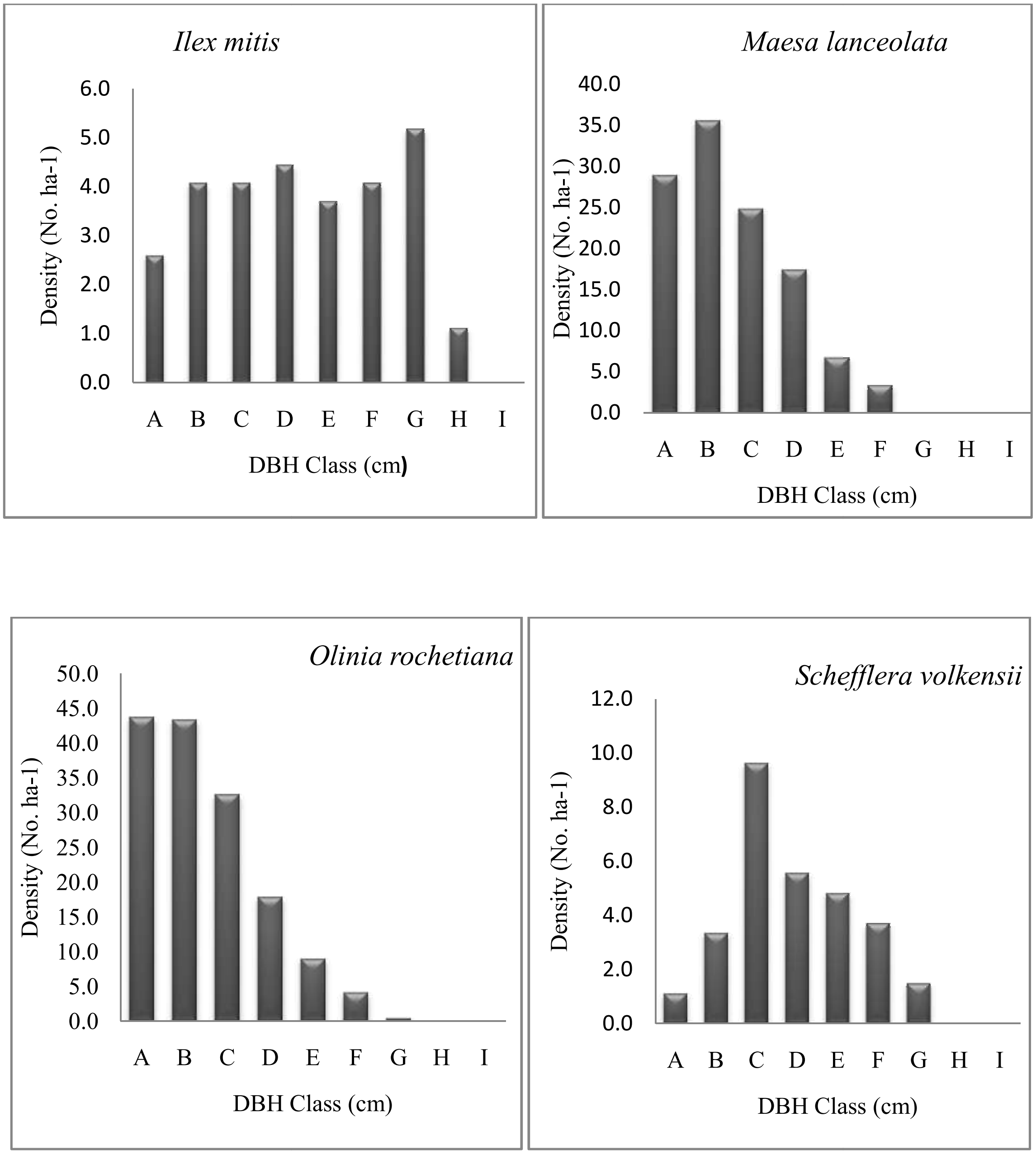


Figure 16.Diameter class distribution of woody species selected from Biteyu Forest. Diameter classes: A = 2.5-10 cm, B = 10-20 cm, C = 20-30 cm, D = 30-40 cm, E = 40-50 cm, F = 50-70 cm, G = 70-90, H = 90-110, I ≥ 110.

4.8. Ordination

Species composition and distribution in the forest is controlled by environmental variables, which can be explained by ordination method. For instance, if two sites are close to each other in ordination, they have similar vegetation and if two sites have similar vegetation, they in turn have similar environment. Moreover, if two sites are far away from each other in ordination, they have dissimilar vegetation, and perhaps the reverse is also true (Jongman *et al*., 1987). CCA was applied with the objective of determining environmental gradients within species data and assessing the relative importance of environmental variables. After CCA has been applied, a Monte Carlo test technique with automatic forward and backward selection of variables was used for statistical significance. In this analysis, the model refers to the constrained variable and residual refers to the unconstrained variable of the ordination. ChiSquare is the corresponding inertia and D_f the corresponding rank. The test statistic F, or more correctly "pseudo-*F*" is defined as the ratio of Chisq to D_f of the model divided by the ratio of Chisq to the Df of the residual (Table 13).

Table 13. Result of Permutation test of Environmental variables

Variable	Df	Chisq	F	N.perm	Pr (>F)
Altitude (m)	1	0.2020	2.1393	199	0.005**
Slope (^{0})	1	0.1776	1.8807	199	0.005**
OC (%)	1	0.1379	1.4605	199	0.10*
TN (%)	1	0.0692	0.7328	199	0.975
EC (mS/cm)	1	0.1609	1.7032	199	0.025**
pH	1	0.1290	1.3655	199	0.190
P(ppm)	1	0.1245	1.3185	199	0.200
K(ppm)	1	0.1489	1.5766	199	0.030**
C:N	1	0.1055	1.1174	199	0.390
Sand (%)	1	0.0876	0.9270	199	0.725
Clay (%)	1	0.0902	0.9555	199	0.745
Disturbance	1	0.1206	1.4768	199	0.09*

Residual	17	1.6056

*Note: Df = degree of freedom, F = F test model, Pr = probability, Sig = ** significant level at p<0.05 and * significant level at P<0.1*

CCA was plotted using cover abundance values of plant species and data of significant environmental variables. The direction of the arrow shows the gradient of the environmental variable whereas the length shows its importance. Ordination of plots with significant environmental variables (P ≤ 0.1) were shown using CCA (Figure 17). The constrained variables indicated in the diagram significantly influence the species distribution pattern of the respective community structure in the Biteyu forest. Some of the constraints were aliased because they were collinear/redundant. For instance, silt (%), with higher *P*-value was not explained in the permutation test.

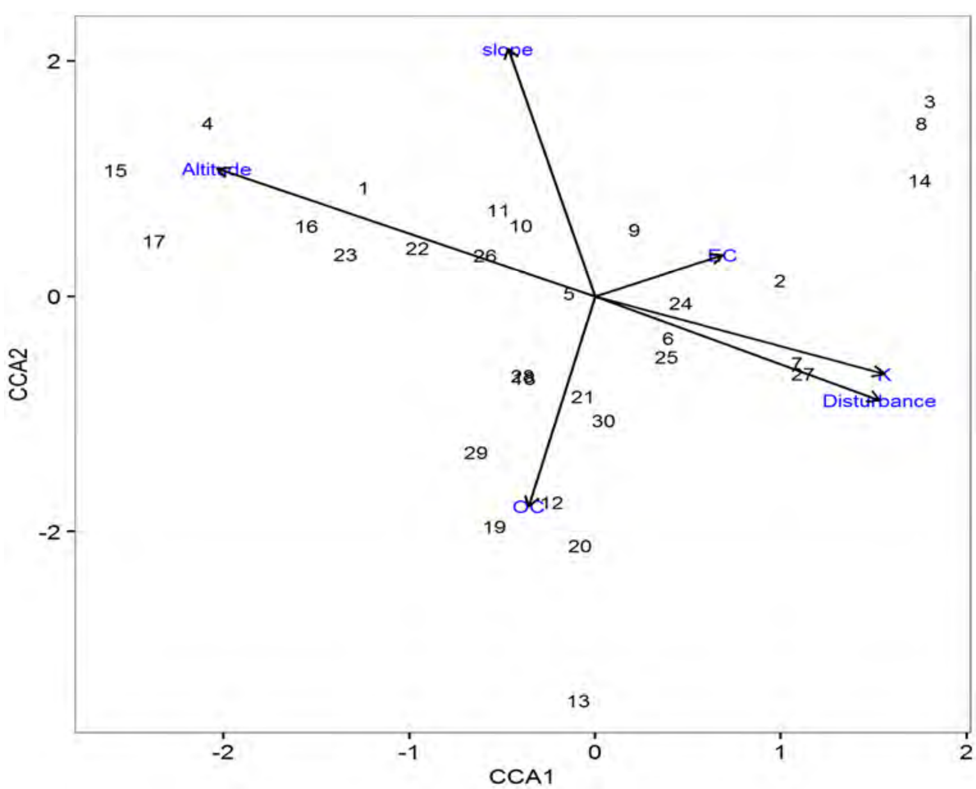


Figure 17. CCA ordination of sample plots and environmental variables

The constraining variable with the highest score associated to CCA1 (0.537) was altitude and it is the most important variable in weighing axis one in ordination. Similarly, slope was the most important constraining variable in weighing axis two with the biplot score of 0.786. Disturbance and K were also the important environmental variable in weighing axis one with a biplot score of 0.405 and 0.409 respectively (Table 14). Importance of the constraining variables is in accordance with their scores where axis one is the most important in explaining variation of patterns in species composition, and then variation explained by higher axes decreases successively (Kent and Cooker, 1992). Consequently, the eigenvalue for axis one was 0.275 which is higher than the eigenvalues of the remaining five axes (Table 14).

Moreover, cumulative proportion of variance explained by the first six axes of the joint plot in the constraining biplot was 74.34%. About 23.84% and 19.30% of the proportion of variation explained by the six CCA axes were explained by CCA1 and CCA2 respectively. This showed that 43.14% of the variation in patterns of plant species distribution and plant community formation was explained by CCA1 and CCA2. It is generally accepted that constraining variables highly correlated with axis one contributed more eigenvalues indicating that more variation was explained by these constraining variables (Table 14). The constraining variable highly correlated with axis one was altitude. Disturbance and K environmental variables to a lesser extent than altitude were correlated to axis one. However, among the constraining variables, altitude in particular, contributed significantly in explaining pattern of variation in the species distribution and type of plant community formation. About 17.72% of the variation in the pattern of plant species distribution and community formation was explained by axis one, followed by axis two (14.35%), axis three (12.62%) and axis four (11.04%). Similarly, the importance of higher axis in explaining the variation in the pattern of plant species distribution and community type formation were decreased successively.

Table 14. Biplot scores for constraining variables

Variable	**CCA1**	**CCA2**	**CCA3**	**CCA4**	**CCA5**	**CCA6**
Altitude (m)	-0.537	0.5674	0.181	0.231	0.2844	-0.103
Slope (^{0})	0.009	0.7863	0.059	0.090	-0.0828	-0.259
Disturbance	0.405	-0.4908	-0.065	-0.441	-0.0052	0.114
OC (%)	-0.264	-0.6421	-0.168	0.043	-0.2157	-0.507
EC (mS/cm)	0.084	-0.0075	-0.854	-0.315	0.0867	-0.084
K (ppm)	0.409	-0.3253	-0.535	0.358	0.1195	-0.178
Eigenvalue	0.2753	0.2230	0.1961	0.1716	0.1577	0.13159
Proportion Explained	0.1772	0.1435	0.1262	0.1104	0.1015	0.08468
Cumulative Proportion	0.1772	0.3206	0.4468	0.5572	0.6587	**0.74341**

4.9. Relationship between Community type and Environment variable

There were many environmental variables considered to identify which variables are really determining the pattern of species distribution and formation of the three plant communities. The mean difference in these environmental variables among the three community types was compared using one way ANOVA and Post-Hoc multiple comparison test was performed by using Tukey's HSD. The three community types differ significantly from each other with regard to slope and altitude (Table 16). Thus, the formation of the three plant communities in Biteyu forest significantly responds to slope and altitude among other environmental variables (Table 15).

Table 15. ANOVA table for the mean Difference of Environmental variables among three communiy types.

Variable	$F_{2,27}$	***P*-value**
Altitude	3.976	0.05*
Slope	4.501	0.021*
OC	2.886	0.073
TN	0.444	0.646
EC	0.021	0.979
pH	1.438	0.255
P	1.462	0.250
K	3.26	0.054
C:N	1.713	0.199
Sand	1.117	0.342
Silt	0.764	0.476
Clay	0.325	0.725
Disturbance	2.119	0.140

** Significant at P < 0.05*

4.10. Disturbances in Biteyu forest

As the Biteyu forest is an open accessed, it is usual to find a number of cattle and dead stump as an indicator of human disturbance in the forest (Figure 18). The cattle interference indicates the number of livestock counted in each plot such as sheep, cows, donkeys and oxen in each plot. There was a strong relationship (R^2 = 0.803, $F_{1, 28}$ = 114, P < 0.001) observed between the cattle interference and altitude indicating the number of cattle recorded in each plot decreases with altitude. The mean of cattle interference in the Biteyu forest was 4.77 ± 2.12 per ha.

On the other hand, the numbers of cut wood stumps vary with altitude in each plot. The variation on the cut wood stumps depend on altitude as the topography of Biteyu forest is rugged and undulating which will relatively limit the access to cut the forest. The tree species richness decreases with altitude. This is indicated by the linear regression analysis which showed that there was strong relationship (R^2 = 0.208, $F_{1, 28}$ = 7.37, P < 0.01) between the number of wood stumps and altitudinal gradient eventhough the gradient is short (Appendix V). The mean of wood stump in the Biteyu forest was 26.67±9.37 per ha. This showed the cutting of larger trees in the forest have been a serious threat to the forest and hence leads to the forest degradation.

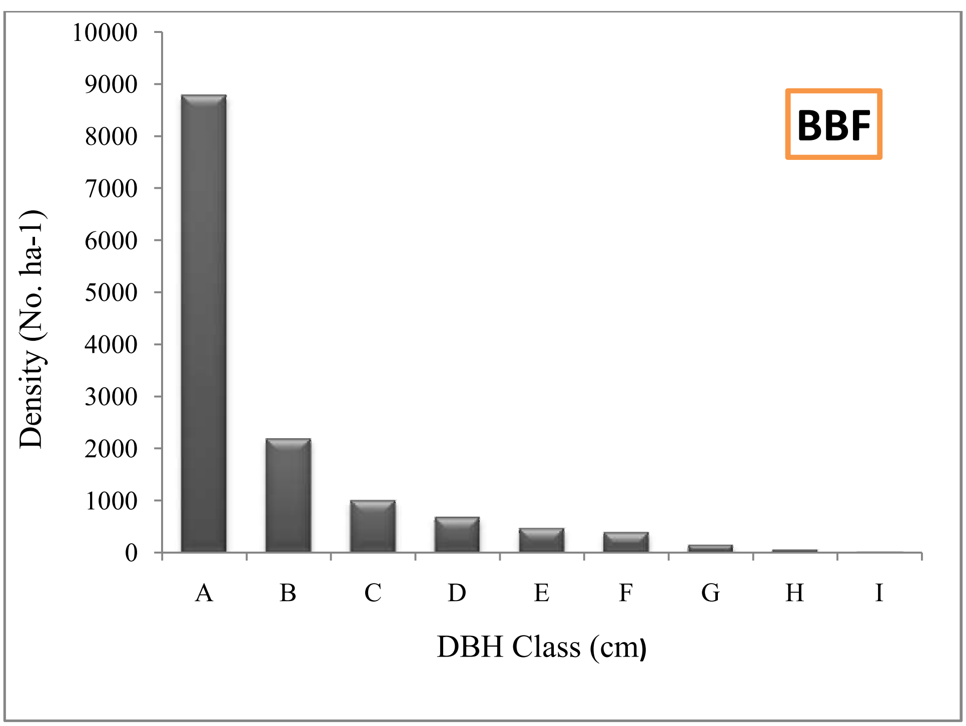
Figure 18.Cutting and Grazing activities in the Biteyu forest

4.11. Structure of Biteyu and Boter-Becho forest

4.11.1. Size class Distribution

The decreasing patterns of stem density across diameter class distribution (Figure 19) were observed in both forests which show the general trends of population dynamics (inverted J shaped curve) but there was an interruption in the last class for Biteyu forest. The analysis showed that the smallest diameter class (2.5-10 cm) in Biteyu and Boter-

Becho forests represented 37.05% and 64.2% of the total stem density respectively. The diameter classes between 10 and 30 cm comprised a stem density of 41.08% in the Biteyu and 23.23% in the Boter-Becho forest. Similarly, the diameter class between 30 and 50 cm contains stem density of 15.76% for Biteyu and 8.27% for Boter-Becho forest. The number of individuals within the largest DBH class (>50 cm) represented only 6.09% in *Biteyu* and 4.3% in Boter-Becho Forest. *Podocarpus falcatus* was a tree species with the largest diameter (198.73 cm) recorded in Biteyu forest and the same tree species was recorded with the largest diameter (181.53 cm) in Boter-Becho forest. The tree species with DBH range greater than 110 cm was contributed only by *Hagenia abyssinica* and *Podocarpus falcatus* in Biteyu Forest and *Syzygium guineense* Subsp. *afromontanum, Olea capensis* subsp. *macrocarpa, Juniperus procera and Podocarpus falcatus* in Boter-Becho forest.



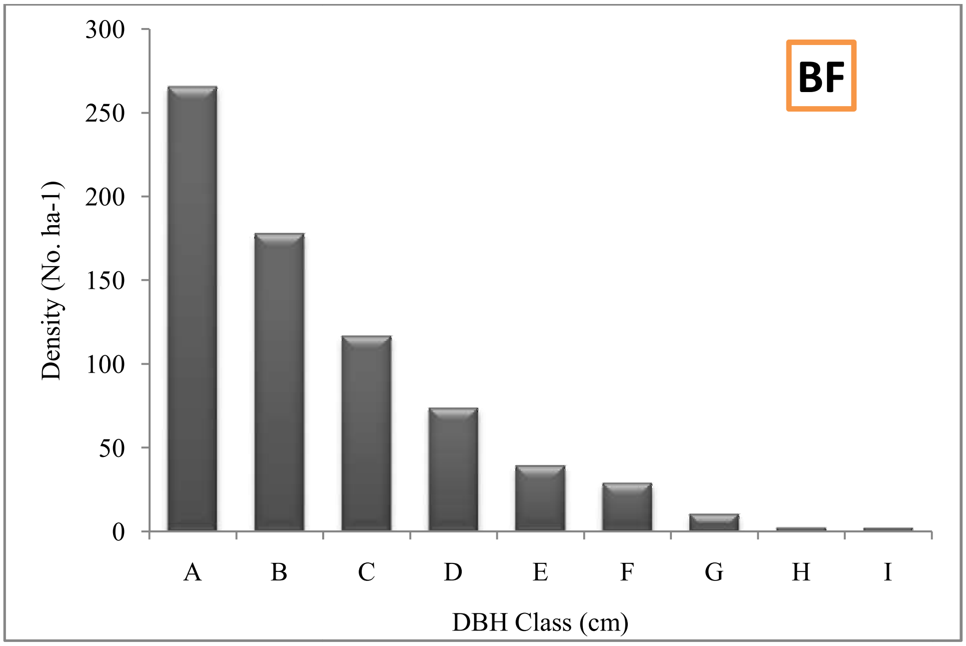
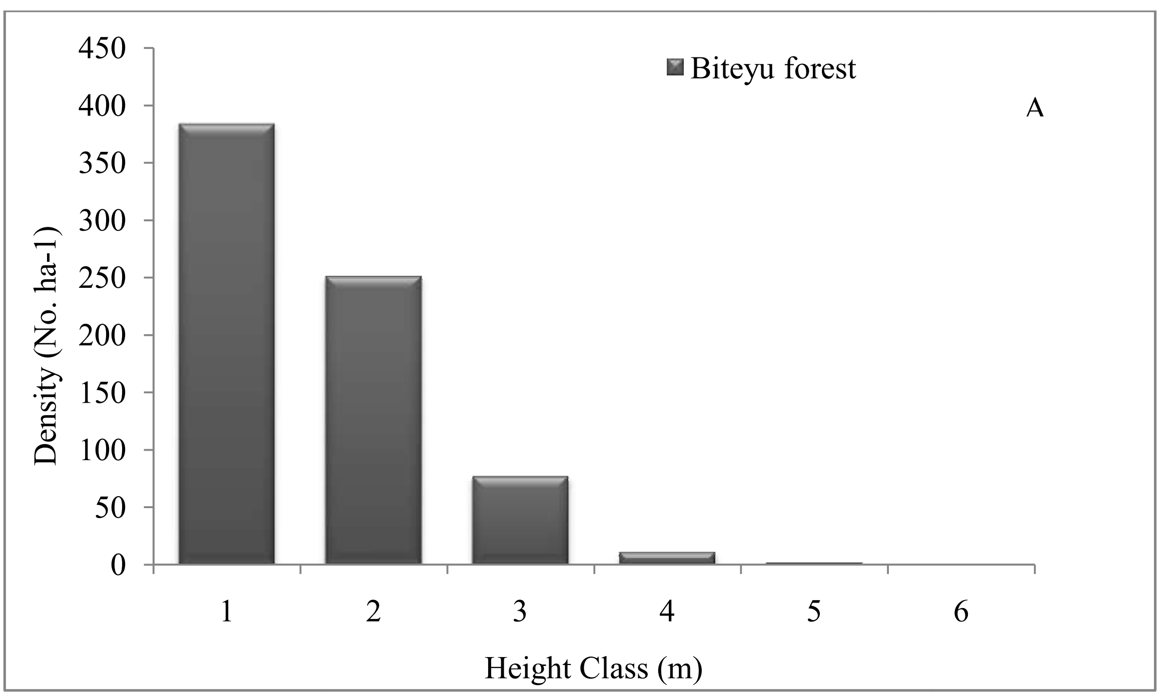


Figure 19.Stem density across different diameter class (Diameter classes: A = 2.5-10 cm, B = 10-20 cm, C = 20-30 cm, D = 30-40 cm, E = 40-50 cm, F = 50-70 cm, G = 70-90, H = 90-110, I ≥ 110).

4.11.2. Height class Distribution

The woody species from both forests were grouped into six major height classes due to the shorter height limit in Biteyu forest. These are 2.0-5.0, 5.0-10, 10.0-15.0, 15.0-20.0, 20.0-25.0 and greater than 25 m (figure 20 A&B). The pattern of height class distribution of the woody species indicates a high proportion of individuals in the lowest height class and a few individuals in the largest height class in both study forests. Consequently, the first height class in Biteyu forest contributed 53.8% (383.70 stems ha^{-}1) followed by the second height class which contributed 34.70% (251.11 stems ha^{-}1) of the total stem density. The third, fourth and fifth height class contributed only 12.30% (88.90 stems ha^{-1}) of the total tree density of the Biteyu forest.

The density of each height class decreases with increase in height and reaching the total absence of trees above 25 m in the Biteyu forest. This may have happened due to the selective cutting of large sized trees for timber and house construction among others. Similar to that of Biteyu forest, the first height class in Boter-Becho forest also contributed 53.55% (1146.64 stems ha^{-1}) followed by the second height class which contributed 36.67% (785.13 stems ha^{-1}) of the total stem density. Moreover, the third, fourth and fifth height class together contributed 9.77% (209.38 stems ha^{-1}) of the total stem density in the Boter-Becho forest. The last height class contributed very small proportion (0.15%) of the total stem density in the Boter-Becho forest, which was represented by only one individual tree, *Olea capensis* subsp. *macrocarpa*. The largest height estimated during the survey is 40 m that was represented by the above-mentioned species. This might indicate that the forest is young and at the state of secondary tropical forest development compared to the highly disturbed and degraded forest with no representative of the height class above 25 m in Biteyu forest.



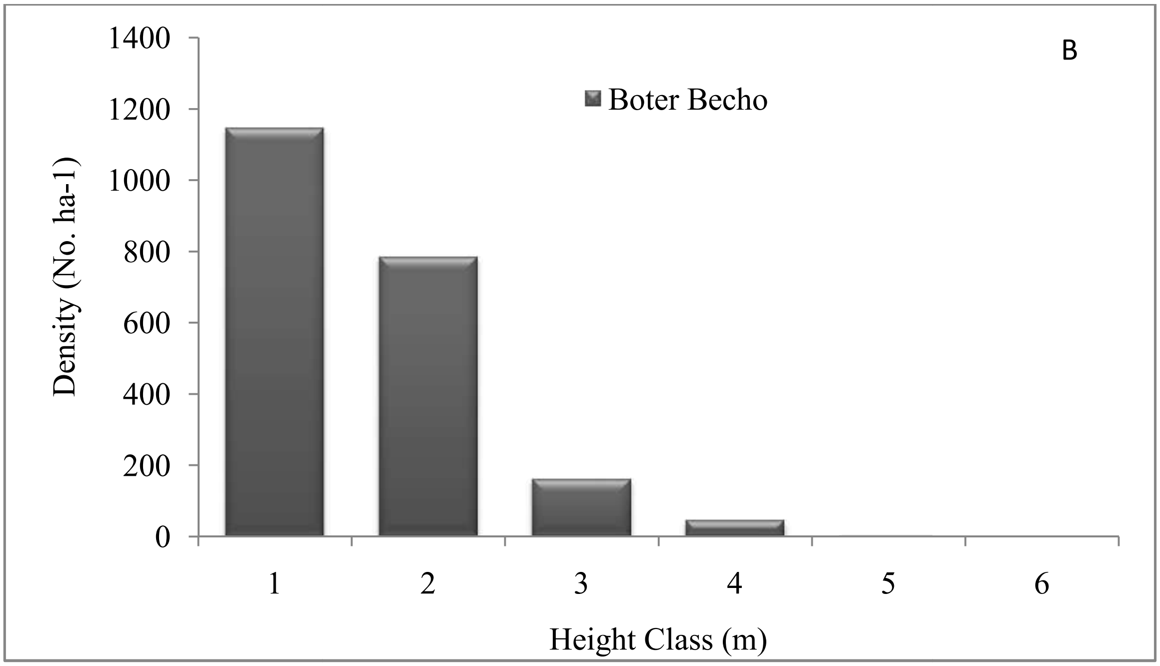


Figure 20.Stem density across height class distribution A/ Biteyu and B/ Boter-Becho (Height class (m), 1=2.5-5.0, 2 = 5.0-10.0, 3 = 10.0-15, 4 = 15.0-20.0, 5 = 20.0-25, 6 = >25)

4.11.3. Basal Area (BA)

Dominance is the basal area (m^2 ha^{-1}) coverage of each individual woody species in the forest. The total basal area of the *Biteyu* forest was 41.64 m^2 ha^{-1} whereas that of Boter-Becho was 72.93 m^2 ha^{-1}. The presence of high density, high frequency and high BA indicate the overall dominance of species of the given forest. Accordingly, *Olinia rochetiana*, *Ilex mitis, Maytenus addat, Maesa lanceolata, Podocarpus falcatus, Schefflera volkensii, Olea europaea* subsp. *cuspidata, Myrsine melanophloeos and Hagenia abyssinica* were the nine most important woody species of the forest contributing to the highest proportion (87.9%) of basal area of the Biteyu forest (Table 16). The considerable contribution of these species to the basal area of the forest is attributed to a/ the higher density than their DBH in individuals of the tree species such as *Olinia rochetiana* and *Maesa lanceolata*. Many of the individuals of these two species have the lowest DBH classes and the basal area contributed by these classes is small. However, the sum of the basal area of individuals of these species is higher than individuals of the remaining seven species mentioned above. b/ intermediate and high DBH classes have relatively small number of individuals per hectare such as *Ilex mitis, Maytenus addat, Podocarpus falcatus, Schefflera volkensii, Olea europaea* Subsp. *cuspidata*, *Myrsine melanophloeos* and *Hagenia abyssinica*.

Table 16. Basal Area and Density of nine most important woody species in Biteyu forest

Species	Basal Area (m^2ha^{-1})	%	Density No ha^{-1}	%
Olinia rochetiana	6.71	16.29	150.74	21.00
Ilex mitis	5.91	14.34	29.26	4.08
Maytenus addat	5.46	13.24	40.37	5.62
Maesa lanceolata	5.42	13.15	116.67	16.25
Podocarpus falcatus	4.79	11.62	63.70	8.88
Schefflera volkensii	3.39	8.22	29.26	4.08
Olea europaea subsp. *Cuspidata*	1.8	4.37	19.63	2.73
Myrsine melanophloeos	1.40	3.41	23.70	3.30
Hagenia abyssinica	1.34	3.26	24.44	3.41
Total	36.24	87.90	497.78	69.35

Unlike Biteyu forest, a total of 18 species account for 93.42% of the total basal area and 57.71% of the total stem density occupied by all the species of the forest in Boter-Becho forest. *Syzygium guineense* subsp. *afromontanum* is the most dominant species, which accounts for 21.76% of the total basal area followed by *Olea capensis* subsp. *macrocarpa*, accounting for 14.87% of the total basal area. However, the stem density of *Olea capensis* subsp. *macrocarpa* is higher than (18.42%) that of *Syzygium guineense* subsp. *afromontanum* (10.77%) in the *Boter-Becho* Forest (Table 17).

Table 17. Tree species with the highest Basal Area in Boter-Becho forest

Species	BA ($m^2\ ha^{-1}$)	%	Density ($No.ha^{-1}$)	%
Syzygium guineense subsp. *Afromontanum*	15.87	21.76	211.42	9.87
Olea capensis subsp. *Macrocarpa*	10.85	14.87	268.23	12.53
Podocarpus falcatus	4.42	6.06	142.56	6.66
Olinia rochetiana	3.63	4.97	56.80	2.65
Ficus sur	4.37	5.99	46.18	2.15
Macaranga capensis	4.06	5.56	53.99	2.52
Croton macrostachyus	3.86	5.29	43.81	2.05
Polyscias fulva	3.30	4.52	19.71	0.92
Pouteria adolfi-friedricii	4.18	5.73	89.67	4.18
Apodytes dimidiata	2.47	3.38	42.87	2.00
Juniperus procera	1.86	2.55	5.47	0.26
Prunus africana	2.22	3.04	28.48	1.33
Ilex mitis	2.24	3.07	57.59	2.69
Allophylus abyssinicus	1.39	1.90	27.54	1.28
Celtis africana	0.85	1.16	15.80	0.74
Hagenia abyssinica	0.86	1.17	10.01	0.47
Milletia ferruginea subsp. *darassana*	0.95	1.30	60.25	2.81
Myrsine melanophloeos	0.76	1.04	55.39	2.58
Total	68.14	93.42%	1235.77	57.71%

4.12. Carbon stocks in the dominant tree species of both forests

Biteyu forest is characterized by having tree species with high density including *Olinia rochetiana, Maesa lanceolata, Podocarpus falcatus and Vernonia rueppellii.* However, the highest biomass carbon was stored in *Ilex mitis* with a carbon stock of 4.92 ± 0.65 and 1.22 ± 0.15 t ha^{-1} in AGB and BGB, respectively. This is attributed to the fact that *Ilex* *mitis* frequently occurred in the study plots and have wider size class compared to its stem density. *Maytenus addat, Olea europaea* subsp. *cuspidata, Podocarpus falcatus, Olinia rochetiana, Myrsine melanophloeos* are among the tree species with higher biomass carbon stock in Biteyu forest (Table 18).

Table 18. Carbon stock ± se (t C ha^{-1}) of above and belowground Biomass of tree species with high IVI values in Biteyu forest.

Species Name	Carbon in AGB	Carbon in BGB
Bersama abyssinica	0.42 ± 0.2	0.076 ± 0.03
Brucea antidysenterica	0.47 ± 0.36	0.2 ± 0.16
Hagenia abyssinica	1.00 ± 0.9	0.43 ± 0.38
Ilex mitis	4.92 ± 0.65	1.22 ± 0.15
Juniperus procera	1.12 ± 0.40	0.32 ± 0.10
Maesa lanceolata	1.00 ± 0.08	0.33 ± 0.025
Maytenus addat	3.96 ± 0.49	0.995 ± 0.108
Myrica salicifolia	1.13 ± 0.37	0.325 ± 0.095
Myrsine melanophloeos	1.15 ± 0.17	0.32 ± 0.045
Nuxia congesta	0.98 ± 0.19	0.29 ± 0.09
Olea europaea subsp. *Cuspidata*	2.60 ± 0.47	0.65 ± 0.114
Olinia rochetiana	1.16 ± 0.09	0.326 ± 0.02
Podocarpus falcatus	1.80 ± 0.84	0.38 ± 0.16
Schefflera volkensii	1.45 ± 0.21	0.415 ± 0.052
Vernonia rueppellii	0.11 ± 0.02	0.04 ± 0.005

Moreover, the simple linear regression analysis between AGB carbon and plot diversity (H) in Biteyu forest showed the positive relationship but not significantly different (R^2 =

0.045, P = 0.263). However, it indicates that slight increase in plot diversity tends to increase the carbon stock of AGB even if the difference is not significant.

Boter-Becho forest is also characterized by tree species with the highest stem density (no/ha) such as *Olea capensis* subsp. *macrocarpa* (381.53), *Syzygium guineense* Subsp. *afromontanum* (223) and *Podocarpus falcatus* (142.10) and *Chionanthus mildbraedii* (94.68) among others. However, *Olinia rochetiana, Syzygium guineense* Subsp. *afromontanum, Ficus sur, Croton macrostachyus, Macaranga capensis, Ilex mitis and Pouteria adolfi-friedricii* are the tree species with the highest average carbon stock in ABG and BGB of *Boter-Becho* forest in descending order (Table 19). There was very weak relationship between mean carbon stock of above and below ground biomass and density of tree species ($F_{1,69}$ = 1.87, R^2 = 0.026, P = 0.176 and $F_{1,69}$ = 1.84, R^2 = 0.026, P = 0.179, respectively) in Boter-Becho forest. The mean difference is also not significant.

Table 19. Carbon stock ± se (t C ha^{-1}) of above and BGB in tree species with highest IVI values in Boter-Becho forest.

Species Name	Carbon in AGB	Carbon in BGB
Olea capensis subsp. *Macrocarpa*	0.87 ± 0.072	0.19 ± 0.013
Syzygium guineense Subsp. *Afromontanum*	2.28 ± 0.18	0.495 ± 0.034
Podocarpus falcatus	0.99 ± 0.17	0.24 ± 0.03
Chionanthus mildbraedii	0.09 ± 0.012	0.04 ± 0.0035
Pouteria adolfi-friedricii	1.06 ± 0.12	0.27 ± 0.03
Psychotria orophilla	0.019 ± 0.0011	0.01 ± 0.00086
Erica arborea	0.02 ± 0.001	0.0096 ± 0.0005
Olinia rochetiana	2.55 ± 0.32	0.56 ± 0.06
Myrsine melanophloeos	0.37 ± 0.04	0.113 ± 0.012

Milletia ferruginea Subsp. *darassana*	0.45 ± 0.06	0.137 ± 0.015
Macaranga capensis	1.68 ± 0.15	0.42 ± 0.033
Bersama abyssinca	0.097 ± 0.02	0.034 ± 0.0054
Ficus sur	2.22 ± 0.21	0.567 ± 0.05
Croton macrostachyus	2.0 ± 0.133	0.523 ± 0.032
Teclea nobilis	0.065 ± .009	0.025 ± 0.003
Ilex mitis	1.07 ± 0.13	0.292 ± 0.04

4.13. Carbon storage in different DBH classes

4.13.1. Carbon Storage in AGB across different DBH classes

The carbon stock in the AGB increases with DBH (Figure 21). The smallest carbon stock was estimated in the DBH class of 2.5-10 and 10.01-20.0 cm despite its high density. The highest AGB carbon was estimated in the DBH > 50 cm for both forests. In Biteyu forest, the smaller size classes (DBH < 20 cm) held 61.86% of the total stem density but a very small fraction (1.6%) of the live AGB carbon density. On the other hand, the smaller size classes held 80.14% of the total stem density and 1.5% of the live AGB carbon in Boter-Becho forest. Tree species with DBH>50 cm comprised of *Podocarpus falcatus*, *Ilex mitis, Hagenia abyssinica, Juniperus procera and Maytenus addat* among others held 58.23% of the average carbon stock in AGB of the Biteyu forest. Similarly, in Botor-Becho forest, the tree species with DBH > 50 cm accounted for 61.6% of the average carbon stock in above ground biomass. Those tree species included in this size class in Boter-Becho include *Allophylus abyssinicus, Apodytes dimidiata, Ficus sur, Hagenia abyssinica, Ilex mitis, Juniperus procera, Macaranga capensis, Olea capensis* subsp. *macrocarpa, Olinia rochetiana, Croton macrostachyus, Podocarpus falcatus, Pouteria adolfi-friedricii, Syzygium guineense* Subsp. *afromontanum, Celtis africana* among others. In Biteyu forest, only individuals with DBH > 90 cm comprised of five species *Podocarpus falcatus*, *Ilex mitis, Hagenia abyssinica, Juniperus procera and Maytenus addat* accounting for more than 75% of the average carbon stock in AGB of the Biteyu forest. Only 25% or less of the average carbon stock in AGB is comprised by all the plant species with a DBH< 90 cm in Biteyu forest. This indicates that the forest is nearly devoid of the larger trees because of the selective logging of trees for fuel, timber, and construction purpose by the nearby community as confirmed also by observation and the personal communication by the elders of the village in the study area.

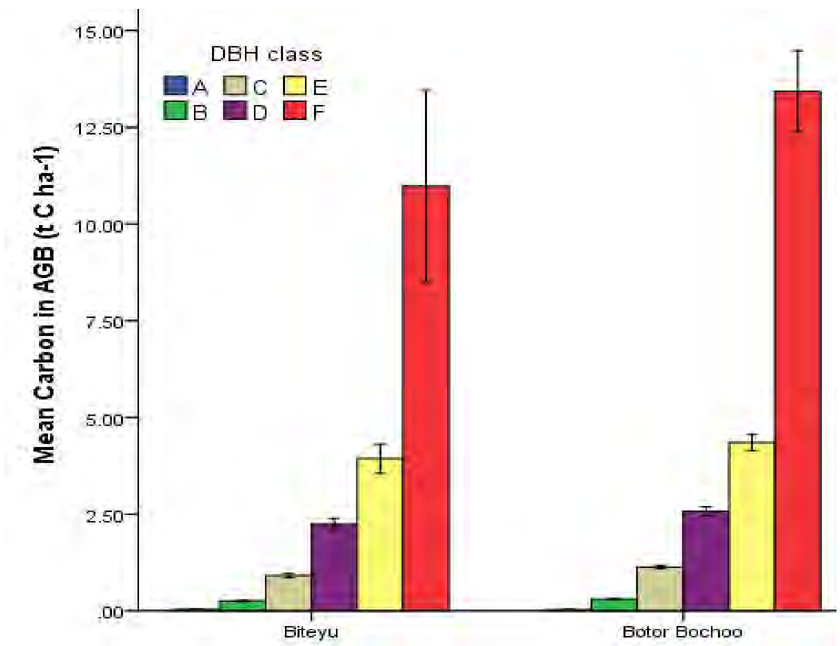


Figure 21.Mean carbon stock in AGB of Biteyu Forest and Boter-Becho Forest (A = 2.5-10, B = 10.0-20, C = 20.0-30, D = 30.0-40, E = 40.0-50, F = >50).

4.13.2. Carbon stock in BGB across different DBH classes of both forests

Since the carbon stock of BGB was estimated from the AGB as recommended by Cairns *et al*. (1997) for tropical forests, the carbon stored increases with DBH in both Biteyu Forest and *Boter-Becho* forest. Accordingly, the smallest carbon store was observed in

DBH range of 2.5-10 cm (0.029 ± 0.0085 t C ha^{-1}) for Biteyu Forest and 0.014 ± 0.0002 t C ha^{-1} for Boter-Becho Forest. The carbon stock estimated in BGB between 40 and 50 cm was 1.06 ± 0.53 t C ha^{-1} in Biteyu Forest and 1.035 ± 0.025 t C ha^{-1} in Boter-Becho Forest. The highest carbon stock was estimated in DBH > 50 cm; 2.35 ± 0.20 for Biteyu Forest and 2.715± 0.098 t C ha^{-1} for Boter-Becho Forest. The highest carbon stock in BGB was estimated in the DBH class of > 50 cm (Figure 22).

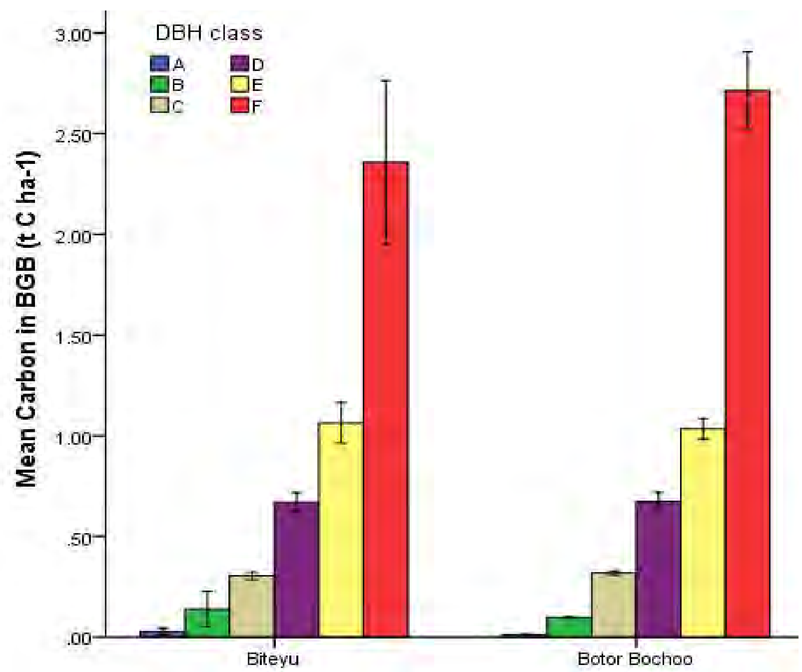


Figure 22. Mean carbon stock in BGB of Biteyu Forest and Boter-Becho Forest (A=2.5-10, B = 10.01-20, C = 20.01-30, D = 30.01-40, E = 40.01-50, F = >50).

4.13.3. Carbon storage in different pools of Biteyu and Boter-Becho forests

The estimates of total carbon stock in Biteyu and Boter-Becho forest was 166.67 ± 16.4 and 393 ± 24 t ha^{-1} respectively (Table 20). The Boter-Becho forest contains carbon stock twice as much as that of Biteyu forest does. This was because the Boter-Becho forest is a regionally protected forest priority area in which the disturbance level is low compared to the open accessed forest of Biteyu in Gurage mountain chain. The estimated carbon stock in AGB and BGB in Biteyu forest was 87.13 ± 11.80 t ha^{-1} and 22.94 ± 2.84 t ha^{-1}, respectively. In Biteyu forest, the plot level variation of carbon in AGB ranges from 5.69 to 257 t ha^{-1}. In the case of soil C, the plot level variation ranges from 36.31 to 72.31 t ha^{-1} in Biteyu forest (Appendix V).

In Boter-Becho forest, the carbon stock in AGB and BGB was 189.4 ± 11.83 and 43.98 ± 2.28 t ha^{-1}, respectively. The plot level variation of the carbon stock in AGB ranges from 8 to 495 t ha^{-1} in Boter-Becho Forest depending up on the variation in DBH and abundance of tree and shrub species. Estimates of soil carbon in Boter-Becho Forest was about 159.31 ± 9.86 t ha^{-1} which is nearly three times as much as the soil carbon stock (56.37 ± 1.73 t ha^{-1}) of Biteyu Forest. The plot level variation of soil carbon ranges from 95 to 495.76 t ha^{-1} (Appendix VI) in Boter-Becho Forest. This may be attributed to the type of soil texture, structure and soil bulk density. Moreover, the estimates of litter carbon in both forests are the smallest compared to other carbon pools containing 0.26 ± 0.011 in Biteyu forest and 0.30 ± 0.014 in Boter-Becho forest.

AGB contains the largest fraction of total carbon stock, accounting for 52.30% in the Biteyu Forest and 48.20% in the Boter-Becho forest. Soil carbon comprises the second largest fraction of the total carbon stock, accounting for 33.81% in the Biteyu Forest and 40.54% in the Boter-Becho forest. Thirdly, the carbon stock stored in the BGB accounts for 13.76% in the Biteyu Forest and 11.20% in the Boter-Becho forest. Litter carbon contains the smallest fraction of the total carbon stock, accounting for 0.16% in the Bitey Forest and 0.08% in the Boter-Becho Forest. One-way analysis of variance showed a significant difference in mean carbon stocks of all carbon pools at $P < 0.001$ except litter carbon ($P > 0.05$) between the two forests (Table 20).

Table 20. ANOVA table for the mean differences in carbon stocks between two forests

Carbon pool	Biteyu Forest	Boter-Becho Forest	*F*-value	*P*-value
Carbon in AGB	87.1 ± 11.80	189.4 ± 11.83	26.747	< 0.001**
Carbon in BGB	22.94 ± 2.84	43.98 ± 2.28	28.106	< 0.001**
Soil Carbon	56.37 ± 1.73	159.3 ± 9.86	45.508	< 0.001**
Litter carbon	0.26 ± 0.01	0.30 ± 0.014	3.829	0.053^{ns}
Total	166.67 ± 16.4	393 ± 24		

**Significant at P < 0.001, ns = not significant

The carbon sequestration potential of Biteyu Forest was about 611.8 ± 60.2 t CO_2eq ha^{-1} and that of Boter-Becho Forest is about 1442.31 ± 88.08 t CO_2eq ha^{-1} (Table 21). With the existing disturbance situation, the carbon sequestration potential of Biteyu forest would be 305,895 ± 30,100 CO_2 eq. Similarly, the carbon sequestration potential of Boter-Becho forest would be 45,576,996 ± 2,783,328 CO_2 eq. Boter-Becho forest has a high carbon sequestration potential, which would generate much more money from the carbon trade, mainly REDD+ initiative, as the forest is relatively protected and managed compared to highly degraded dry forest of Biteyu in Gurage mountain chain.

Table 21. The carbon sequestration potential (Mean ± SE) of the two forests

The unit	Biteyu Forest	Boter-Becho Forest
t C ha^{-1}	166.67 ± 16.4	393 ± 24
t CO_2 eq ha^{-1}	611.8 ± 60.2	1442.31 ± 88.08
t CO_2 eq in total forest area	305,895 ± 30,100	45,576,996 ± 2,783,328

4.14. The Effect of altitude and slope on carbon pool of both forests

The effect of altitude and slope for both dry (Biteyu forest) and moist evergreen montane forest (Boter-Becho) on different carbon pool was determined. A positive but very weak relationship was observed between altitudinal gradient and AGBC in Boter-Becho forest. The linear regression result showed that the change of altitudinal gradeint has a significant effect ($F_{1,69}$ = 4.33, R^2 = 0.059, $P < 0.05$) on the AGBC despite its weak relationship (Figure 23A). The carbon stocks in the soil ranged between 551 t ha^{-1} in 2356 m and 75.89 t ha^{-1} even in lesser elevation of 2323 m above sea level in Boter-Becho forest. The relationship between altitude and SOC at the plot level in the forest were not significant. The linear regression analysis showed that there was no significant ($F_{1, 69}$ = 0.892, R^2= 0.013, P = 0.348) elevational trend observed in the SOC in the forest system (Figure 23B). Similarly, the altitude does not tell us any information about the trend of litter carbon along the gradient in the Boter- Becho forest even though its slope is positive in the regression (P = 0.82) (Figure 23C).

In contrast to altitude, slope showed a very weak positive relationship with all response variables. However, there is no strong evidence to conclude that steepy/slopy nature of the topography is an important predictor for AGBC (P = 0.416), soil carbon (P = 0.44) and litter carbon (P = 0.132) in the moist evergreen montane forest (Boter-Bocho forest). The difference is not significant ($F_{1,69}$ = 2.327, P = 0.132)(Figure 23, D-F).

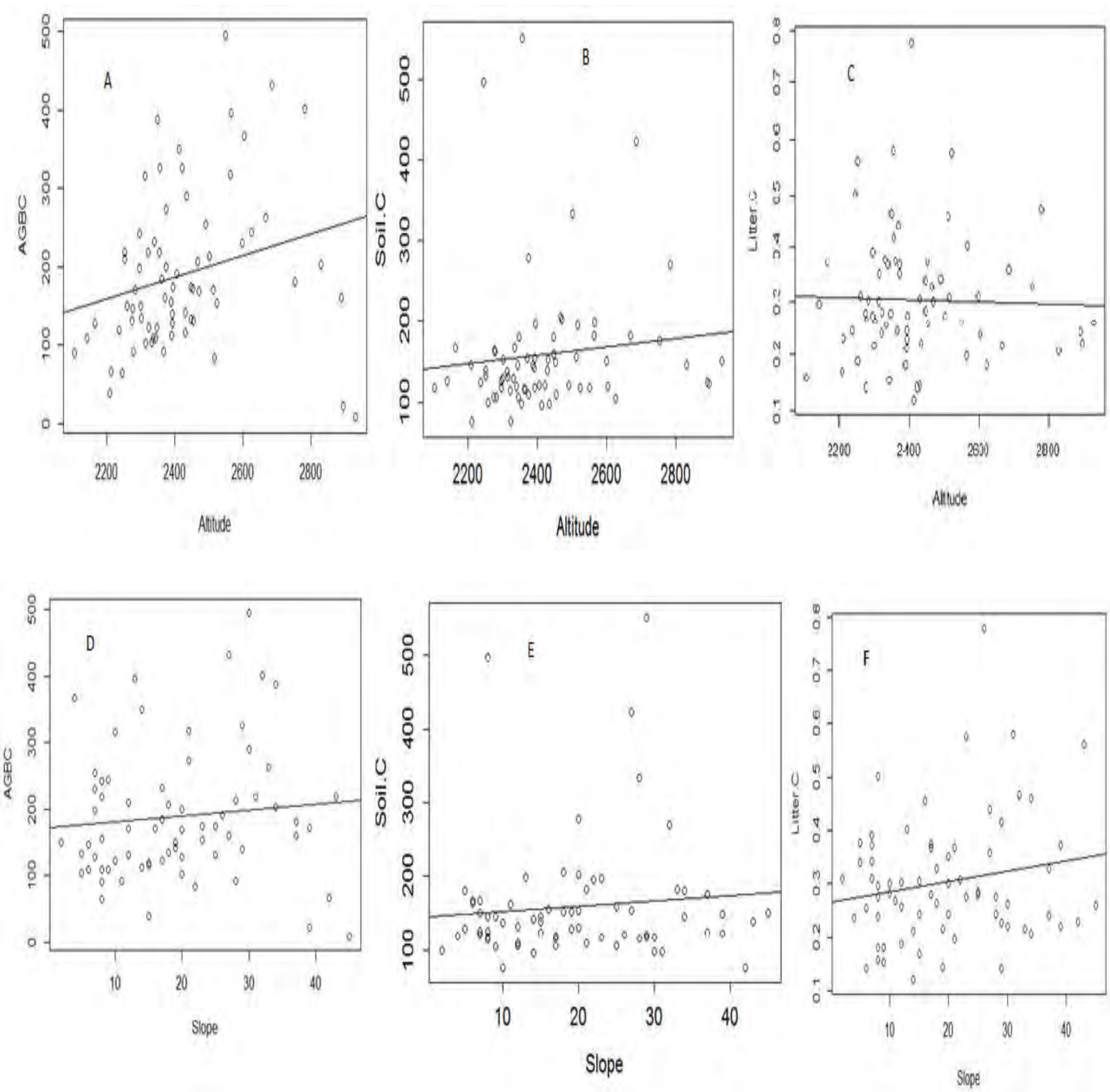
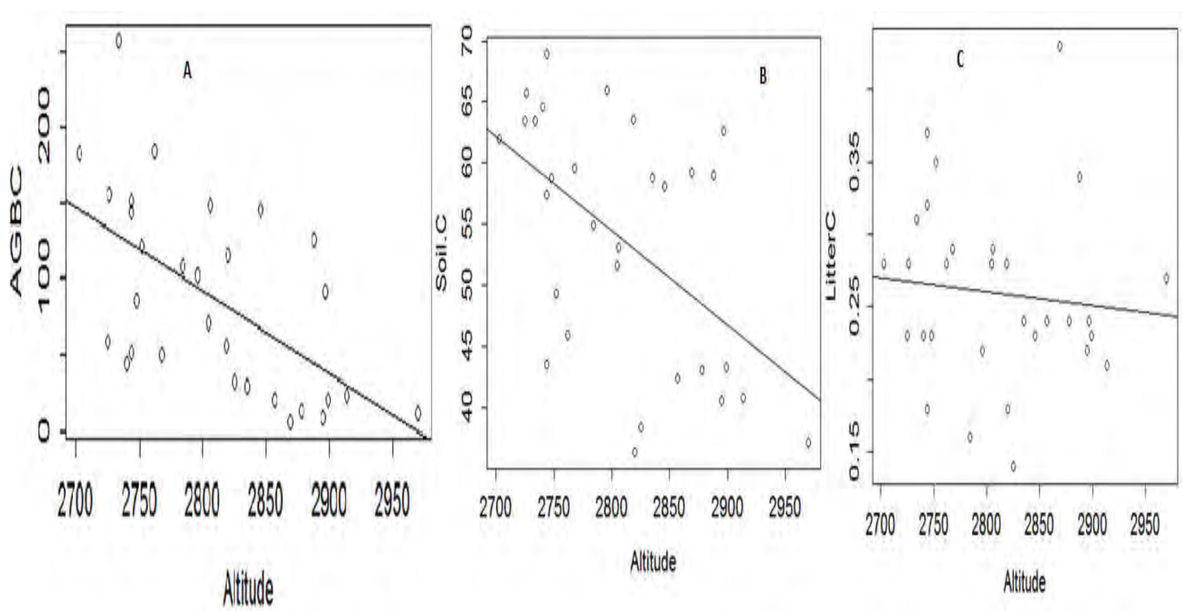


Figure 23.The carbon stocks against altitude (A-C) and slope (D-E) in Boter- Becho forest.

Linear regression analysis showed that there was a negative relationship between AGBC and altitudinal gradient for the Biteyu forest (Figure 24, A-C.). Even if the altitudinal gradient is very short, the effect of altitudinal gradient on the carbon stock of the AGB is significant (R^2 = 0.34, $F_{1, 28}$ = 14.59, P < 0.001). Similarly, the effect of altitude on the BGBC variation does the same as that of AGBC since its value was calculated from the

AGB. Moreover, the effect of the altitudinal variation on the soil carbon is significant ($F_{1, 28}$= 10.9, R^2 = 0.28, P < 0.001) and it shows a strong positive relationship with altitude in Biteyu forest. The linear regression analysis also showed that there is no effect of altitude on the litter carbon in Biteyu forest ($F_{1, 28}$ = 0.299, R^2 = 0.01, P = 0.589). In contrast to the significant effect of altitude on AGBC, slope in Biteyu forest tends to show the negative relationship but not significantly (R^2 = 0.035, P = 0.318) affecting the variation in AGB carbon (Figure 24D). The slope was also very weak and negatively related to the soil carbon giving no evidence of its effect on the soil carbon in Biteyu forest ($F_{1, 28}$ = 3.67, R^2 = 0.1141, p = 0.067) (Figure 24E). Despite the positive and very weak relationship existing between slope and the litter carbon stock, the effect of slope on litter carbon is not significant in Biteyu forest ($F_{1, 28}$ = 3.67, R^2 = 0.018, P = 0.46) (Figure 24F).



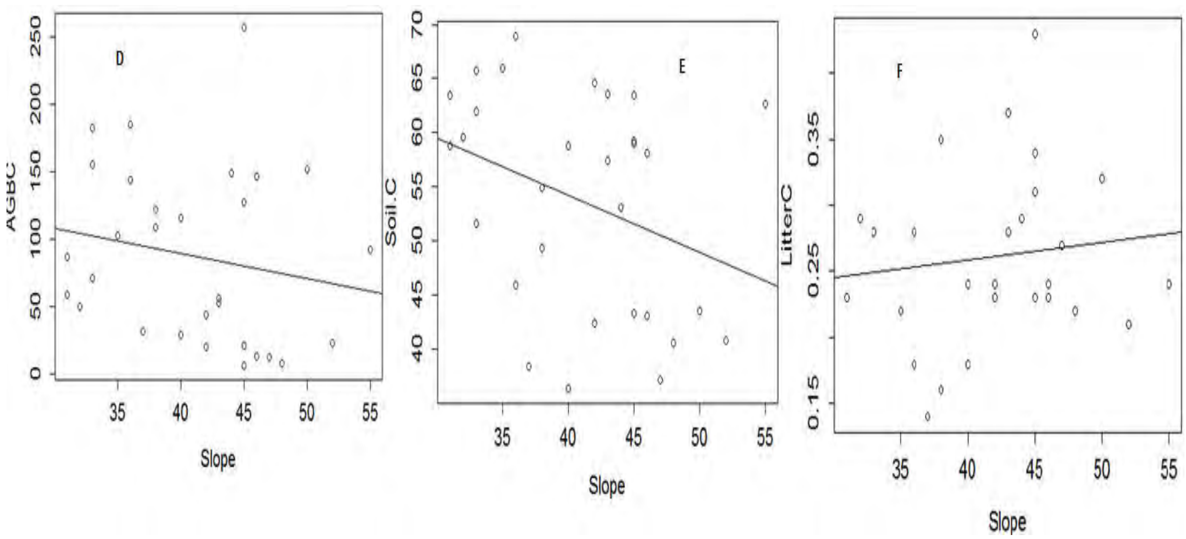


Figure 24. The carbon stocks against altitude (A-C) and slope (D-E) in Biteyu Forest.

4.15. Litterfall Production in Boter-Becho forest

The mean annual dry mass of litterfall in 365 days were 10.76 t $ha^{-1}yr^{-1}$ and 6.64 t $ha^{-1}yr^{-1}$ in LD and HD sites, respectively (Figure 25). The mean difference between the two sites was found significant due to the higher variation in the amounts in the LD site though the lower litter production in highly disturbed site was evident. The difference for litterfall in both sites indicate that the disturbance has a significant effect as can be observed in the Table 24. The mean dry weight of the total litterfall of the Boter-Becho Forest is the average of the two sites, which is equal to 8.7 t ha^{-1} yr^{-1} (Appendix VII).

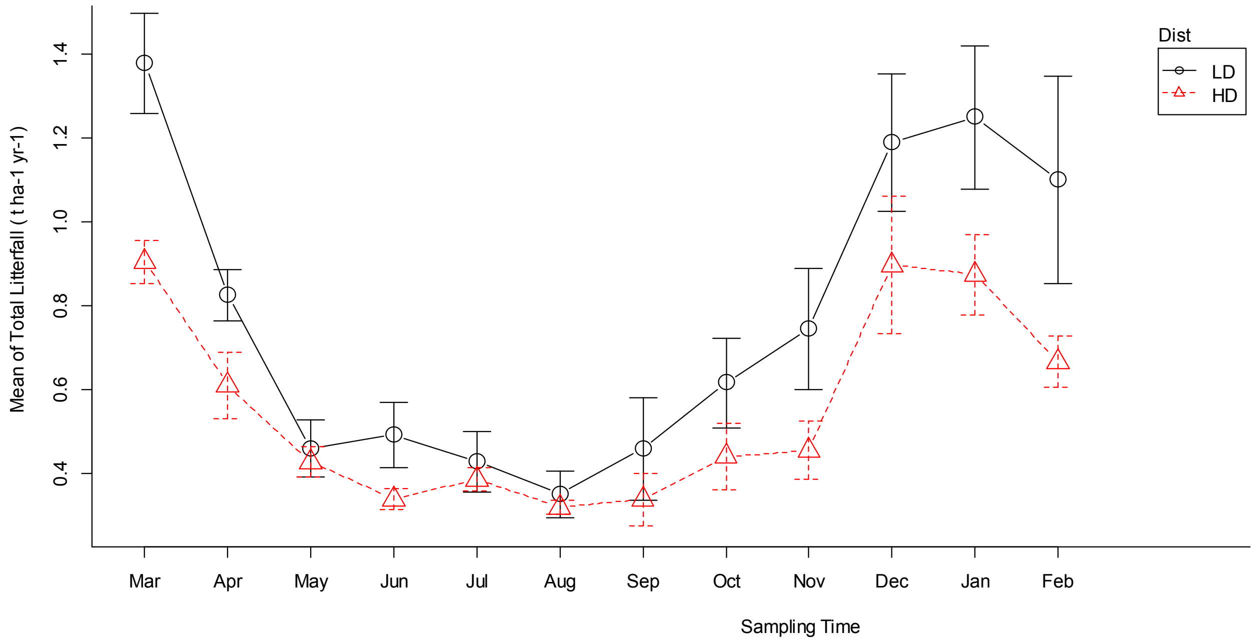


Figure 25.Total litterfall in LD (dotted line) and HD (solid line) sites (n = 5 for each points). Error bars indicate standard error (Se) from the mean

Among the litter fractions, leaf litter comprised 42.23% and 49.5% of the total annual litterfall in LD and HD sites of the forest, respectively. Non-leaf litterfall fractions including twigs, reproductive parts, and others comprised 57.78% and 50.5% of the total litterfall in LD and HD sites, respectively.

The percentage contribution of leaf litter was highest (45%) followed by reproductive parts (24.13%), other (15.92%), and twigs (14.95%) of the total litterfall in the forest (Table 25). The high value of reproductive parts as fraction of litterfall was recorded in March, April, December, January and February in both LD and HD sites of the forest, which might indicate the flowering and fruiting seasons of some of the forest species. A significant difference in the mean of dry weight of litterfall was observed between LD and HD sites of the forest (Table 22).

Table 22. ANOVA table for the mean litterfall fractions (t ha^{-1}) between LD and HD Sites

Litterfall	LD	HD	$F_{1,118}$	*P*-value
Total litterfall	0.77 ± 0.06	0.55 ± 0.03	11.83	<0.001**
Leaf	0.38 ± 0.03	0.27 ± 0.02	8.78	0.004**
Reproductive parts	0.23 ± 0.02	0.13 ± 0.008	26.04	<0.001**
Twigs	0.13 ± 0.01	0.08 ± 0.006	17.12	<0.001**
Other	0.10 ± 0.008	0.07 ± 0.004	8.44	0.004**

*Note: ** significant at $P \leq 0.01$*

However, the seasonal variations affect the litterfall where the wet season inhibits and the dry season facilitates the litterfall in the forest ecosystem. Total litterfall in both sites followed multimodal distribution pattern in which litterfall peaks occurred in March, April, December, January and February. The two major peaks of litterfall were observed in March and January (Figure 25) where March is from the onset of wet season and January is from the dry season. The fractions and the total litterfall showed significant difference between wet and dry season in the forest (Table 23). From this, one can conclude that both disturbance and seasonal variations have a significant effect on the amount of the litter fractions and total litterfall (Table 22 and 23).

Table 23. ANOVA table for the mean litterfall fractions (t ha^{-1}) between wet and dry season

Types of Litterfall	Wet season	Dry season	$F_{1,118}$	*P*-value
Total litterfall	0.55 ± 0.04	0.82 ± 0.05	17.09	< 0.001**
Leaf	0.28 ± 0.02	0.39 ± 0.03	10.983	0.001**
Reproductive parts	0.15 ± 0.013	0.22 ± 0.015	11.393	0.001**
Twigs	0.08 ± 0.006	0.14 ± 0.01	23.79	< 0.001**
Other	0.07 ± 0.004	0.10 ± 0.008	19.144	< 0.001**

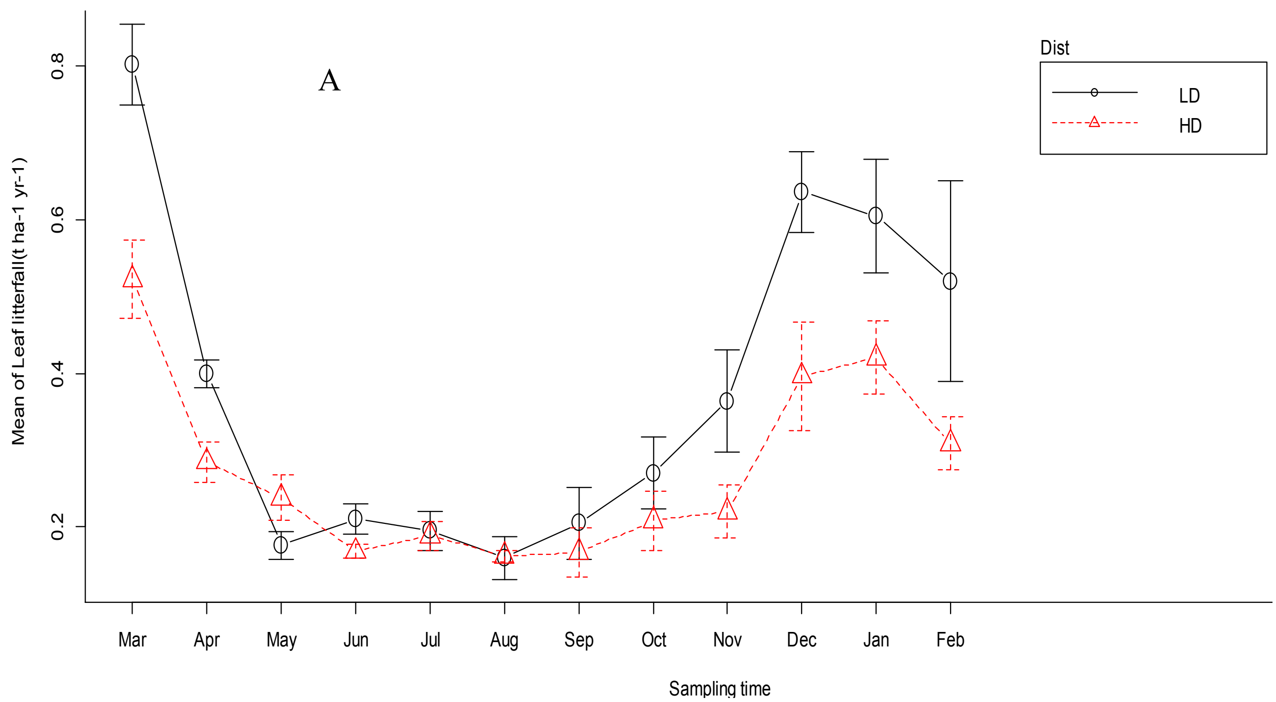
*Note: ** significant at $P \leq 0.01$, n = 70 (for wet season) and 50 (for dry season)*

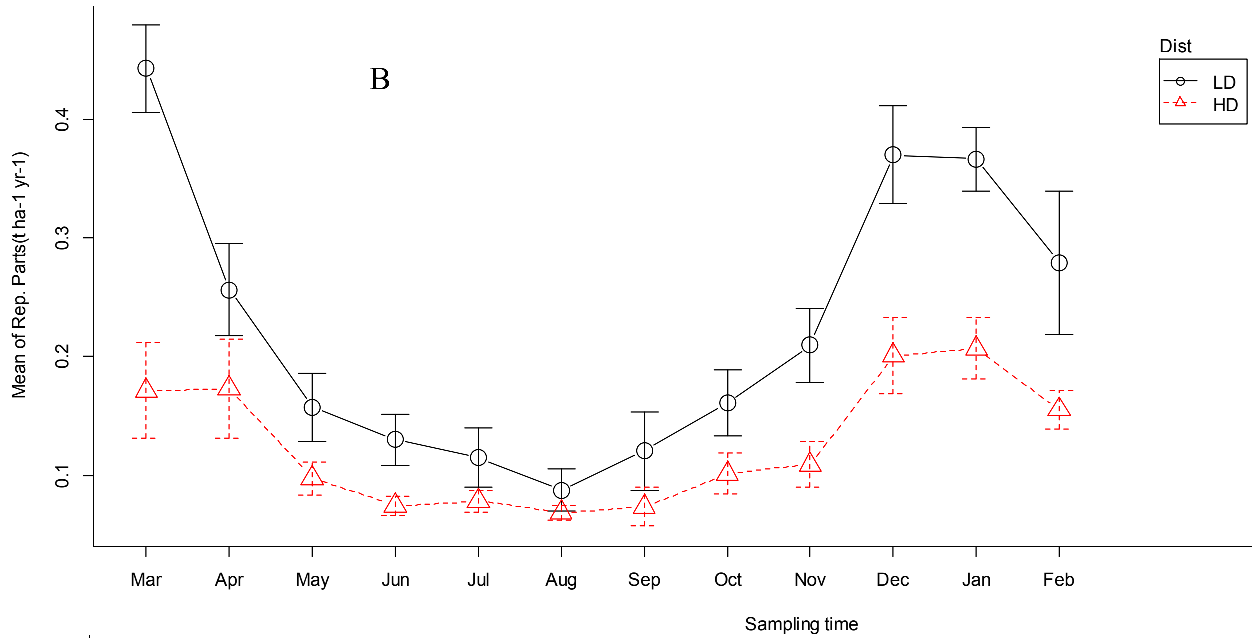
Here, the highly disturbed sites contain forest species with small leaves including *Albizia gummifera, Celtis africana, Podocarpus falcatus* and other similar species. There is also a significant difference observed between means total litterfall in wet (n = 35) and dry season (n = 25) in LD site of the forest ($F_{1,\ 58}$ = 10.197, $P < 0.01$). Similarly, significant mean difference ($F_{1,\ 58}$ = 8.482, $P < 0.01$) of total litterfall was observed in between wet and dry season in HD sites. The amount of litterfall and its components in the two sites were given in Table 24 and all components in the HD sites are smaller in quantity than LD sites.

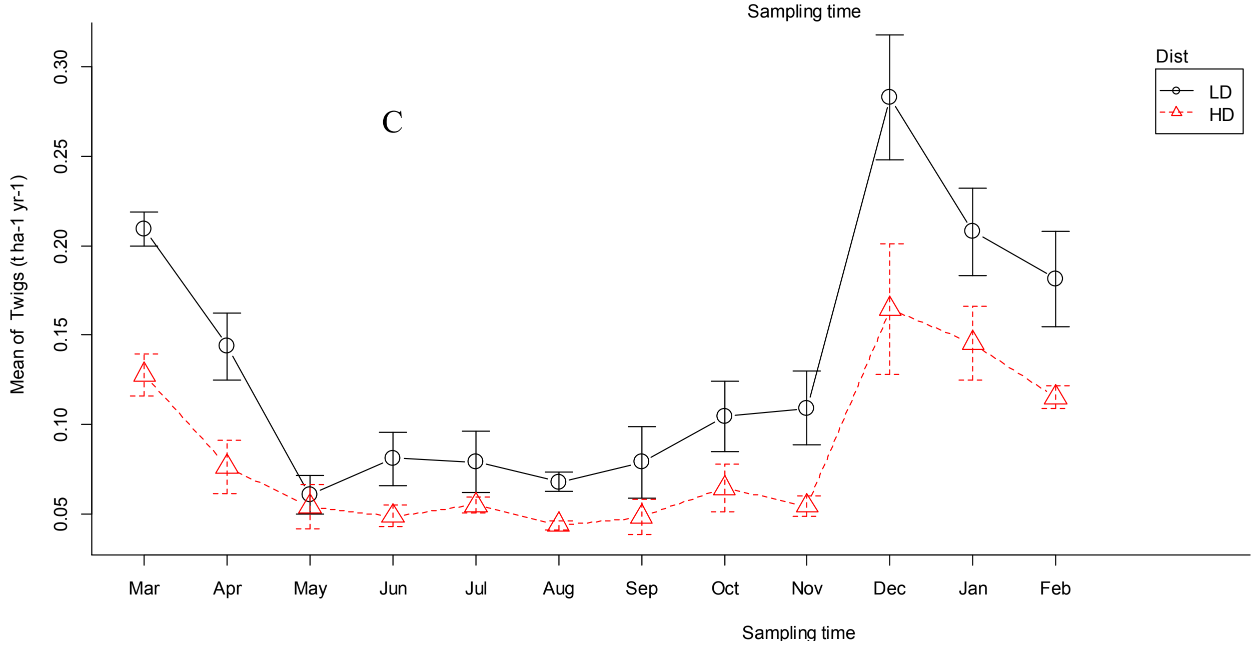
Table 24. Annual litterfall (t ha^{-1} yr^{-1}) in the study forest

	Litterfall (t $ha^{-1}yr^{-1}$)					
Sites	leaves	Rep. parts	twigs	other	total	sample
LD	4.54	2.69	1.60	1.92	10.76	60
HD	3.28	1.51	1.00	0.85	6.64	60
Total	7.83	4.20	2.60	2.77	17.40	120
Percent	45.00	24.13	14.95	15.92	100.00	

During litterfall collection, it was observed that the bottom of some of the litter trap in both sites of the forest has a big hole inside and some of the materials were lost up to the time I maintained it. Due to this, the values of litterfall might be underestimated by 5.0% at LD sites and up to 10% at HD sites of the forest. The dry weight values of litterfall would have been higher than this if the litter trap interruption had not happened during the course of the study. The mean monthly variation of litterfall and its fractions in LD and HD sites are given in Fig. 26.




B
Dist
LD
HD
Mean of Rep. Parts(t ha-1 yr-1)
0.1
0.2
0.3
0.4
Mar
Apr
May
Jun
Jul
Aug
Sep
Oct
Nov
Dec
Jan
Feb
Sampling time


C
Dist
LD
HD
Mean of Twigs (t ha-1 yr-1)
0.05
0.10
0.15
0.20
0.25
0.30
Mar
Apr
May
Jun
Jul
Aug
Sep
Oct
Nov
Dec
Jan
Feb
Sampling time

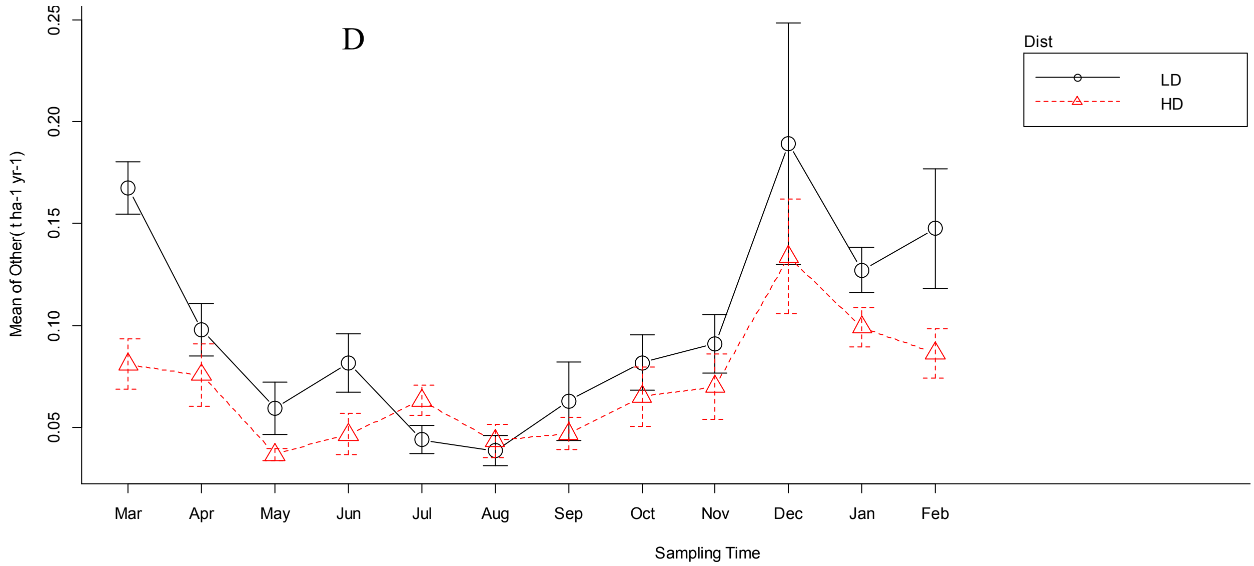


Figure 26.Mean ± se of dry mass (t ha^{-1}) of litterfall fractions (leaf (A), Reproductive parts(B), Twigs(C) and "other"(D)) in LD and HD sites (n=5 for each value).

4.16. The effect of climatic variables on litterfall in Boter-Becho forest

Since the same climate data was collected from the only existing one station for both LD and HD sites, the effect of climate variables (rainfall and temperature) on the total litterfall with its fractions was observed for the whole forest system. The effect of mean monthly temperature and rainfall on the litterfall fraction and total litterfall in the forest was found significant (Tables 25 and 26). The multiple linear regression model explained 30.4% of the variation in total litterfall in the forest. However, from the analysis rainfall has a significant ($P < 0.001$) effect on the amount of litterfall in the forest compared to temperature. The amount of total litterfall decreases with increasing rainfall. Similarly, the relationship between mean monthly temperature and total litterfall is also significant ($P < 0.05$) which brings variation in total litterfall as well as fractions of litterfall.

The difference in quantity of litter production is not only the shift from one season to the other but also the interactions of disturbance level, seasons and sampling time affects the phenological characteristics of forest species. The two way ANOVA showed a significant interaction effects of disturbance* season (F = 4.1, P = 0.045) and sampling time * season (F = 14.194, P < 0.001) on the total litterfall. Moreover, the relationship between mean monthly rainfall and temperature and the amount of litterfall fractions (Table 25 and 26) were analyzed using simple linear regression.

Table 25. Relationship between monthly rainfall and the amount of litterall fractions

Litter	R^2	$F_{1,118}$	P value
Leaf	0.298	50.2	< 0.001**
Reproductive parts	0.266	42.87	< 0.001**
Twigs	0.296	49.54	< 0.001**
Other	0.244	38.11	< 0.001**
Total	0.318	55.05	< 0.001**

*** significant at P< 0.01*

Table 26. Relationship between monthly temperature and the amount of litterall fractions

Litter	R^2	$F_{1,118}$	P value
Leaf	0.086	11.073	< 0.001**
Reproductive parts	0.067	8.443	0.004*
Twigs	0.006	0.712	0.400^{ns}
Other	0.005	0.576	0.449^{ns}
Total	0.061	5.716	0.018*

*Note: ** significance level at P < 0.01, ns=not significant; * significance level at P <0.05*

4.17. Decomposition Pattern of leaf litter in Boter-Becho forest

Mass loss rates in all litterbags were so rapid but highly variable between decomposing periods. Leaf litter decomposition was fast in the first three months of decomposing period and eventually decreases for both sites. The residual mass of leaf litter (mean ± Se, n=70; 35 litter bags for each sites) in the first seven months (March to september), where there was optimum rainfall, was 11.81 ± 0.50 g and 11.83 ± 0.41g for LD and HD sites respectively. Analysis by the independent sample t-test showed that the mean difference in dry mass of decomposing leaf litter in the forest was not significant ($t = 0.025$, $p > 0.05$, $df = 68$) between LD and HD sites in the first seven months. Moreover, the residual mass of leaf litter (mean ± Se, n = 50; 25 litter bags for each sites) at the later five months in both LD and HD sites of the forest are 6.07 ± 0.36 g and 5.48 ± 0.30 g respectively and the difference is still not significant ($t = 1.246$, $P > 0.05$, $df = 48$).

In addition with litter decomposition in the forest, the mean of soil moisture (%) recorded during the experimental period between the two sites showed significant difference ($t = 7.96$, $P < 0.001$, $df = 118$). Similarly, the mean value of soil pH measured during the decomposition experiment between the two sites was also showed a significant difference ($t = 4.13$, $P < 0.001$, $df =118$).

In contrast to the monthly variation in leaf litter decomposition, it was found that there is no significant difference ($t = 0.34$, $P > 0.05$, $df = 118$) noted in residual leaf litter between LD and HD sites of the forest showing no observed significant effect of the local disturbance on litter decomposition. From the experiment, almost similar proportion, 40.92% and 40.82% of the initial dry mass of leaf litter was disappeared in the first seven months in LD and HD sites, respectively. Similarly, about 25.10% and 25.90% of the initial dry mass of leaf litter was decomposed in the later five months in LD and HD sites, respectively. Here, the mean of the leaf litter dry mass collected in the October was used as the initial mass to calculate the percentage of leaf litter decomposed during the later five months. The proportion of initial dry mass disappeared in the later five months was smaller than that of the first seven months. This may be because of two possible reasons; the less soil moisture availability in the later five months, dry season, and this season is at the later stage of leaf litter decomposition leaving the recalcitrant materials, as the experiment was not separate for both seasons. This means that the rapid rate of leaf litter decomposition was observed in the first seven months compared to the five months of the dry season.

At the end of 365 days of the experiment period, 66.02% and 66.72% of the forest leaf litter was decomposed in LD and HD sites respectively. The possible reason for the initial fast rate of leaf litter decomposition might be due to the optimum rainfall that the forest gets at the wet season. Actually, the seasonal rainfall provides moisture to the soil, which becomes an important regulator of litterfall and rate of decomposition in the *Boter-Becho* forests. The following graph shows the mean ± SE of residual leaf litter from the initial dry mass of the two sites of *Boter- Becho* forest (Figure 27).

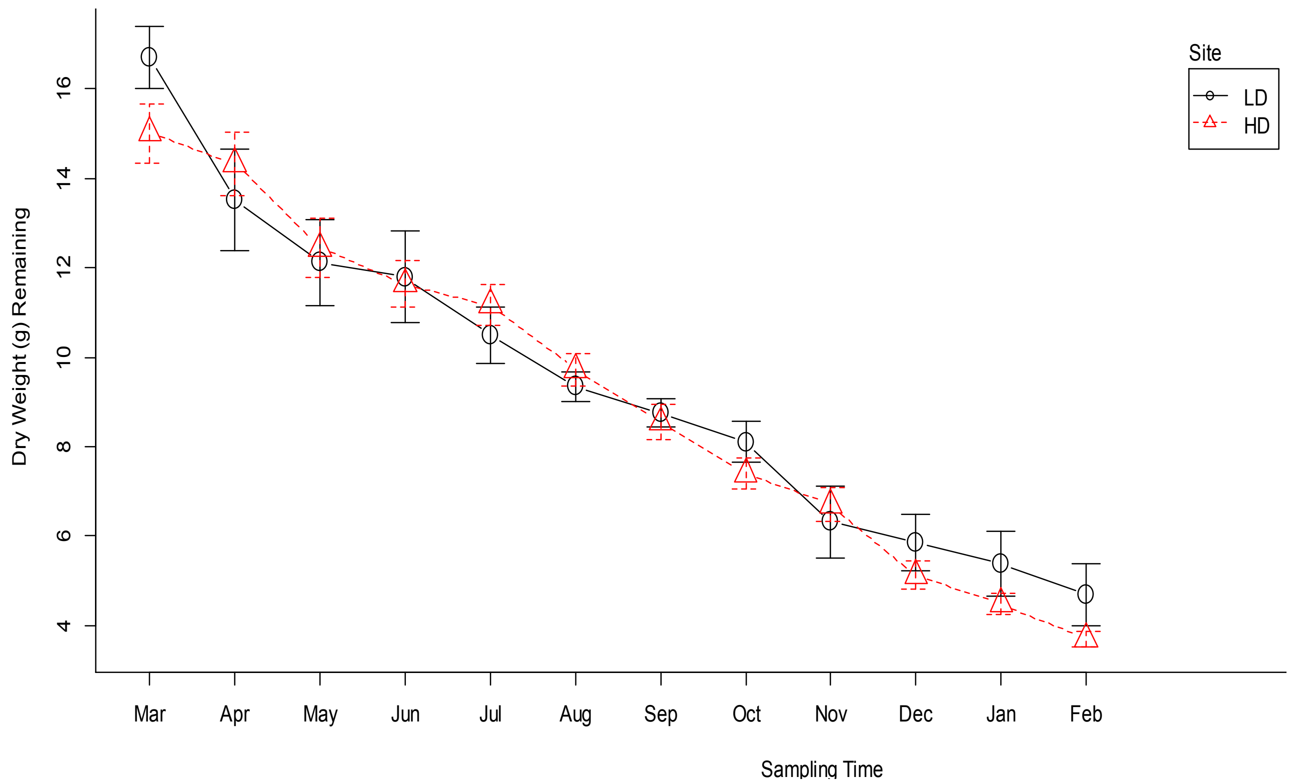


Figure 27.Variation in leaf litter decomposition after 365 days of incubation (n=5 for each points; dotted line = HD, solid line = LD site).

4.18. The effect of climate, soil moisture and soil pH on decomposing litter in Boter-Becho forest

Multiple linear regression analysis showed that there was a good prediction of the decomposing leaf litter by the four explanatory variables such as rainfall, temperature, soil moisture and soil pH. The full regression model is significant and a good fit for the data ($F_{4,115} = 48.98$, $P < 0.001$, $R^2 = 0.63$). The regression model explained 63.0% of the variation in the mass loss of leaf litter during decomposition in both sites of the forest. Among the four predictors or explanatory variables, both rainfall ($t = 7.90$, $P < 0.001$) and temperature ($t = 11.22$, $P < 0.001$) better explains the variation in the dry mass of decomposing leaf litter in *Boter-Becho* forest. However, soil moisture and soil pH did not seem to affect the rate of decomposition directly as supported by the analysis (t = 1.54, P = 0.126 for soil moisture and t = -0.73, P = 0.47 for soil pH) to indicate that the two variables has no direct effect on the leaf litter decomposition in this forest.

4.19. Decomposition rate constant (K) in the Boter-Becho forest

Decomposition constant or sometimes called the decay constant (K) for forest leaf litter was higher during the first three sampling period for HD and the second, third and the last sampling perions in LD site. The mean of decay constant (k) of leaf litter in between two sites are not statistically significant (P = 0.24). The mean of highest decay constant was recorded in April (k = 0.16 month^{-1}) and March (k = 0.16 month^{-1}) for LD and HD sites, respectively (Figure 28). It remains the same from the fourth to the eighth sampling period in both LD and HD sites of the forest. The mean values of the two sites into account, the decomposition constant increases in the first three sampling periods and then, start to decrease from fourth to eighth sampling periods. The highest mean monthly decomposition constant (0.185 month^{-1}) in Boter-Becho was recorded in May (third sampling time). This is because the optimum rainfall and temperature recorded during this month, which facilitates the activities of decomposers with suitable soil moisture compared to others.

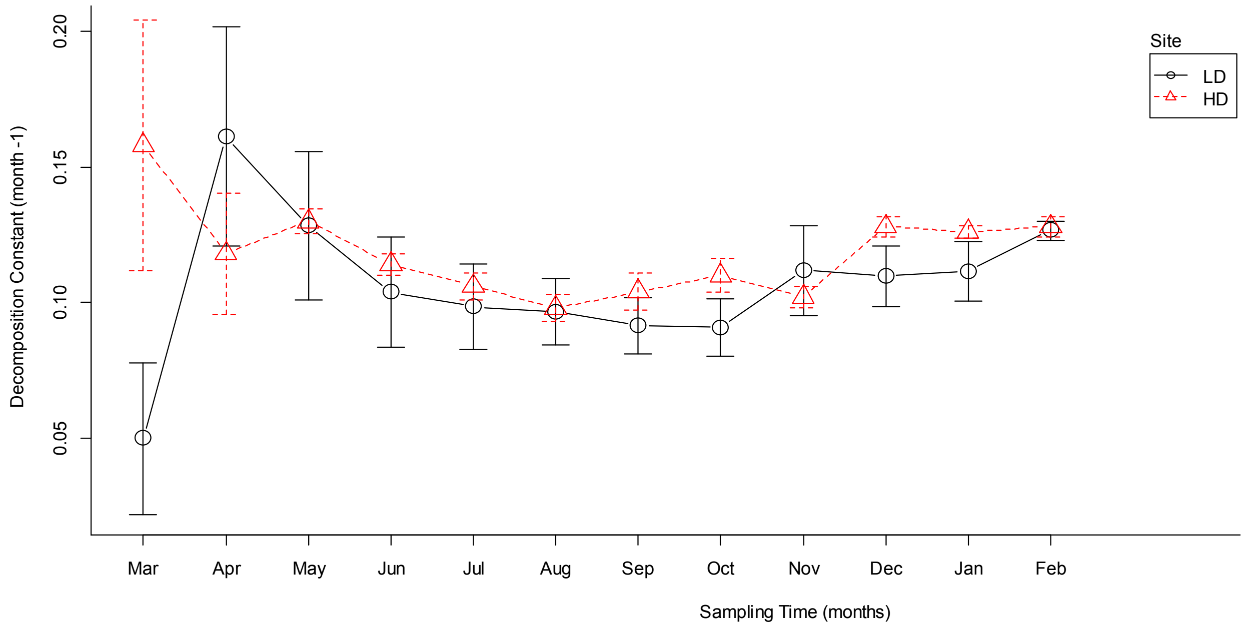


Figure 28.The decomposition constant (K) for two sites of the forest (n=5 for each point)

The annual decomposition rate constant (k) for the HD sites of the forest (1.53 year^{-1}) is higher than that of the LD sites (1.28 year^{-1}). In general, the mean value (1.405 year^{-1}) here represent the annual decomposition constant of Boter-Becho forest which is one of the Moist Evergreen Afromontane Forests of Ethiopia (Appendix VIII).

4.20. The Relationship between Dry mass, Carbon and Nutrients remaining in Boter-Becho Forest

The linear regression analysis showed that carbon and nutrients of decomposing leaf litter strongly predict the decomposition of leaf litter in LD sites of the forest (Table 27). There was strong relationship ($R^2 \geq 0.20$, $p \leq 0.05$) between dry mass and the litter nutrients remaining during decomposition in LD sites (Table 27).

Table 27. Relationship between Dry mass, Carbon and Nutrients remaining in LD sites

C & nutrient	Intercept	Slope	R^2	$F_{1,34}$	P value
C (%)	3.536	0.114	0.481	31.476	< 0.001**
N (%)	4.686	0.062	0.206	8.83	0.005**
P (%)	5.104	0.052	0.365	19.50	< 0.001**
K (%)	6.017	0.056	0.356	18.83	< 0.001**
C:N	3.60	0.07	0.247	11.14	0.002**

*Note: **Correlation is significant at the 0.01 level (2-tailed). n=36*

Similar to the LD sites, the linear regression analysis also showed that the three variables; C, N, P have strong relationship with the monthly leaf litter decomposition in HD sites of the forest. A very weak and non-significant relationship was noted between mass of leaf litter and K and C: N content in decomposing leaf litter (Table 28).

Table 28. Relationship between Dry mass, Carbon and Nutrients remaining in HD sites

C & nutrients	Intercept	Slope	R^2	$F_{1,34}$	P value
C (%)	2.113	0.117	0.819	154.16	< 0.001**
N (%)	4.78	0.068	0.596	50.25	< 0.001**
P (%)	5.382	0.052	0.338	17.38	< 0.001**
K (%)	8.59	0.024	0.017	0.576	0.453^{ns}
C:N	12.37	-0.046	0.065	2.37	0.133^{ns}

*Note: **Correlation is significant at the 0.01 level (2-tailed).* ^{ns} *= not significant, n=36*

4.21. Carbon and Nutrient release pattern during decomposition in Boter-Becho Forest

4.21.1. Carbon and Nitrogen Release pattern

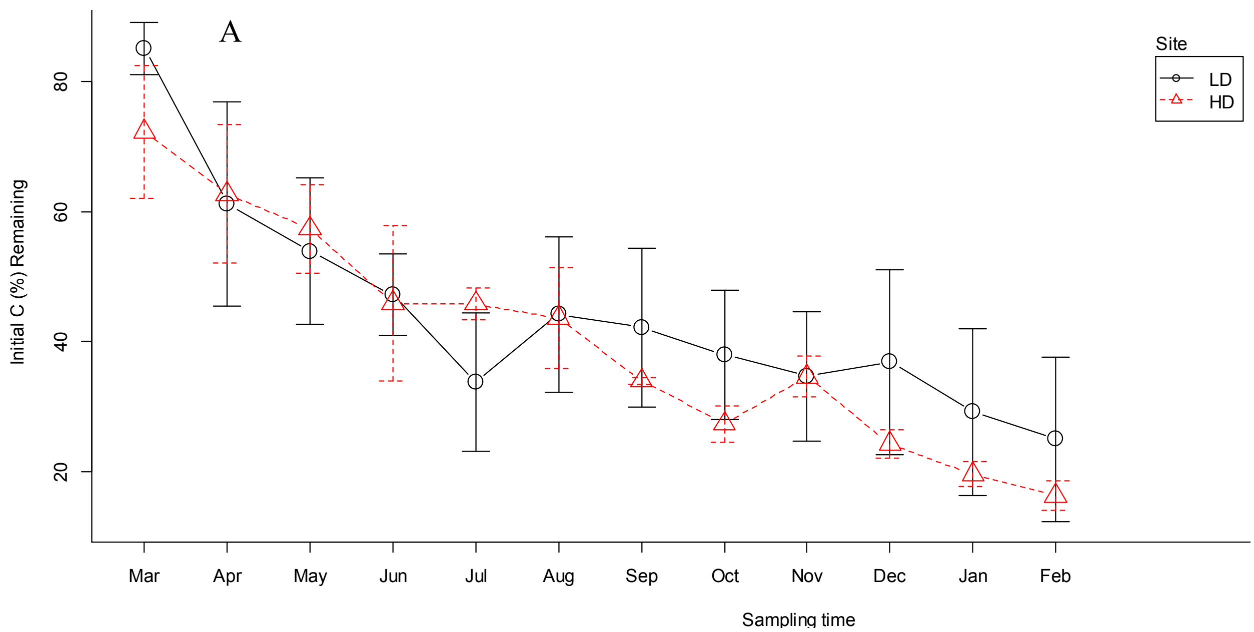
There was a rapid rate of carbon loss from the decomposing leaf litter in the first five months (0-150 days) for LD sites and in the first four months (0-120 days) for HD sites after the onset of the litter decomposition. The carbon remaining (%) in LD sites started to increase from July to August and then eventually decreases up to November of the study year. Finally, the decrease in the C remaining (%) continues until the end of the experiment in LD sites. Similarly, the C remaining (%) during litter decomposition in HD sites remains more or less the same from June to August and then started progressively to decrease up to October. There was a little increase observed in November and then a rapid decrease up to the end of the experiment in HD sites (Figure 29A). An average of about 56.26% of the carbon was remained by releasing 43.74% into the soil or to the atmosphere in LD site for the first five months. Similarly, in HD sites, 56.84% of the carbon was remained by releasing 43.16% into the soil system or atmosphere in the first five months. The release of carbon into the soil/atmosphere continues progressively in both sites of the forest as was depicted in the figure 29A below. Finally, at the end of 365 days of decomposition period, 44.26% and 40.30% of the carbon was remained by releasing 55.74% and 59.7% into the soil system or to the atmosphere in LD and HD sites, respectively.

When both sites of the forest compared, there was no statistically significant difference in the mean of C remaining (%) during decomposition (Table 32). Similar to the carbon concentration, the differences were not significant for all other initial nutrients remaining between LD and HD sites of the forest. In fact, the carbon release from the organic matter of leaf litter increases as decomposition period increases in both sites of the forest. It was observed that the release pattern of carbon from decomposing leaf litter in both sites depend on seasonal variation of a year which depend on the climatic variables (Figure 29 A and B). As evidence, the higher carbon release was observed in the first seven months of a decomposing period in both sites of the forest where the area gets optimum rainfall.

This may be because of the influence of decomposition by the decomposers activity in the soil system as they get suitable soil moisture, temperature and rainfall.

An increase in N remaining (%) was observed in the first four sampling periods (0-120 days) in HD sites of the forest but a slight decrease in the first three sampling periods (0-90 days) in LD sites (Figure 29B). A rapid decrease in N remaining (%) was noted from June to August and a slight decrease continues from September to the end of the experiment in HD sites. The monthly rapid decreasing pattern was observed in May (0-90 days), July (0-150 days) and November (0-210 days) in nitrogen remaining (%) in decomposing leaf litter in LD sites. A slight increase in June (0-120 days), August (0-180 days) and September (0-210 days) was observed for initial N remaining (%) during decomposition in LD sites. Nevertheless, a slight decrease of N remaining (%) continues up to the end of the decomposition period in LD sites as can be observed from December to February in the figure 29B.

The result showed an inconsistent decrease and increase in N remaining (%) during litter decomposition. This fluctuation of N remaining (%) in decomposing leaf litter during the experiment period may be due to the difference in litter type though not considered in species-wise and availability of N (%) in the soil system of the forest. The researcher may also relate these results in such a way that N-poor leaf litter tends to accumulate N and N-rich leaf litter tends to release N even though it is difficult to explain this trend directly from the present study. The one way ANOVA test showed no significant difference in N remaining (%) in decomposing leaf litter between the two sites of the forest (Table 30). At the end of 365 days of decomposition period, about 62.75% and 60.25% of the N remained (%) in decomposing leaf litter by releasing 37.25% and 39.75% into the soil system or to the atmosphere in LD and HD sites of the forest respectively. Even though the difference was not significant, the relatively higher value of N was released in the HD than LD site. This may be due the presence of nitrogen fixing dominant tree species of family *Leguminaceae* such as *Albizia gummifera, Calpurnia aurea* and *Milletia ferruginea* Subsp. *darassana* in this site.



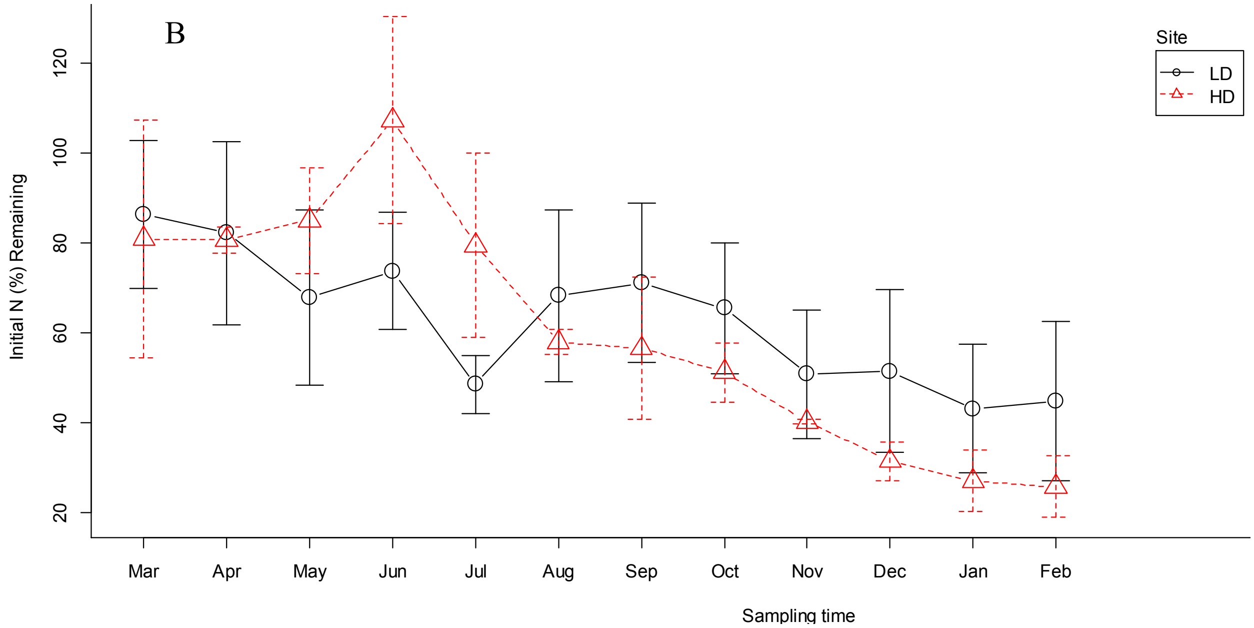


Figure 29. The variation of C (A) and N (B) remaining (%) in leaf litter during decomposition (n = 3 for each points).

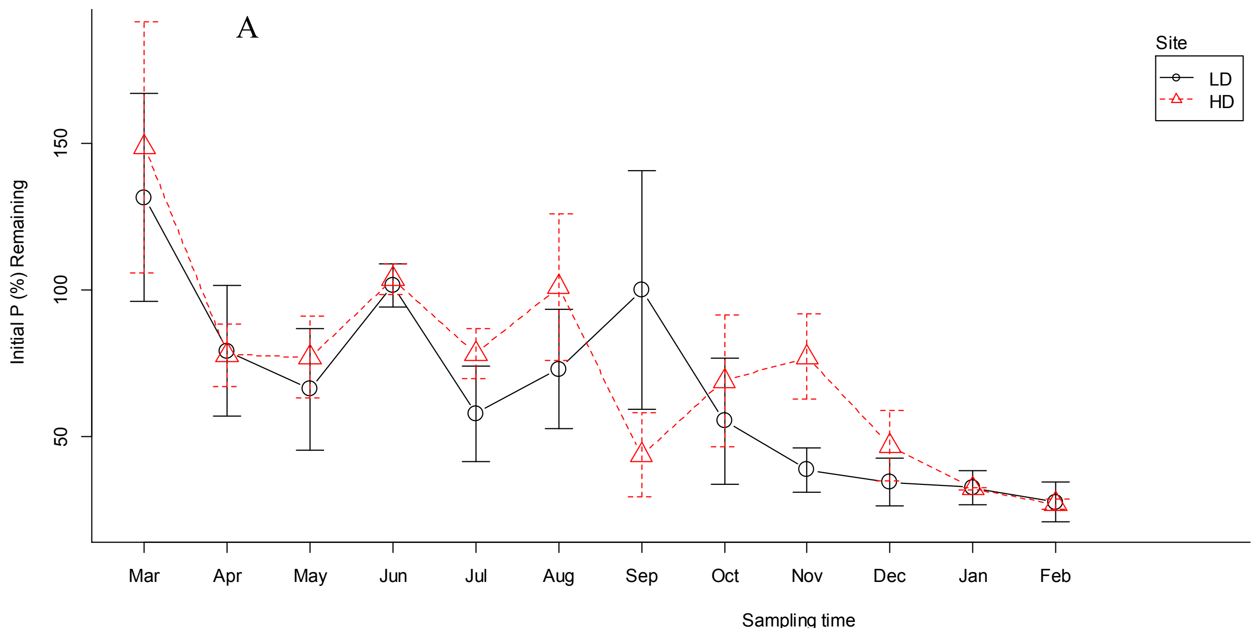
4.21.2. Phosphorus (P) and Potassium (K) Release pattern

The initial P remaining (%) in the decomposing leaf litter showed a large temporal variability in the course of decomposition experiment (Figure 30A). The concentration of P decreased during the first three months (0-90 days) over the initial one and then simultaneously increased in the fourth sampling time (0-120 days) for both sites of the forest. The release of P in the first three months was comparatively faster in LD sites than HD sites showing the rapid P mineralization. In contrast, P increase was simultaneously observed in the fourth sampling time at both sites. At LD sites, the P immobilization (increased P concentration over the initial one) was observed from 5^{th} to 7^{th} sampling time and then P mineralization (the rapid decrease of the P concentration remaining or fast release of P from the litter) was observed from September up to the end of the experiment (Figure 30A). At HD sites, a slight increase in initial P (%) remaining was observed in the fourth,sixth, eighth, and nineth sampling time and a fast decrease was observed in July and September, and from November up to the end the decomposition experiment.

The initial P remaining (%) in the decomposing leaf litter was not significantly different between the two sites of the forest (Table 31). Despite the inconsistent pattern of increase and decrease, the initial P remaining (%) at the end of 365 days was 66.50% and 73.50% by releasing 33.50% and 26.44% in LD and HD sites, respectively.

A rapid decrease of K remaining (%) in the first four months (0-120 days) of decomposition periods was observed indicating fast K- release into the soil system in LD sites (Figure 30B). An increase in K remaining (%) was noted in July, September, November sampling times during decomposition at the same site. In contrast, a drastic decrease in K remaining was noted in August and October, and similarly a very slow decrease was observed in the last three months from December to February sampling time in LD sites. In case of HD sites, there was K increase over the initial K remaining (%) in the first two sampling time (0-60 days) observed. On top of this, there was also small increase in K remaining (%) during decomposition in the fourth, sixth, and nineth sampling time. At the same site, a decrease in K (%) over the initial one was observed in the third, fifth, and eighth sampling period indicating K- release to the soil system and apparently a slow decrease continues after November till the end of the decomposition experiment (Figure 30B).

When both sites compared in terms of the K remaining (%) during the litter decomposition, the mean difference was statistically significant (Table 29). At the end of the decomposition period, the initial K remaining (%) was found to be 42.85% and 26.91 % by releasing 57.15% and 73.09% of the initial concentration in LD and HD sites of the forest, respectively.



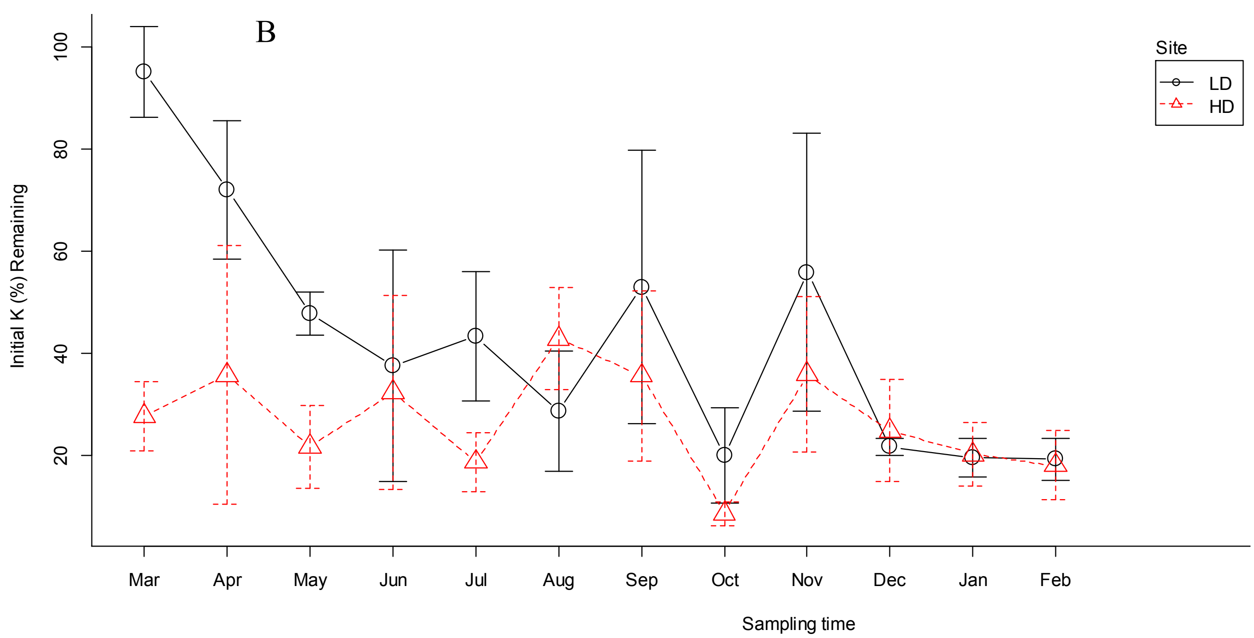


Figure 30.The variation of A/ P(%) and B/ K(%) remaining in leaf litter during decomposition (n=3 for each points, dotted line = HD site, solid line = LD site).

When means compared, one way ANOVA showed not significant difference in C and nutrients (%) remaining in decomposing leaf litter of the two sites except for K (Table 29). The probable reason might be the possession of the same climate (rainfall and temperature), the proximity of the two sites and some forest species overlap among others. The initial litter chemistry by which the nutrient release was calculated is indicated in table 30. The order of nutrient release rates was ranked as K > C > N > P in both sites indicating the nutrient release pattern of the Boter-Becho forest. The linear regression analysis also showed that there was no correlation between initial leaf litter chemistry and C, N, P concentration released ($P > 0.05$) during the decomposition. A significant positive relationship between initial litter K and K released ($P = 0.021$) was indicated probably due to high leaching in the forest considering the two sites.

Table 29. ANOVA table for carbon and nutrient (%) remaining between LD and HD sites.

C and Nutrients remaining (%)	LD (Mean±SE)	HD (Mean ±SE)	$F_{1,70}$	P-value
C (%) remaining	44.26 ±3 .80	40.30 ± 3.24	0.628	0.431
N (%) remaining	62.75 ± 4.60	60.24 ± 5.40	0.126	0.723
P (%) remaining	66.50 ± 7.23	73.55 ± 7.04	0.031	0.487
K(%) remaining	42.86 ± 5.23	26.9 ± 3.43	6.487	0.013*

*Note: *the difference is significant at 0.05 level (2-tailed), n=72*

Table 30. Initial carbon and nutrient concentration analyzed from the litter mixtures in both sites

Site	Plot	C (%)	N (%)	K (%)	P (%)	C:N
LD	LD1	50.13	3.67	0.0145	0.0248	13.66
	LD2	48.78	3.23	0.0139	0.0251	15.10
	LD3	46.91	3.16	0.0150	0.0151	14.84
HD	HD1	44.89	2.26	0.0133	0.0144	19.86
	HD2	42.38	2.68	0.0147	0.0236	15.81
	HD3	39.34	2.71	0.0102	0.0126	14.52

Note: LD1, LD2,LD3 are plots in low disturbed sites and HD1,HD2,HD3 are plots in highly disturbed sites

CHAPTER FIVE

5. DISCUSSION, CONCLUSION AND RECOMMENDATIONS

5.1. DISCUSSION

5.1.1. Floristic Composition and Diversity of Biteyu forest

Although Biteyu forest is highly impacted by anthropogenic disturbance, the plant species richness is higher than the other fragmented dry Afromontane forests of the country (Mesfin Tadesse, 1993; Abate Ayalew *et al.*, 2006; Alemineh Alelign *et al.*, 2007). The higher species richness was due to the higher herb species favoured by disturbance. Similar study by Mekonnen Biru (2003) in the same place some 12 years ago identified the total of 177 species belonging to 143 genera and 68 families. Moreover, he classified the vegetation of Biteyu forest into six community types at dissimilarity level between 0.75 to 0.85. However, from the present study a total of 190 plant species belonging to 154 genera and 73 families were identified (Appendix I and II). The species richness have been increased due to high human disturbance in the forest. This is evidenced from the indicator species of forest disturbance identified, as mentioned in the study done by Abate Ayalew *et al*. (2006). These species include *Phytolacca dodecandra, Croton macrostachyus, Kalanchoe petitiana, Bersama abyssinica* and *Sanicula elata.* Species richness of Biteyu forest was compared with other similar forests (Table 31).

Table 31. Comparing the species richness of different Afromontane forest with Biteyu

Forest	Richness	References
Menagesha Suba State Forest	112	Dinkissa Beche (2011)
Chilimo forest	213	Teshome Soromessa and Ensermu Kelbessa (2013)
Wof Washa Forest	250	Demel Teketay and Tamirat Bekele (1995)
Gedo Forest, West Shewa	235	Birhanu Kebede (2010)
Anabe Forest, Southern Wello	120	Mesfin Tadesse (1993)
Donkoro forest	174	Abate Ayalew *et al.* (2006)
Afromontane forest of Wondo Genet	240	Mamo Kebede *et al*. (2013)
Biteyu Forest	190	Present study

With regard to endemism in the flora area, about 20 (10.53%) endemic plant species were identified from the Biteyu forest, which shows a relatively better species diversity of endemic plants compared to the moist afromontane forest. This support the report of many authors (Friis *et al*., 2001; Friis and Sebsebe Demissew, 2001; Teshome Soromessa and Ensermu Kelbessa, 2013; Demel Teketay and Tamrat Bekele, 1995) in which the proportion of endemic plant species in afromonatne forests of Ethiopia ranges between 10-15% of the total number of species sampled. This is mainly the case for the dry afromontane forest of Ethiopia unlike the moist afromontane forest, which is very poor in diversity of endemic plants. For instance, 12% (29 species) of the plant species occurring in the *Wof Washa* forest are endemic as reported by Demel Teketay and Tamrat Bekele (1995).

Similarly, eighteen of 213 plant species (8.45%) recorded in Chilimo forest, dry afromontane forest, are endemic to the flora area (Teshome Soromessa and Ensermu Kelbessa, 2013). Furthermore, Fekadu Gurmessa *et al*. (2013) also recorded 18 species (10% of the total) from Komto Moist Afromontane Forest, which relatively contain more endemic species than other smilar forests. Several other studies attempted to support Friis *et al.* (2001) who reported on endemism in Afromontane Forests of Ethiopia. For instance, Shiferaw Belachew (2010) recorded 10 endemic plant species from Sesse forest of southwest Ethiopia. Lemessa Kumsa (2010) recorded 18 endemic plant species from Gura Lopho afromontane forest of west Ethiopia. Dereje Denu (2006) recorded 12 (6.6%) endemic plant species from Bibita forest, southwest Ethiopia. Furthermore, Feyera Abdena (2010) recorded 12 (7.8%) endemic plant species from Chato Natural Forest, Horo Guduru Wollega Zone, West Ethiopia.

5.1.2. Phyto-geographical comparison of Biteyu with other forests in Ethiopia

As described in Tadesse Woldemariam *et al*. (2008), it seems not feasible to compare the species diversity of one forest directly with the other partly due to variations in size, survey methods used, and objective of the study. Even if those factors mentioned above could be the evidences for the differences, the overall species richness of the forest can give a general impression of their diversity and phyto-geographical similarity. Accordingly, the direct comparison of Biteyu forest, considering 52 species, with other similar Dry Afromontane forests showed higher affinity with Chilimo, Menagesha Suba, Wof Washa, Angada and Gedo forest (Table 32).

All these forests belong to the Dry Evergreen Montane Forest of the country and topographically Biteyu forest is in the same altitudinal range with the others. However, Biteyu forest showed a very low similarity with Bibita forest with a similarity coefficient of 0.41. Unfortunately, this forest is located in different climate regime compared to the Biteyu forest i, e. Bibita forest is classified under Transitional Afromontane Rainforest while the Biteyu forest is classified under Dry Evergreen Montane Forest by which both forests share small number of common species. The highest resemblance of Biteyu forest was observed with Gedo and Angada forest with the similarity coefficient of 0.85 and 0.76, respectively. These forests possess climatically similar characteristics compared with the Biteyu forest. On top of this, their distance on the landscape scale to the Gurage Mountain chain where Biteyu forest located is approximately closer. They both have possessed the characteristic similar species such as *Juniperus procera, Ilex mitis, Olinia rochetiana, Maytenus addat, Maesa lanceolata, Podocarpus falcatus, Syzygium guineense, Bersama abyssinica, Apodytes dimidiata, Olea europaea* Subsp. *cuspidata* with Biteyu forest.

Gedo Forest, located in Cheliya District of West Shewa Zone in Oromia National Regional State has altitudinal ranges of 1300-3060 m a.s.l. However, the floristic study by Birhanu Kebede (2010) has considered the altitudinal ranges of 2200-2800 m a.s.l, which is quite similar in topographic feature with the Biteyu forest of Gurage Mountain chain. Inaddition, the Angada forest is the part of southeast highlands, which is the extension of Harerge, Arsi and Bale highland massifs (Demel Teketay, 1996) but Biteyu forest is the part of the extension of the central highlands of the country. Even if the altitudinal range of Merti-Woreda where Angada forest located is approximately 1050 to

2800 meters above sea level, the forest altitudinal range considered by Shambel Alemu (2011) for his study was from 2000 to 2800 m asl. This made the area quite similar phytogeographically with the Biteyu forest.

Table 32. Phytogeographical comparison of Biteyu forest with other six forests

Forest	Altitude(m)	No. species compared	Similarity Coefficients
Chilimo[1]	2400 - 2900	31	0.74
Menagesha Suba[1]	2300-3000	26	0.64
Wof Washa[1]	2100-3600	29	0.71
Angada[2]	1050-2800	32	0.76
Bibita[3]	1750-2200	14	0.41
Gedo[4]	1300-3060	38	0.85

Source: [1]*Tamrat Bekele (1993),* [2]*Shambel Alemu (2011),* [3]*Dereje Denu (2006),* [4]*Birhanu Kebede (2010),*

5.1.3. Forest Structure of Biteyu and Boter-Becho Forests

For the present study, forest structure was analyzed as to the concept defined by Leemans (1989) who considered tree density, basal area, tree height and DBH for the stand structure. Accordingly, the tree density at Biteyu forest is smaller than Boter-Becho forest in both diameter classes (>10 and >20 cm) because of the influence of man on the forest ecosystem and the difference in forest type (Table 35). Boter-Becho forest, the moist afromontane forest, is denser in both diameter classes than the Biteyu forest indicating less human disturbance. According to Grubb *et al*. (1963), it was reported that the ratio of 'stem density at DBH class >10 cm to 'stem density at DBH class > 20 cm can be used as a measure of the distribution of the different size classes. The ratio of stem density at DBH > 10 cm to DBH > 20 cm in Biteyu forest was 1.65. When compared, this value is smaller than that of Chilimo and Menagesha forest but denser than that of Wof-Washa forest (Table 33) due to high human influence in the Biteyu forest.

On top of this, stem density ratio of DBH > 10 to DBH > 20 cm in Boter-Becho forest was 1.84 indicating the predominance of individuals of tree species in the DBH > 10 cm compared to DBH > 20 cm. The prevalence of small to medium-sized individuals and the presence of some large trees in this forest may indicate that the forest is in a late stage of secondary development. Moreover, the highest stem density in Boter-Becho forest was recorded in the DBH < 10 cm, which may indicate the good regeneration status of the forest as it has been managed by the Oromia Forest and wildlife enterprise.

Table 33. Comparison of stem density (no. ha^{-1}) of the Biteyu and Boter-Becho with other forest

FOREST	DBH > 10 cm(a)	DBH > 20 cm(b)	a/b	References
Chilimo, DAF	638	250	2.55	Tamrat Bekele (1993)
Menagesha, DAF	484	208	2.33	Tamrat Bekele (1993)
Wof-Wosha, DAF	329	215	1.53	Tamrat Bekele (1993)
Jibat, MAF	565	287	2.0	Tamrat Bekele (1993)
Denkoro, DAF	526	285	1.85	Abate Ayalew *et al*. (2006)
Angada forest, DAF	372.8	252	1.48	Shambel Alemu, (2011)
Gedo Forest, DAF	832	464	1.79	Birhanu Kebede (2010)
Biteyu, DAF	451.11	273.33	1.65	Present Study
Boter-Becho, MAF	789.66	428.79	1.84	Present Study

The average basal area of Biteyu and Boter-Becho forest from the present study were 41.64 m^2 ha^{-1} and 72.93 m^2 ha^{-1}, respectively. The basal area of the Biteyu forest is smaller than that of the Boter-Becho forest mainly due to less number of the larger and taller tree species because of selective cutting of larger trees such as *Juniperus procera* and *Podocarpus falcatus* for timber and house construction purpose. Since basal area is a function of diameter at breast height, the larger the DBH, the larger also the basal area. For instance, nine most important woody species of the forest contributed the highest proportion (87.9%) of basal area in the Biteyu forest (Table 14). Similarly, Desalegn Tadele *et al*. (2013) identified five species accounted about 78.2% of the basal area while *Prunus africana* alone contributed about 45% of the basal area of Zengena forest. In Biteyu forest, *Ilex mitis*, have a stem number of 29.26 ha^{-1} but contributed 14.34% of the total basal area due to possession of relatively the larger DBH. Moreover, *Syzygium guineense* Subsp. *afromontanum* contributed 21.76% to the total basal area even if it has a lower stem density (211.42 no. ha^{-1}) compared to other tree species with high density. It has been shown from the survey that most of the individuals in the Biteyu forest are found in the lowest DBH class, which is also reflected by the high aggregation of individuals in the lowest height class.

According to Birhanu Kebede (2010), the total basal area of Gedo Forest was 35.45 m^2 ha^{-1}; of which 61.87% is contributed by *Ficus sur, Podocarpus falcatus, Prunus africana, Olea europaea* subsp. *cuspidata, Olinia rochetiana* and *Apodytes dimidiata.* Similarly, 87.9% of the total basal area of Biteyu forest is contributed by the plant species such as *Olinia rochetiana*, *Ilex mitis, Maytenus addat, Maesa lanceolata, Podocarpus falcatus, Schefflera volkensii, Olea europaea* ssp. *cuspidata, Myrsine melanophloeos and Hagenia abyssinica*. The three plant species (*Olinia rochetiana, Olea europaea* subsp. *cuspidata, Podocarpus falcatus*) are commonly dominating the basal area of the Gedo and Biteyu forest indicating the DAF nature of both forests.

According to Curtis & McIntosh (1951), importance value indicates structural dominance and ecological significance of tree species in the forest. The tree species with the top IVI values in Biteyu forest were *Olinia rochetiana, Maesa lanceolata, Ilex mitis, Maytenus addat, Podocarpus falcatus, Schefflera volkensii* and *Olea europaea* subsp. *cuspidata.* Many studies in Dry Afromontane parts of the country reported few species to have high IVI values (Demel Teketay, 1996; Ermias Aynekulu, 2011; Desalegn Tadele *et al*., 2013). However, some of the species with high IVI are attributed to the high relative density such as *Olinia rochetiana* (20.50), *Maesa lanceolata* (16.70) and *Podocarpus falcatus* (9.11). About 37.05% of the total stem density was represented by the smallest diameter class (<10 cm) in Biteyu forest. Similarly, Desalegn Tadele *et al*. (2013) reported that 87% of the individuals had DBH classes of less than 10 cm indicating the dominance of small-sized individuals in the Zengena forest, which is having more individuals than Biteyu forest in the same DBH classes. Moreover, in Zegie Peninsula, about 89% of the individuals in the forest had a DBH value of less than 7.5 cm (Alemineh Alelign *et al*., 2007).

Some tree species, which have high stem density but their basal area, are not as large as their density indicating the size difference between species as indicated also in other studies (Tamrat Bekele, 1993; Gillespie *et al*., 2000). The plant species with the largest basal area in the forest could be the most important in terms of economic, cultural, ecological and social significance. Several authors worked on ecology of different forests have reported different value of the total basal area of each forest (Tamrat Bekele, 1993; Gillespie *et al*., 2000; Abate Ayalew *et al*., 2006). The basal area of the Biteyu forest (41.64 m^2 ha^{-1}) is larger than similar dry forests of the country due to the sampling method differences and presence of small number of larger trees, which contributed, to the basal area calculation. Some other dry afromontane forests of Ethiopia have such figures reported in Chilimo (30.1 m^2 ha^{-1}) Menagesha (36.1 m^2 ha^{-1}) and Gedo forest (35.45 m^2 ha^{-1}) (Tamrat Bekele, 1993; Birhanu Kebede, 2010). The smaller basal area (22.03 m^2 ha ^{1}) than the present values is recorded for tropical dry forests in Central America (Gillespie *et al*., 2000). Similarly, the total basal area of Zengena forest, a remnant montane forest, which looks similar in structure with Biteyu forest, in northwestern Ethiopia was 22.3 m^2 ha^{-1} (Desalegn Tadele *et al*., 2013). However, Biteyu forest was higher in basal area than that of Zengena forest due to the presence of small number of important species with relatively bigger diameter. Zengena forest possesses high density of small sized individuals compared to the Biteyu forest. On the other hand, the total basal area of Boter-Becho forest, moist afromontane, is closely similar with the figures reported by various authors in similar forests (Feyera Abdina, 2010, reported a total basal area of 65 m^2 ha^{-1} in Chato Natural Forest; Fekadu Gurmessa *et al*., 2013 reported a total basal area for Komto Forest of 50.72 m^2 ha^{-1}).

According to Clark and Clark (2000), tropical forest structures were known to vary with climate human impacts and topography. The patterns of density diameter distributions of plants in the Biteyu forest are the results of human disturbance and topography. Accordingly, the forest structure of Biteyu has been affected by the uncontrolled cutting of trees (counted from the cut wood stump) and cattle interferences among anthropogenic disturbances. This can also be explained in terms of population structure. The difference in species population structure in the Biteyu forest leads to the differences in forest structure. *Olinia rochetiana, Maesa lanceolata*, *Nuxia congesta* (not shown) and *Hagenia* *abyssinica* have shown an inverted J-curved structure in Biteyu forest. Many tropical forests possess such structure by which species represented by an inverted J-curve indicating good regeneration and recruitment potential (Tamrat Bekele, 1993; Desalegn Tadele, 2013; Richards, 1996). Moreover, economically most important trees such as *Juniperus procera* and *Podocarpus falcatus* had shown interrupted inverted J curve type of population pattern. This may be attributed to the human disturbance existing in the forest such as cutting for house construction and fencing services (Lieberman *et al*., 1996; Desalegn Tadele *et al*., 2013). *Olea europaea* subsp. *cuspidata* and *Schefflera volkensii* followed a Gauss curve type of species structure. This structure showed few individuals in the lower and higher size classes and increasing number of individuals around the middle classes. This population structure was due to hampered regeneration caused by poor recruitment and selective removal of the species at higher diameter classes. Many authors who worked in afromontane forests of the country (Demel Teketay, 1997; Desalegn Tadele *et al*., 2013; Ermias Lulekal *et al*., 2008; Tamrat Bekele, 1993) described this forest structure in their study. The other population pattern was represented by the *Ilex mitis* where there was an increase in species density in the first four consecutive DBH classes. The density starts to decrease, then increases, and finally decreases at the largest DBH class. This is irregular pattern, which may be caused by poor regeneration and recruitment indicating the browsing, grazing and selective removal of some of the forest species.

The local disturbance has a significant impact on biodiversity conservation and species composition, which in turn may lead to loss and fragmentation of forests as reported by Foaham and Jonkers (1992). As an evidence of justification, several authors have examined the long-term effects of selective cutting of important trees on the forest structure and composition (Plumptre, 1996; Chapman and Chapman, 1997; Struhsaker, 1997). For instance, studies on forests of Uganda have shown poor tree regeneration even >20 years after logging (Struhsaker, 1997) whereas the forest structure requires >50 years to return to pre-logging conditions (Plumptre, 1996) so that the current study forest needs urgent intervention by the responsible stakeholders.

5.1.4. Plant Community-Environment Relationship in Biteyu Forest

According to many authors (Gauch, 1982: Ter Braak, 1994), classification and ordination as methods of multivariate analysis, can provide more detailed and comprehensive information on the distribution pattern of plant species and their response to the underlying environmental gradients. Bolstad *et al*. (1998) described in such a way that vegetation in highland regions respond to small-scale variation in terrain like slope, which affect microclimatic conditions such as temperature and soil moisture. Plant species distribution in turn is affected by temperature and moisture, which are the major microclimatic conditions.

Variation in environmental gradients may explain the floristic heterogeneity of the area. Specific environmental gradients determine the distribution of dominant species, which in turn affects the identified plant communities (Feyera Senbeta, 2006). Similary, from the analysis, slope and altitude was found to be highly significant environmental variable among others, which are responsible for the patterns of community formation in the study forest (Table 15).

In the ordination, altitude was the most important variable in weighing axis one so that it explained considerable amount of variation in the pattern of species distribution and structure of plant community formation. Environmental variables highly correlated with axis one are highly responsible for weighing the axis. Thus, altitude strongly affects species composition and forest structure by having an effect on other environmental variables. Moreover, slope followed by altitude was the most important constraining variables in weighing axis two in the ordination. Consequently, it strongly affects species composition and structure of Biteyu forest. According to Tewelde Birhan Gebre Egziabher (1988), slope has a strong effect on soil chemical properties as to the soils on steeper slopes are influenced by bedrock and tend to be less moist and less acidic. Environmental variables, considered in this study, correlated with CCA axes 1-6 to varying degrees contribute their own share in determining the plant species distribution pattern and plant community formation (Table 14). However, the terrain variables; altitude and slope in larger proportion, directly or indirectly influences these environmental variables for the pattern and formation of community structure in the forest.

5.1.5. Biomass Carbon in Dominant Tree Species of both forests

In Biteyu forest, the highest average biomass carbon was stored in *Ilex mitis, Maytenus addat, Olea europaea* subsp.*cuspidata, Podocarpus falcatus, Olinia rochetiana, Myrsine melanophloeos* and *Schefflera volkensii* in descending order. These are the dominant species remaining in the forest and with the highest IVI values. They contain few large sized individuals that contribute larger proportion of the total biomass carbon. Thus, the plant species with large sized individuals have significant contribution to the carbon storage in this forest. Their removal significantly alters the biomass dynamics of the forests. Consequently, low AGB carbon values and high density of small stems recorded in the Biteyu forest would suggest the significant human disturbance, which was also reported by Alves *et al*. (2010) elsewhere. Similarly, the smaller size classes (DBH < 20 cm) held 61.86% of the total stem density but a very small fraction (1.6%) of the live AGB carbon in Biteyu Forest. Other studies also support that majority of AGB was found within taller trees and trees with a larger diameter. Moreover, these taller trees with larger diameter, hold the largest stocks of carbon within biomass, and are often impacted by forest degradation and deforestation (Gibbs *et al*., 2007). As evidence, forest degradation and deforestation are the largest source of carbon emissions in tropics and are significant contributors to climate change within many tropical countries (Pan *et al.* 2011). It may also be the single greatest threat to the global biodiversity as mentioned in Hill and Curran (2003).

On the other hand, the Boter-Becho Forest is characterized by the plant species with highest average carbon stock in ABG and BGB such as *Olinia rochetiana, Syzygium guineense* Subsp. *afromontanum, Ficus sur, Croton macrostachyus, Macaranga capensis, Ilex mitis* and *Pouteria adolfi-friedricii* in decreasing order. The smaller size classes (DBH < 20 cm) held 80.14% of the total stems density but only 1.5% of the live AGB carbon in *Boter-Becho* forest. This showed that those larger sized class (DBH > 20 cm) species mentioned above held only 19.86 % of the total stem density but the largest (98.5%) live AGB carbon. Accordingly, Thompson *et al*. (2009) stated that mature and diverse forests are more likely to provide greater ecosystem resilience, providing a better guarantee of the long-term persistence of forests and therefore the carbon pools.

5.1.6. Carbon stocks in Different pools of both forests

The average total carbon stock of Biteyu and Boter-Becho forest (Table 22) from the present study was relativey similar with other studies in Ethiopia. Similar studies done in different forest ecosystems of Ethiopia showed significant difference in the amounts of total carbon stocks estimated. Abel Girma *et al*. (2014) and Tibebu Yelemfrhat *et al*. (2014) found out an average of 348.8 t ha^{-1} and 568.314 t ha^{-1} in Mount Zequalla Monastery forest and Lowland Area of Simien Mountains National Park, respectively. Similarly, Muluken Nega *et al*. (2015) and Belay Melese *et al*. (2014) found out an average of 507.29 and 583.27 t ha^{-1} in Adaba-Dodola Community forest and Arba-Minch Ground Water Forest, respectively. Hamere Yohannes *et al*. (2015) also estimated an average of 523.64 ± 29 t ha^{-1} in Gedo Forest of Oromia Regional State. Moreover, Adugna Feyesa *et al*. (2013) estimated an average of 614.72 ± 35.79 t ha^{-1} in Egdu forest. This difference is attributed to the difference in topography, sampling type, forest type, soil characteristics, level of disturbance, and forest structure and species composition of the forest ecosystem among others.

The average total carbon stock of Biteyu forest is smaller than that of other studies done in the dry evergreen montane forest (for instance Gedo forest, Hamere Yohannes *et al*., 2015) due to the higher anthropogenic disturbance mainly selective cutting of large trees. The carbon stock value of Boter-Becho Forest resembles to the carbon stock value of study done by Abel Girma *et al*. (2014) in Mount Zequalla Monastery forest compared to other studies done in Ethiopia, which are higher than the present value due to methodological differences. Similar study outside the country (Delaney *et al*., 1997) estimated the AGBC of 354 t ha^{-1} in upper montane moist forest in Venezuela, which is closely related to the carbon stock estimation of Boter-Becho Forest of the present study. Moreover, Zhu *et al*. (2010) reported the mean total ecosystem C density of 237 t ha^{-1} (ranging from 112 to 338 t ha^{-1}) across all the forest stands in Northeast China. Of the total ecosystem carbon density, 153 t ha^{-1}, 14 t ha^{-1} and 70 t ha^{-1} was stored in vegetation biomass, in forest detritus and in soil organic matter (1-m depth) respectively.

The carbon stock of forest detritus (14 t C ha^{-1}) here is much larger than the litter carbon for Biteyu (0.26) and Boter-Becho forest (0.30 t C ha^{-1}) because in the forest detritus of Northeast China included standing dead trees, fallen trees, and floor material but only leaf litter and herbs were included in the sample of the present study. On the contrary, Swai *et al.* (2014) reported smaller carbon pools; the mean carbon stocks 48.37, 45.71 and 0.26 t C ha^{-1} in AGB tree species, herbs and SOC respectively in Hanang forest, Tanzania. For further information, the mean carbon stocks of different carbon pools in different Forests of Ethiopia were compared with the present study in the Table 34.

Table 34. Estimates of carbon stocks (t ha^{-1}) in different carbon pools by different authors

Authors	Forest Name	AGBC	BGBC	Soil C	Litter C
Hamere Yohannes *et al*. (2015)	Gedo Forest	281	56.1	183.69	0.41
Muluken Nega *et al*. (2015)	[a]ADCF	319.43	-	186.4	1.06
Abel Girma *et al*. (2014)	[b]MZMF	237.2	47.6	57.6	6.5
Belay Melese *et al.* (2014)	[c]AGWF	414.70	83.48	83.80	1.27
Mohammed Gedefaw *et al*. (2014)	[d]TGF	306.37	61.52	274.32	0.90
Tibebu Yelemfrhat *et al*. (2014)	[e]LLSMNPF	270.89	54.18	242.51	0.019
Adugna Feyissa *et al.* (2013)	Egdu Forest	278.08	55.62	277.56	3.47
Mesfin Sahile (2011)	[f]MSSF	133	26.99	121.28	5.26
Tulu Tolla (2011)	[g]AACF	122.85	25.97	135.94	4.95

The Present study	Biteyu forest	87.13	22.94	56.37	0.26
Present study	[h]BBF	189.4	43.98	159.31	0.30

Note: [a]*Adaba-Dodola Community Forest,* [b]*Mount Zequalla Monastery Forest,* [c]*Arba Minch Ground Water Forest,* [d]*Tara Gedam Forest,* [e]*Lowland Area of Simien Mountains National Park,* [f]*Menagesha Suba State Forest,* [g]*Addis Ababa Church Forests,* [h]*Boter-Becho forest*

Biteyu forest from the present study has a mean carbon stock of 87.13 ± 11.80 t C ha^{-1} and 22.94 ± 2.84 t C ha^{-1} in above and belowground biomass, respectively. The mean total carbon stock of AGB for tropical dry forests ranged between 30-126 t C ha^{-1} (Houghton, 1999; Defries *et al.,* 2002; Brown, 1997; IPCC, 2006; Swai *et al.,* 2014) in which the carbon stock of AGB (87.13 t C ha^{-1}) in Biteyu forest from the present study is in the same range. Moreover, Gillespie *et al.* (1992) reported that dry biomass may vary from 50 to 550 t ha ^{1}and explained that the causes for this wide variation in biomass production are soil, climate, species, stand density, stand age, and management.

Biteyu forest have low stem density compared to other dry evergreen forests in the country, and it can explain the lower estimates for biomass and carbon stock as similarly indicated in other studies (Hamere Yohannes *et al*., 2015; Abel Girma *et al*., 2014; Tibebu Yelemfrhat *et al*., 2014; Mwakisunga and Majule, 2012; Munishi *et al*., 2009). The direct forward reason for this is the direct anthropogenic impact on the forest including selective logging of trees, fuel wood collection, and subsistence agriculture, grazing and browsing effect.

The total AGB contributed 52.30% in Biteyu Forest and 48.2% in Boter-Becho Forest. Large trees (DBH > 50 cm) contributed more than 58.23% of the average carbon stock in AGB of the Biteyu Forest whereas more than 61.6% of the average carbon stock in AGB of the *Boter-Becho* Forest. However, in Biteyu forest large trees (DBH > 90 cm) contributed more than 75% of the total AGB. Similar research carried out in the Brazilian Atlantic forest indicated that large trees contribute to 78% of the total AGB (Lindner, 2010). Moreover, a similar study in Amazon has estimated that about 1% of all the tree species account for half of the carbon locked in the vast South American rainforest. Although the region is home to an estimated 16,000 tree species, researchers found that only 182 species dominated the carbon storage process. Almost 20% of the world's terrestrial vegetation carbon stock is stored in the tropical forests of Amazonia. It is vital to the Earth's carbon cycle, storing more of the carbon than any other terrestrial ecosystem (Fauset *et al.,* 2015). However, when the biomass removal is too severe as in Biteyu Forest unlike Boter-Becho Forest which is relatively protected, the morphology, health, and regeneration ability of the trees are seriously affected resulting in a severely altered forest ecosystem as was reported in (Fauset *et al.,* 2015; MarenandVetaas, 2007).

The findings of this study showed that carbon storage and partitioning among different components in both forests and other similar forests of the country vary greatly. This variation is mainly due to the difference in forest type, species composition, forest structure, soil nutrient availability, climate, disturbance regime and topography, which was similarly reported by various scholars (Gillespie *et al.*, 1992; Yong *et al.*, 2011; Zhu *et al*., 2010; Houghton, 2005).

5.1.7. Effect of Altitude and Slope on different Carbon Pools in both forests

Altitudinal gradients in tropical forests are a powerful tool to improve our understanding of the relationship between environmental parameters and ecosystem structure (Malhi *et* *al*., 2010). In tropical montane forests with elevations ≥ 1000 m, AGB correlations were slightly stronger with elevation and slope but never significant (Girardin *et al.,* 2014). However, there was a positive but weak correlation was observed between AGB carbon and altitude, and the correlation was significant in Boter-Becho Forest (Figure 23A). This is attributed to altitudinal gradient, the plots variation in the species richness, the presence of large stemmed trees at optimum altitude and the moisture variation mainly in Moist Evergreen Montane vegetation of Ethiopia. Similar studies in tropical montane forests have been reported by various authors (Abel Girma *et al*., 2014; Mwakisunga and Majule, 2012; Tibebu Yelemfrhat *et al*., 2014; Munishi *et al*., 2009; Adugna Feyissa *et al*., 2013; Mwampamba, 2009). For instance, an increase in biomass carbon stocks with increasing elevation was reported in the range of 2031 and 2312 m a.s.l in rungwe forest, southern highland of Tanzania (Mwakisunga and Majule, 2012). Moreover, Alves *et al*. (2010) found a progressive increase in stem density and live AGB with elevation.

The number of stems and live AGB differed significantly among forest types: stem density and estimated AGB were higher in submontane and montane forests than in other forest types. Swai *et al.* (2014) reported that there was significant difference in tree carbon along an altitudinal gradient except in SOC and herbaceous carbon, which were not significantly different in the three layers along an altitudinal gradient in Hanang forest, Tanzania. Moreover, Abel Girma *et al.* (2014) noted a significant variation in carbon stock of different carbon pool with exception for soil organic carbon stock. They observed that the amount of carbon stock in above and below ground biomass showed increasing pattern with increasing altitude whereas litter and SOC stocks showed decreasing pattern with increasing altitude. It was reported that SOC pool size is mainly determined by C output (decomposition), which generally decreases with increasing altitude (Garten and Hanson, 2006). However, SOC stock in the Boter-Becho forest is not significantly correlated to altitude eventhough an overall increasing trend was observed with altitude (Figure 23C).

In contrast, Hamere Yohannes *et al*. (2015) described that altitude has significant effect on all carbon pool except litter biomass and soil organic carbon. According to them, there was a decrease of carbon stock in total AGB and BGB at higher altitude having the significant effect. Similarly, the carbon stock in litter biomass and soil also decreases as altitude increases but not significant in Gedo Dry Evergreen forest, Ethiopia. Similar to the Hamere Yohannes *et al*. (2015), the carbon stock estimated in AGB and BGB of Biteyu forest is negatively correlated to the altitudinal gradient and also the altitudinal effect is significant on the carbon stock of AGB and BGB. This is to mean that the carbon stock of AGB and BGB decreases with altitude. This is attributed to the less species diversity and strong dominance of a single to a few species at higher altitude compared to the trend at lower altitude in dry tropical afromontane forest.

According to Dossa *et al*. (2013), altitude was consistently the most important factor in determining species diversity for all components. Similarly, change in woody species richness has potential implications for carbon storage, which often declines with increasing elevation (Girardin *et al*., 2014; Zhu *et al*., 2010). Total AGB decreased by 50 to 70% between 1050 and 3060 m (Moser *et al*. 2011). The difference in forest structure and biomass stocks may be found among sites in forests over short distances. This is attributed to short altitudinal gradients in tropical regions, which may exhibit stronger edaphic discontinuities over short distances (Daws *et al*., 2002; Takyu *et al*., 2003).

Moreover, atmospheric pressure variability, air temperature and solar radiation are primarily a function of elevation (Alves *et al*., 2010) but light availability, soil moisture, soil temperature, and soil nutrients are expected to co-vary along short elevational gradients in the tropics due to steep topography (Silver *et al*., 1999; Takyu *et al*., 2003; Aiba *et al.,* 2004).

Even if it was not significant, the SOC stock positively but very weakly correlated with altitudinal gradient in Boter-Becho Forest. Similarly, it was noted that the SOC was positively correlated with altitudinal gradient in Biteyu forest, the altitudinal variation having a significant effect on the SOC. This variation in SOC along altitudinal gradient might be due to the decreasing temperature and increasing precipitation in addition with a reduction in litter decomposition at higher altitude. Similar studies have done by various authors support this finding (Raich *et al*., 2006; Du *et al*., 2014; Saeed *et al*., 2014). According to Tan *et al*. (2004), the effect of altitude on SOC was observed by controlling geologic deposition processes, soil water balance and soil erosion. More importantly, the above authors noted that the variations in SOC stocks were highly related to decreasing temperature and increasing precipitation with altitude, which in turn resulted in decreased litter decomposition at high altitude sites though the altitudinal range is quite different compared to the present study. Soil organic carbon increased with increase in altitude with high correlation coefficient (Mwampamba, 2009; Yong *et al*., 2011; Saeed *et al*., 2014; Koppad and Tikhile, 2015).

Many authors also reported an increase in the SOC stocks with increasing altitude even with SOC stocks at high elevations demonstrated to be higher than the AGB carbon stocks in lowland rainforests (Malhi *et al*., 1999; Wilcke *et al*., 2008; Zimmermann *et al*., 2010). Furthermore, Griffiths *et al*. (2009) mentioned that SOM accumulation is likely driven by a reduction in decomposition rates rather than an increase in primary productivity at higher altitude. This can be supported by a decrease in temperature with increasing elevation and simultaneous reduction in organic matter decomposition, which would increase soil C content. On the other hand, Sheikh *et al.* (2009) reported the contrasting result; the stocks of SOC were found to be decreasing with increasing altitude from 141.6 to 124.8 t C ha^{-1} in subtropical forests. Abel Girma *et al*. (2014) reported that SOC stocks showed decreasing pattern with increasing altitude in Mount Zequalla Monastery in Ethiopia**.** Many other authors have reported the decreasing pattern of SOC with altitude in different forest types and ecosystems (Hamere Yohannes *et al*., 2015; Griffiths *et al*., 2009).

Yong *et al.* (2011) reported that the SOC stock is significantly affected by both altitude and slope. The larger variation of SOC stock along altitudinal gradient on southern slope attributed to the destruction of original vegetation and the establishment of forest plantation. It could be one of the important factors affecting the spatial distribution of soil organic carbon. They also found that on northern slope, soil organic carbon content increased with increasing altitude. Moreover, the study done by Fentaw Yimer *et al*. (2006) in Ethiopia showed that SOC varied significantly among three vegetation communities at different topographic aspects. Furthermore, Fu *et al*. (2010) studies showed wide variation in distribution and stocks of SOC across four vegetation types.

Litter carbon in Boter-Becho forest along the altitudinal gradient does not show any trend even if its slope in the regression is positive and non-significant. Similarly, it does not show any trend in Biteyu forest though the slope of regression is negative and non-significant. Adugna Feyissa *et al.* (2013) noted similar result that the litter carbon stock showed irregular patterns along altitudinal gradient and the relationship was not statistically strong. Nevertheless, many studies reported the decreasing trend of litter carbon with increasing altitude and the difference was statistically significant (Abel Girma *et al*., 2014; Hamere Yohannes *et al*., 2015; Griffiths *et al*., 2009).

Maren *et al.* (2015) reported that topographic factors affect mountain forests through their direct influence on radiation and moisture. On top of this, human disturbance also plays a significant role by affecting vegetation and soil characteristics in the mountain forest. However, the topographic factors, mainly the altitude factor, have significant effect on the biomass of vegetation and soil organic carbon from the present study on Biteyu forest. More importantly, in Biteyu forest, there was strong evidence to report that the altitude of the forest has a significant effect on the carbon stock of AGB, BGB, and soil though it was not significant for the litter carbon (Figure 24 A-C). The major reason for this might be, partly explained due to low species composition and richness along altitudinal gradients particularly for Biteyu forest. On the other hand, the terran factor (slope) tends to show a very weak but positive relationship with all response variables in Boter-Becho forest (Figure 23 A-F).

5.1.8. Litterfall Production in Boter-Becho forest

It was reported from the present study that the mean total litterfall in LD and HD sites of the Boter-Becho Forest was 10.76 t $ha^{-1}$and 6.64 t ha^{-1}, respectively. The annual average of the two sites (8.7 t ha^{-1}) represents the total litterfall of Boter-Becho forest, classified in the moist evergreen montane forest of the country. This result is supported by similar studies done in tropics by several authors (Staelens *et al*., 2011; Chave *et al*., 2010; Yang *et al*., 2005; Murphy and Lugo, 1986; Rai and Proctor, 1986; Singh *et al*., 1999; Vituosek and Sanford, 1986; Williams and Gray, 1974). The amount of total litterfall in this forest is in the range of tropical moist forest; 3.6 to 12.4 t ha^{-1} yr^{-1} as reported by Vituosek and Sanford (1986). Williams and Gray (1974) reported 5.5–15.3 t ha ^{1} yr^{-1} in equatorial rainforests and still the value of litterfall in the present study is in this range. Moreover, the total annual litterfall in a dry tropical forest of India ranged between 4.88-6.71 t ha^{-1}yr^{-1} (Singh, 1992) which is a bit less than the value reported in the present study probably because Boter-Becho Forest is grouped under a moist tropical forest. Further study in the tropics (Cuevas and Lugo, 1998) also showed relatively similar result with the present study. According to Cuevas and Lugo (1998), litterfall ranged from 8.1 to 14.3 t ha^{-1}yr^{-1} with an average of 11.1 t ha^{-1} yr^{-1} for the ten tropical tree species.

From Southern Ethiopia, Ambachew Demessie *et al*. (2012) also reported litterfall production under broad-leaved plantation species and natural forest in the range of 9.7 to 12.6 t ha^{-1}yr^{-1} by which the value of the litterfall in the present study is in the same range. In Ethiopian Highlands, Nigatu Lisanework and Michelsen (1994) reported annual fine litter production ranging from 5.0 to 6.5 t ha ^{1} in tree plantations and 10.9 Mg ha ^{1} in a natural forest relatively higher than the present value. In some tropical forests in the Western Ghats, India, Sundarapandian and Swamy (1999) found average annual litterfall of 5.63–8.65 t ha^{-1}. Furthermore, from 400 litterfall measurements of different forest ecosystem analyzed by Zhang *et al.* (2014), the mean annual litterfall varied among and within various ecosystems in the range of 3–11 Mg ha^{-1} y^{-1}.

The total litterfall recorded from the present study is also in the same range of total litterfall in mixed stands of southern China, which varied from 4.89 to 11.45 t ha^{-1} (Wang *et al*., 2008). Of the total litterfall, leaf litter contributed 42.23% and 49.5% of the total annual litterfall in LD and HD sites of the forest respectively with the average value of 45.87% in the forest system. The leaf litter contribution to the total litterfall in the forest was 45.87% followed by reproductive parts (24.13%). The order of annual litterfall production in the present forest of tree components was leaf litter > reproductive parts > others > twigs.

The total annual litterfall varied from 8.65 to 9.09 t ha^{-1} yr^{-1} in the study done on subhumid tropics of Eastern India. Of which, leaf litter contributed from 59 to 60 %, wood litter from 30 to 31.6 %, flower litter from 2.4 to 3.0 % and pod litter from 6.2 to 6.8 % (Gawali, 2014). Of the mean annual litterfall at the mixed stand (7.69 t ha^{-1} yr^{-1}) of southern China, leaf litter contributed the highest portion (68%) compared to other components (Wang *et al*., 2008). Similarly, of the total litterfall ranged between (4.88-6.71 t $ha^{-1}yr^{-1}$) in the tropics, 65-72% was leaf litterfall and 28.35% wood litterfall (Singh, 1992). Several studies support that leaf litter contributed the highest proportion to the mean annual litterfall in tropical as well as temperate forests and plantations (Ewel, 1976; Hoque *et al*., 2015; Kumar and Tewari, 2014; Oziegbe *et al*., 2011) which is instrumental in nutrient cycling in the forest ecosystem.

5.1.9. Seasonal Pattern of litterfall production in Boter-Becho forest

Litter production followed a seasonal pattern with lowest values during wet seasons and highest values in dry seasons in tropical forest ecosystem (Parsons *et al*., 2014; Zhang *et* *al*., 2014; Scheer *et al*., 2009; Sanches *et al*., 2008; Tanner *et al*., 1998). Similarly, the present study showed the higher litterfall in dry compared to wet season indicating that the seasonal variation significantly affects the litterfall in the forest. The possible reason for this might be the presence of deciduous forest species with broad leaves, which shed their leaves in the dry season in the forest. On top this; the highest litterfall in dry season could also be the morphological mechanism of tree species to combat drought stress during the dry period. This result coincides with the peak of litterfall recorded in tropics within dry season (Ewel, 1976; Gawali, 2014; Arunachlam *et al.,* 1998; Sundarapandian and Swamy, 1999). There was a significant difference between LD and HD sites in the mean of dry weight of the total as well as fractions of litterfall in Boter-Becho forest. Similar studies support this finding (Barnes *et al*., 1998; Dirham, 1998). The lower litterfall production in wet season was due to the variation in rainfall pattern. The relationship of litterfall production pattern with rainfall was reported for tropical forest (Lonsdale, 1988. Liu, 2012; Parsons *et al*., 2014).

Rainfall gives the best prediction model for leaf litterfall in tropical forests (Lonsdale, 1988) but both rainfall and temperature were also good predictors of litterfall according to Liu (2012) and Parsons *et al*. (2014). The highest variation for litterfall in the forest was explained by rainfall in the present study as it was highly significant compared to temperature. This is attributed to fact that rainfall may have a two-fold influence on litter production and nutrient input; water stress in dry periods increased shedding of senescent leaves and in contrast, heavy rainfall at some time of a year force the shedding of non-senescent leaves. This in turn provides a nutrient pulse through higher qualities of leaf litter (Cuevas and Lugo, 1998). Moreover, Qiu *et al.* (1998) suggested that rainfall, temperature and light might play an important role in leaf fall and flushing among dominant canopy species in the forest.

The interaction of temperature and rainfall was significant for litter production in the tropics (Vitousek, 1984; Wood *et al.* 2005; Chave *et al*., 2010; Simmons, 1996; Blankinship *et al*., 2011; Wu *et al*., 2011). The present study showed that the litterfall is strongly but negatively related to the rainfall compared to temperature. Linear regression analysis showed that 30.4 % of the variation in annual litterfall was explained by rainfall, whereas temperature explained only 4.6 % variation in total litterfall. However, in opposite to the present study, a positive correlation was reported between litterfall, and temperature and rainfall (Gawali*,* 2014). In general, litter production in a forest is responsible for the formation of the forest floor and this formation is a long-term process, which is indicative of the nutrient-cycling rate (Gosz *et al*., 1976; Parker 1983; Morrison, 1991; Johansson, 1995).

5.1.10. Decomposition Dynamics of leaf litter in Boter-Becho forest

In the present study, climatic variables (mainly Temperature and rainfall) control the litter decomposition. It is not easy to examine the effect of litter quality on the decomposition process in the forest ecosystem from this study as the litter was composed of the mixture of different plant species to represent the forest. However, the climate variables explained the litter decomposition. The present study showed distinct temporal variation in leaf litter decomposition in both sites of the forest. No single parameter explains the process of litter decomposition in the soil, which suggests that the combinations of environmental and biological factors are involved in the process (Berg and McClaugherty, 2008).

The study area posses long rainy season (seven months) and short dry season (five months). The long rainy season facilitates fast rate of leaf litter decomposition in both LD and HD sites as both sites are proximate to each other. This shows that climate controls the mixed litter decomposition of this moist evergreen montane forest of Ethiopia. The evidence as to various researchers (Vitousek and Sanford, 1986; Lavelle *et al*., 1993; Pandey *et al*., 2007; Tripathi *et al*., 2006; Latter *et al.* 1998) is that climate has a direct effect on leaf litter decomposition due to the effects of temperature, rainfall and moisture. Of the explanatory variables considered (rainfall, temperature, soil pH, soil moisture), rainfall and temperature is the most important factor which affects the rate of forest leaf litter decomposition in the present study. It has been known that the effect of soil pH depends up on the interaction between rainfall and temperature. Thus, slow rate of litter decomposition occurs both at low pH (low temperature and high rainfall) and high pH (high temperature and low rainfall) and rapid decomposition occurs at intermediate pH levels (Bernhard-Reversat *et al.*, 2003; Berg and McClaugherty, 2008).

The study showed the rapid rate of leaf litter decomposition in the first three months (90 days) of the litterbag experiment in both sites. Even if the decomposition rate is not as rapid as in the first three months, it continues up to the sixth (180 days) decomposition period. This seemed to be probably due to the effect of optimum rainfall during this decomposition period. Moreover, rapid mass loss was strongly related to the soluble labile carbon during the early decomposition phase and at the later stage the amount of carbon and the chemical composition of the carbon, containing molecules (e.g. starch, cellulose, and lignin) were considered as strong determinants of decomposition rate (Berg and McClaugherty, 2008; Harmon *et al*., 1990). High soil moisture during the first seven months (wet season) enhanced the leaf litter decomposition in both LD and HD sites of the forest.

The study also showed strong positive correlation between rainfall and soil moisture in both LD and HD sites of the forest (Table 28 and 29). This shows a trend that strong increase in rainfall consequently increases the soil moisture, which in turn stimulates an increase in the decomposition rate of the leaf litter in the forest. Many authors (Liu *et al*., 2005; Devi and Yadava, 2007) have observed that rainfall has a prime effect on the rate of litter decomposition in the tropical forests. Similarly, Austin and Vitousek (2000) described the effect of rainfall on the physical process of chemical component leaching from litter; the more the rainfall the faster the leaf litter decomposition despite the litter types. Moreover, the initial N and P contents, C: N ratio in leaf litter are good indicators of the decomposition rate among other (Wang *et al*., 2008) though it is not feasible to conclude from the present study. In the first seven months of the decomposing period from the present study, the rainfall was high so that the rainfall enhances the growth of both bacteria and fungi.

The leaching action of rainfall also releases water-soluble substances from the decomposing litters (Keith *et al*., 1997; Bernhard-Reversat *et al.*, 2003). Averti and Dominique (2011) also observed from their study in the forest groves of Central Africa that the rate of litter decomposition was enhanced by the soil moisture in the rainy season of the decomposing period. Latter *et al* (1998) also outlined that rainfall has positive effects on litter decomposition in tropical ecosystems. According to many authors, (Aerts, 1993, 1997; Melillo *et al*., 1993; Robinson *et al*., 1995) climatic change will affect litter decomposition rates as well as changes in litter production and litter chemistry, which are likely to occur because of changes in plant productivity and of plant species composition.

Compared to the first seven months, a relative decrease in rate of leaf litter decomposition was observed in the later five months of the decomposing period in both sites of the forest (Figure 27). This was partly due to the fact that the later five months were dry conditions and less favorable for the activities of soil organisms involved in the decomposition process as observed in Tripath and Singh (1992) and Liu *et al*. (2005). Moreover, this period is in the later phase of decomposition by having remained the recalcitrant compounds. The dry conditions or water shortage brought about negative impacts on the rate of litter decomposition by increasing osmotic pressure, which in turn increases the energy requirements of the microorganisms for osmo-regulation (Harris, 1981). A studies on the invertebrate indicated that the decrease in soil moisture content in dry season causes the migration of soil organisms deeper into the soil by reducing the exposure of the litters to them (Garay *et al*., 1986; Killham *et al*., 1993).

The variation in decay constant, among other factors, may be attributed to the difference in species composition of the forest. Since the litterbag in this study was the mixture of the different forest plant species with different physiological characteristics and litter quality, it is very difficult to conclude which factor is the driving one in the litter decomposition of this forest. Rather it is possible to say that the mixing up of different species leaf litter in the decomposition experiment speeds up the rate of decomposition compared to single-species litter (Liu *et al*., 2005). Similarly, Gartner and Cardon (2004) reviewed 30 papers on decomposition dynamics in mixed species leaf litter specifically on the pattern of mass loss, changes in nutrient concentration and decomposer abundances and activity. They found that decomposition patterns are not always predictable from single species dynamics. Accordingly, when litters of different species are mixed in decomposition experiment, the mass loss is more often (but not always) increased and even exceeds the expected decay by 20% or some times less. Thus, the mixed leaf litter decomposition from the present study supports this trend by representing the litter decomposition dynamics of the whole forest ecosystem.

5.1.11. Nutrient release pattern at different stages of decomposition

The present study showed no significant differences between the two forest sites except the K remaining (%), which is significantly different (Table 31). The insignificant difference between the two sites may be due to their similarity in edaphic and climatic conditions since they are from the same forest and the sites are closer to each other. Moreover, the local disturbance did not seem to have a considerable impact on the C, N and P dynamics in the forest ecosystem contrasting with other studies, which found the influence of species composition on the pattern of nutrient release (Torreta and Takeda, 1999). In the case of K, the local disturbance showed an impact on K dynamics as K is the most leachable cation and non-structural element compared to others. In the HD sites, the mean of K (%) released to the soil system is higher than that of the LD sites of the forest (Table 31). The seasonal variation have shown a significant effect on the leaf litter decomposition in both sites of the forest as concluded in the study conducted by Hasanuzzaman and Hossain (2014).

Even if there was a descent in P remaining (%) in the whole experimental period, an intermittent decrease and increase was observed (Figure 30A). In general, an increase in

P remaining (%) at some point of decomposition in both sites of the same forest can be supported by other similar studies (Hasanuzzaman and Hossain, 2014; Xuluc-Tolosa *et al*., 2003). According to Xuluc-Tolosa *et al*. (2003), P increased initially and then declined towards the end of the experiment. In contrast, a rapid rate of P release was observed at early decomposition period in the present study. This is attributed to a significant portion of P in leaves are inorganic forms and leaching might explain a major part of the P release from the leaf residues (Lisanework Nigatu and Michelsen, 1994; Tesfaye Teklay, 2007) otherwise there was no significant difference observed between litter P content and respective P mineralization (t = 0.247, P = 0.82) in both sites. Similarly, Kwabiah *et al*. (2001) reported that 54-82% of total P in leaves from six agroforestry tree species was in water-soluble forms, of which a significant proportion could be lost by leaching.

Compared to N availability, P availability in many tropical sites is very low. This is attributed to a low P supply through weathering of parent material and by tight absorption of orthophosphate in the widespread oxisols and ultisols (Vitousek and Sanford, 1986). Consequently, it might be expected that variability in P supply is an important control of litter decomposition in the tropical forests (Vitousek and Sanford, 1986). However, this is not the case for the Boter-Becho forest as evidenced from the P increase observed at some point of decomposition particularly in between third sampling and seventh sampling period. This may be attributed to a net accumulation in residual litter at some points of the decompositon period in the study forest.

According to Heal *et al*. (1997), N is one of the factors limiting rate of litter decomposition as it determines the growth and turnover of microbial biomass, mineralizing the organic carbon. Berg and Laskowski (2006) identified an accumulation phase followed by a release phase for the litter N during decomposition among other models. In the case of HD sites, it follows a similar pattern by characterizing early immobilization in the first four months (0-120 days) and the release or mineralization eventually continues. However, N dynamics in the LD site follows another model where a leaching phase (phase I) is followed by an accumulation (phase II) and release phase (phase III) as described in Berg and Laskowski (2006). Accordingly, early N release was observed in the first three months following an increase and then a release in the LD sites compared to relatively low N release in HD sites.

A peak increase of N remaining (%) in decomposing leaf litter was measured in the fourth sampling time for both sites of the forest. The reasons for this increase in wet season could be the result of uptake of an increased pool of available rhizosphere N and nitrogen fixation, particularly for the leguminous species, which have contributed to an increase N level during the wet season. It is true that both sites commonly have leguminous species (*Milletia ferruginea* Subsp. *darassana*) and their leaves were included in the forest leaf mixture considered in litterbag experiment. Many other authors (Lisanework Nigatu and Michelsen, 1994; Tolsma et al., 1987; Tesfaye Teklay, 2007) made similar conclusions. Nevertheless, from the present study, the soil pH has shown a significant correlation ($r = 0.369$, $P = 0.027$, $n = 36$) with the N (%) in decomposing litter in LD sites but not in HD sites ($r = 0.19$, $P = 0.264$, $n = 36$). In the LD sites, the soil pH may have an impact on the N-release. Moreover, the level of soil pH is not significantly different in the remaining analyzed mineral nutrients. The soil pH measured in LD sites range from 4.41- 6.6 whereas in HD sites it ranges from 4.01-6.14. N-mineralization is restricted at low pH levels being the optimal pH for soil biomass growth has been established near neutrality. Nevertheless, in soils with pH values between 4 and 5, a significant N-mineralization has been noted indicating that the microorganisms can be adapted to acid conditions (Shah *et al*., 1990). This trend seems to work for the Boter-Becho forest eventhough the role of pH depends on the interaction between rainfall and temperature for effective decomposition to take place as witnessed by the work of Bernhard-Reversat *et al.* (2003).

A slight descent continues up to the end of the experiment by releasing this nutrient into the forest soil system. High N loss observed in the fifth sampling time of decomposition in LD sites (Figure 29A) may be attributed to stimulation of microbial activity and decomposition due to wetter conditions in the forest despite the influence by the initial litter chemistry. Moreover, Heal *et al.* (1997) mentioned that net release or net increase of nitrogen could be estimated from the organic material's C: N ratio or N concentration. Thus, if the C: N ratio is < 20 or the N concentration $>2.5\%$, N will be released by enhancing rapid decomposition of materials. If C: N ratio is much higher than 20, N is likely to be immobilized until decomposition and respiration lower the C: N ratio. All samples collected from forest leaf litter in the present study have shown original C: N ratio of less than 20 (Table 32). Thus, the nitrogen release in the two sites of the forest has shown a tendency slightly to increase through time in the decomposition period in both sites of the forest. Leaching also can be explained for N release as much as 25% of it in the leaves may be removed by leaching (Vitousek, 1984) particularly at the early phase of decomposition.

A rapid decrease in the K from the decomposing leaf litter was observed in the first four months in LD sites of the forest and in the first, third, fifth and eighth sampling times in the HD sites. The decrease in K was much higher in the early stage than the later stage and similarly, the mean difference in K remaining between the two sites was significant (Table 31). The faster decrease of K from the decomposing leaf litter in the present study was mainly because K is a non-structural element, highly mobile and most leachable cation during decomposition, which was supported by various similar works by various authors (Berg and Laskowski, 2006; Guo and Sims, 2002; Tisdale, 1995; Hasanuzzaman and Hossain, 2014). Moreover, K is not incorporated in to organic structures and hence is less affected by leaf chemistry and soil faunal activity (Tian *et al.*, 1992; Ribeiro *et al.*, 2002). However, from the present study it was determined that the initial litter chemistry strongly correlated with the K release in both sites and it seems that rapid K release was determined by high leaching. Smilarly, with regard to mineralization, Taylor (1989) noted that 85% of K and 50% of P loss in the leaves was by leaching in tropical conditions.

In general, the decrease of initial carbon and nutrients concentration in early stage observed from the present study may be due to the loss of the soluble forms of nutrients at the initial stages of decomposition, which was also noted in a study by Mahmood *et al.* (2007). On the other hand, a slower decrease of initial carbon and nutrients (%) towards the later stages of leaf litter decomposition may be due to microbial oxidation of recalcitrant components, physical and biological fragmentation. Many authors (Mahmood *et al.*, 2014; Hasanuzzaman and Hossain, 2014) noted similar observations. At some points of decomposition in the first seven months, increased nutrients (N, P, and K) concentration in decomposing leaf litter was observed (Figures 29B, 30A and B). This phenomenon is attributed to immobilization in the residual leaf litter acting the decomposing leaf litter as a surface for fungi or heterotrophic organisms (Berg and Laskowski, 2006; Hossain *et al*., 2011; Mahmood *et al*., 2014, Lin *et al.,* 2007).

5.2. CONCLUSIONS

5.2.1. Floristic Analysis of Biteyu forest

The floristic composition of Biteyu forest is composed of high herb species. Shrub and tree species together accounted for small number of the total species. Low species richness of trees in this forest is due to selective logging of some useful tree species such as *Juniperus procera* and *Podocarpus falcatus*. Cattle intereference and selective logging (as counted from the cut wood stump) were indicators of human disturbance, which had a considerable effect on the forest structure and composition to lowered densities of economically useful plant species.

A larger number of plant endemics were recorded from Biteyu forest. Herbs accounted for the highest proportion of (60%) the endemic species. Of the total endemic species, Asteraceae was the richest family followed by Lamiaceae. Because of high disturbance caused by cattle interference (grazing and browsing) and selectve logging of tree species by the surrounding community, less heterogenity among the three community types were observed. This was due to the dominance of a few woody species in common within the three plant communities of Biteyu forest. Those species include *Olinia rochetiana*, *Ilex mitis, Maesa lanceolata, Maytenus addat, Bersama abyssinica, Brucea antidysenterica.* Moreover, important value index of woody species showed that a vey few species dominated the Biteyu forest. These woody species contributed the highest proportion of all species recorded. The dominance of these species may be attributed to their success in regeneration, least preferred by browsing, pathogen resistance and seed dispersal mechanism.

The species population structure of Biteyu forest revealed four major patterns. These were inverted J-curve shaped (high number of individuals in the lower DBH classes), interrupted inverted J-curve shaped represented by *Juniperus procera* and *Podocarpus falcatus*, Gauss curve type, increase in density as DBH increases and then a decrease in the middle diameter as represented by *Ilex mitis.* These structural differences in the Biteyu forest was due to the anthorpogenic effect which may lead to the forest degradation, biodiversty loss and hence to local extinction of plant species.

In the constrained ordination, slope, altitude, available K, disturbance, EC and OC were the environmental variables that significantly contributed to the formation of the plant communities and pattern of species distribution in Biteyu forest. Among the six significant environmental variables, slope and altitude were highly significant in contributing ($p < 0.05$) to the formation of community structure in the Biteyu forest. Cattle interference (grazing and browsing) and selective logging (cut wood stump) were found to be a major threat to the forest degradation and forest biodiversity loss in Biteyu forest.

5.2.2. Forest Structure of Biteyu and Boter-Becho Forest

The DBH classes considered for both forests showed the general trends of population dynamics (inverted J-curve shaped) even if the interruptions among the DBH classes were observed in Biteyu forest. Similar to the DBH class distribution, the six height classes also followed the inverted J-curve shaped population dynamics for both forests. Accordingly, the first height class in Biteyu forest contributed 53.8% followed by the second with 34.7% of the total stem density. Similarly, 53.6% and 36.7% of the total stem density in Boter-Becho forest was contributed to the first and second height class, respectively. The tree height beyond 25 m was totally absent in Biteyu forest but present with some representatives in Boter-Becho forest. As said many times, this was attrbuted to the selective cutting of large trees for various uses by the forest dependent people. With regard to Basal area, the nine most important woody species of Biteyu forest contributed to the highest proportion (87.90%) of the total basal area. This was attributed to the larger DBH of these species. On the other hand, 18 species accounted for the highest proportion (93.42%) of the total basal area in Boter-Becho forest. Thus, the Boter-Becho forest was twice as dominant as the Biteyu forest due to the possession of large stemmed tree species.

5.2.3. Carbon stock and the influence of terrain variables in both forests

In the Biteyu forest, the highest biomass carbon was stored in *Ilex mitis* due to its frequent occurence and wider DBH compared to other high-density woody species. On the other hand, seven tree species in Boter-Becho forest possesses high biomass carbon stocks due to their wider DBH and high density. In both forests, the larger trees (DBH $\geq$ 50 cm) held the highest biomass carbon stock. In Biteyu forest, individuals with DBH $\geq$ 90 cm comrised of only five species contributed to more than 75% of the average carbon stock in AGB. This showed that larger tree species are the major carbon sink and that the selective logging of the larger trees in the forest imposes a serious threat mainly to the AGB carbon pool.

The estimates of total carbon stock in Boter-Becho forest showed twice as much as that of Biteyu forest. This was because Boter-Becho forest is managed forest by the regional government but the Biteyu forest is an open access so that the local disturbance is high. Consequently, Boter-Becho forest possesses higher density of large stemmed trees than Biteyu forest so that they contain large biomass carbon stock. The largest fraction of total carbon stock was stored in AGB followed by soil carbon in both forests. Among the four carbon pools, a significant difference in the mean of carbon stock was estimated between two forests except the litter carbon. This difference might be arised from the difference in forest area, sample size, local disturbance level, climatic conditions, topography, and soil type and vegetation structure among others.

Due to the aforementioned reasons, the estimated carbon sequestration potential of the Boter-Becho forest is the highest compared to the Biteu forest. Nevertheless, the total carbon stock estimated for both forests could be included in the Ethiopian's Forest Reference Level (FRL) submision to the UNFCCC for the implementation of REDD+ initiatives.

Altitude significantly affects the biomass carbon in Boter-Becho forest though the relationship betwen altitude and biomass carbon is positive and very weak. In contrast to altitude, slope showed a very weak positive relationship with all response variables in the same forest. In Biteyu forest, a significant negative relationship between AGBC and altitudinal gradient was observed. Moreover, a significant effect of the altitudinal variation on the soil carbon was observed in Biteyu forest. This means that the amount of soil carbon increases with altitudinal gradient. In contrast, slope in Biteyu forest tends to show the negative relationship with soil carbon but the mean difference was not significant.

5.2.4. Litterfall, Leaf litter Decomposition and Nutrients Release Pattern

The total litterfall measured in Boter-Becho forest is in the same range of tropical forests. This indirectly may tell us that the forest productivity and nutrient cycling are relatively in good status though it needs further study with long-term data. The two highest peaks of litterfall production were observed in March and December. The dry weight of total litterfall as well as its fractions in dry season was significantly higher than wet season in the forest. Of the fractions in the total litterfall, leaf litter contributed the highest portion followed by the reproductive parts in the forest. Seasonal variations in litterfall dynamics were brought about by the variation in rainfall and temperature. However, the variation in total litterfall as well as fractions in the Boter-Becho is significantly affected by the variation in monthly rainfall. Thus, the marked variability between rainfall and litterfall could probably be explained in part by seasonal patterns in phenology.

During decomposition experiment, mass loss rates in all litterbags were relatively rapid in the early stage but highly variable in the decomposing periods. Fast leaf litter decomposition was measured in the first three months. There was a distinct variation noted in residual mass of leaf litter among the months of decomposing year. The annual decomposition rate constant (*K*) measured for Boter-Becho forest (1.405 $year^{-1}$) was in the range of moist tropical forests. At the end of 365 days, 66.37 % of the leaf litter was decomposed in Boter-Becho forest and temperature and rainfall were the most important climatic factors among the four variables regulating the leaf litter decomposition in the forest ecosystem. There was strong positive and significant relationship between the residual mass of leaf litter, and carbon and the nutrients remaining during decompsition in Boter-Becho forest.

The pattern of carbon and nutrient dynamics seemed to have followed the pattern of mass loss during decomposition despite some slight differences. The two sites of the forest did not show significant difference in carbon, major nutrients except K because both sites are in the same forest and closer to each other with similar edaphic factors, climatic variables, and some forest species overlap. Similarly, the local disturbance does not seem to have a considerable impact on the C, N and P dynamics in the forest since the two sites did not show a significant difference. Indirectly, one may conclude that forest soil of Boter-Becho forest has no limitation of N and P as more variation of increase than the decrease was observed during leaf litter decomposition. A rapid release of K was noted throughout the decomposition period probably due to the rapid leaching. In contrast to other studies, the initial litter quality did not show significant relationships with the nutrient release in the decomposition period except for K, which showed the opposite due to its high leaching being non-structural element.

A decrease of dry mass remaining, C and nutrients in the early stage of decomposition in both sites of the Boter-Becho forest is because the temperature in these periods is higher and warmer than the later stage despite the contents of litter quality. The temperature in this stage with optimum rainfall enhances the rate of decomposition and hence the C and nutrient release to the soil or to the atmosphere. Due to larger warming in June to August season in Africa as predicted by Hulme *et al*. (2005), there will be between 2 6 ºC warmer in 100 years. Thus**,** factors that increase the rate of decomposition and hence carbon and nutrient release could serve to increase the amounts of carbon-based gases in the atmosphere. It can also be concluded that a global warming might have an increasing influence on the rate of wet season decomposition in Boter-Becho forest, which will in turn increase carbon-based gases to the atmosphere.

5.3. RECOMMENDATIONS

Based up on the results of the present study, the following recommendations as implications for forest management/conservations were forwarded.

5.3.1. Biteyu forest

Several anthropogenic disturbances such as agricultural expansion, cattle interference (grazing and browsing), selective logging for timber and house construction, and fuelwood collection reported to take place in Biteyu forest. Moreover, it was noted that the community in three villages immediately around the Biteyu forest mainly depend on the forest for their livelihood leading to continuous forest fragmentation. Therefore, there should be a need to intervene further human disturbances within the forest so that it can sustain its ecological functioning. Accordingly,

- community education and awareness creation on the use and ecosystem services of Biteyu forest should be provided by Woreda Natural Resource Management offices.
- Encouraging alternative income generating activities to enhance livelihood diversification such as poultry, modern beehives, improved crop/fruit varieties should be done by the Regional and local governments.
- Woreda Agricultural and Rural development offices in consultation with NGOs should introduce multipurpose tree species in order to buffer the forest conservation zone and promote the sustainable utilization of forest resources.

- Attention should be given by the forest management authorities in order to rehabilitate, restore the degraded areas of the forest through reforestation and natural regeneration of the missing indigenous tree species such as *Podocarpus falcatus,* and *Juniperus procera*.
- Communal grazing land should be allocated for the surrounding community in order to reduce grazing and browsing pressure on the forest ecosystem.
- A participatory and community centred approach of forest management scheme should be established and promoted in the area by NGOs, Federal and Regional gov'ts.
- There has to be an appropriate policy implementation and coordinated institutional agreement with the local people in order to effectively manage and or conserve the forest and its resources in the area.

5.3.2. Biteyu and Boter-Becho forest structure

It was understood from the result that height and diameter class distribution as well as IVI of woody (Biteyu forest) species indicated the harvesting potential of economically important indigenous species. The dominance of economically less valuable species such as *Olinia rochetiana, Ilex mitis* and *Bersama abyssinica* was greater than the more valuable species particularly in Biteyu forest. Moreover, the density of tree species in the larger DBH class in Boter-Becho forest was lower than that of similar forests in the Southwestern parts of the country due to the agricultural expansion and other anthropogenic disturbance as confirmed by the information from the forest guards and author's personal obervation. Consequently, the carbon stored in the AGB of trees was the most directly affected by selective logging and the most relevant in reforestation and rehabilitation efforts being done in REDD+ initiative. This was because the largest fraction of total carbon stock was stored in AGB followed by the soil pool in both forest ecosystems.Thus,

- Recommendations made in section 5.3.1 above also similarly works for this section.
- Establishment of permanent forest plots particularly in Boter-Becho Forest should be conducted for long-term monitoring of changes in carbon stocks, to evaluate forest growth response to climate and land use change, and to be a source of quality reference data for international mechanisms to reduce emissions from deforestation and forest degradation.
- The elevational and slope variations on abundance, species richness and all the carbon pools should be taken into account in designing sustainable forest management plan and hence enhancing forest conservation and ecosystem services.
- The Federal and Regional Governments should formulate Forest policy, which mainly gives recognition to the tenure rights of people living in and around forests, for sustainable management of forests, for enhancement of forest carbon stocks.
- Taking into account the total estimated values of C-stocks from both forests, it is noteworthy to explore the implementation of carbon credit systems such as REDD+ in the future, if sustainable management of forests take place. Thus, this information should be included in the country's FRL submission to UNFCCC in the context of results-based payments for the implementation of REDD^{+}.
- Federal and Regional government of the country should develop a more resilient and strong forest conservation strategy, which encompasses a different land use types, social and economic needs of local community, tenure rights and local capacities.

5.3.3. Litterfall, Leaf litter decomposition and nutrient release pattern

The seasonal litterfall dynamics play an important role in the process of forest carbon and nutrient cycles. It was concluded that future changes in seasonal rainfall patterns in response to anthropogenic disturbance will likely to have direct and indirect impacts for the litterfall dynamics of Boter-Becho forest in particular and moist evergreen montane forest of the country in general. The dry season favors the litterfall so that it increases soil organic matter on the forest floor. Therefore, quantifying rates of litterfall and seasonal pattern of litterfall dynamics are quite important for determining the forest ecosystem functions.

- Determining the nutrient release pattern through decomposition of leaf litter is fundamentally important for the ecological equilibrium and maintenance of the forest ecosystems. A study should be conducted in order to determine how much of the carbon and nutrients are released to the soil and how much of them is released to the atmosphere in the forest ecosystem for climate change mitigation strategy.
- This result from Boter-Becho forest may be the first step towards predicting litter decomposition dynamics under seasonal variation in different disturbance level. In order to make the generalization of the present findings, similar studies with the wider scope in the same as well as other forests of the country should be conducted.
- Carbon and other nutrients loss in the decomposing leaf litter progressively decreased during the decomposition process from the initial levels with rapid release in the early stage. Therefore, permanent plot for monitoring the long-term data on litter decompsition, carbon and nutrient dynamics should be established in Boter-Becho forest for monitoring forest ecosystem. Moreover, separate litter decomposition

experiment for wet and dry season should be conducted in the future to understand the seasonal variation in the rate of decomposition and nutrient dynamics in the forest ecosystem.

- The release patterns of other macronutrients such as magnesium, calcium, sulphur during decomposition of forest litter along with the study of soil microbial population should be conducted in the future study. Moreover, the lignin and cellulose contents of the forest species should be chemically analyzed to understand whether the litter decomposition is controlled by these contents or other climatic factors in the forest ecosystems.

6. REFERENCES

Abate Ayalew, Tamrat Bakele and Sebsebe Demissew (2006). The undifferentiated afromontane forest of Denkoro in the central highland of Ethiopia: A floristic and structural analysis. *SINET*: *Ethiopian Journal of Science* **29**:45-56.

Abel Girma, Teshome Soromessa and Tesfaye Bekele (2014). Forest Carbon Stocks in Woody Plants of Mount Zequalla Monastery and its Variation along Altitudinal Gradient: Implication of Managing Forests for Climate Change Mitigation**.** *Journal of STAR* **3**(2): 132-140.

Aboal, J.R., Arévalo, J.R. and Fernández, Á. (2005). Allometric relationships of different tree species and stand above ground biomass in the Gomera laurel forest (Canary Islands). *Flora* **200**: 264-274.

Adams, M.B. and Angradi, T.R. (1996). Decomposition and nutrient dynamics of hardwood leaf litter in the Fernow whole-watershed acidification experiment. *Forest Ecology and Management* **83**: 61–69.

Adugna Feyissa and Teshome Soromessa (2017). Variations in Forest Carbon Stocks Along Environmental Gradients in Egdu Forest of Oromia Region, Ethiopia: Implications for Sustainable Forest Management. *American Journal of Environmental Protection* **6** (1-1): 1-8.

Adugna Feyissa, Teshome Soromessa and Mekuria Argaw (2013). Forest Carbon Stocks and Variations along Altitudinal Gradients in Egdu Forest: Implications of Managing Forests for Climate Change Mitigation. *J. STAR* **2**(4): 40-46.

Aerts, R., Overtveld, K.V., Mitiku Haile, Hermy, M., Deckers, J., Muys, B.(2006). Species composition and diversity of small Afromontane forest fragments in northern Ethiopia. *Plant Ecology* **187**:127–142.

Aerts, R. (1997). Climate, leaf litter chemistry and leaf litter decomposition in terrestrial ecosystems: a triangular relationship. *Oikos* **79**:439–449.

Aerts, R. (1993). Biomass and nutrient dynamics of dominant plant species from heathlands. **In**: *Heathlands: Patterns and Processes in a changing Environment*, pp.51-84 (Aerts, R. and Heil, G.W. eds). Kluwer Academic Publishers, Dordrecht.

AGO. (2002). Field measurement procedures for carbon accounting. Bush for Greenhouse Report 2, Australian Greenhouse Office, Canberra, Australia.

Agren, G. I. and Bosatta, E. (1996). Quality: a bridge between theory and experiment in soil organic matter studies. *Oikos* **76**: 522-528.

Aiba, S., Kitayama, K. and Takyu, M. (2004). Habitat associations with topography and canopy structure of tree species in a tropical montane forest on Mount Kinabalu, Borneo. *Plant Ecology* **174**:147–161.

Alemineh Alelign, Demel Teketay, Yonas Yemshaw and Edwards, S. (2007). Diversity and status of regeneration of woody plants on the peninsula of Zegie, northwestern Ethiopia. *Tropical Ecology* **48**(1): 37-49.

Alexandrove, G.A. (2007). Carbon Stock Growth in a Forest Stand: the Power of Age, *Carbon Balance and Management* **2** (4): 1-5.

Allen, S.E. (1974). *Chemical Analysis of Ecological Materials*, second ed. Blackwell Scientific Publications, Oxford, UK.

Alves, L.F., Vieira, S.A., Scaranello, M.A., Camargo, P.B., Santos, F.A.M., Joly, C.A. and Martinelli, L.A. (2010). Forest structure and live aboveground biomass variation along an elevational gradient of tropical Atlantic moist forest (Brazil). *Forest Ecology and Management* **260**: 679-671.

Ambachew Demissie, Singh, B.R., Lal, R. and Strand, L.D. (2012). Leaf litter fall and litter decomposition under Eucalyptus and coniferous plantations in Gambo District, southern Ethiopia. *Soil and Plant Science* **62**: 467- 476.

Anderson, J.M. and Ingram, J.S.I. (1993). *Tropical Soil Biology and fertility*. A hand book of method. 2nd ed. Wallington, UK.

Anderson, J.M., Ineson, P. and Huish, S.A. (1983). Nitrogen and cation mobilization by soil fauna feeding on leaf litter and soil organic matter from deciduous woodlands. *Soil Biology and Biochem*istry **15**: 463–467.

Anderson, J.M and Swift, M.J. (1983). Decomposition in Tropical Forest. **In**: *The Tropical Rain Forest, Ecology and Management*, pp.289-309 (S.L. Sultan, A.C. Chadwick and T.C. Whitmore, eds). Blackwell, Oxford.

Arunachalam, A., Arunachalam, K.M., Prindey, H.N. and Tripath, R.S. (1998). Fine litterfall and nutrient dynamics during forest regrowth in the humid subtropics of North-eastern India. *Forest Ecology and Management* **110**: 209-219.

Ashton, D.H. (1975). Studies of litter in *Eucalyptus regnans* forests. *Australian Journal of Botany*

23: 413-433.

Austin, A.T. and Vitousek, P.M. (2000). Precipitation, decomposition and litter decomposability of *Metrosideros polymorpha* in native forests on Hawaii. *Journal Ecology* **88**:129–138.

Averti, I. and Dominique, N. (2011). Litterfall, accumulation and decomposition in forest groves established on savannah in the Plateau Teke, Central Africa. *Journal of Environmental Science and Technology* **4**:601–610.

Baccini, A., Laporte, N., Goetz, S.J., Sun, M. Dong, H. (2008). A first map of tropical Africa's aboveground biomass derived from satellite imagery. *Environmental Research Letter* **3**:1–9.

Baccini, A.and Asner, G.P. (2013). Improving pantropical forest carbon maps with airborne LiDAR sampling. *Carbon Management* **4**: 591–600.

Baishya, R., Bariks, S.K. and Upadhaya, K. (2009). Distribution pattern of aboveground biomass in natural and plantation forests of humid tropics in northeast India. *Tropical Ecology* **50** (2): 295-304.

Baker, T.R., Phillips, O.L., Malhi, Y., Almeida, S., and Arroyo, L. (2004). Variation in wood density determines spatial patterns in Amazonian forest biomass. *Global Change Biology* **10**: 1–18.

Banin, L., Feldpausch, T.R., Phillips, O.L *et al.* (2012). Cross-continental comparisons of maximum tree height and allometry: testing environmental, structural and floristic drivers. *Global Ecology and Biogeography* **21**: 1179–1190.

Barbour, M.G., Burk, J.H. and Pitts, W.D. (1987). *Terrestrial Plant Ecology*. 2nd Edition. Benjamin Cummings Publishers, New York.

Barkman, J.J. (1979). The investigation of vegetation structure and structure. **In**: *The study of vegetation,* pp. 123-160 (Werger, M.J.A. ed.) Junk, The Hague.

Barnes, B.V., Zak, D.R., Denton, S.R. and Spurr, S.H. (1998). *Forest Ecology*. 4th ed. Wiley. New york, USA. 774p.

Barthlott, W., Kier, G. and Mutke, J. (1999). Biodiversity-The uneven distribution of a treasure. *NNA Reports* 12/1999: 18-28.

Batjes, N.H. (1996). Total carbon and nitrogen in the soils of the world. *Eur. J. Soil Sci*. **47**: 151–163.

Basuki, T.M., van Laake, P.E., Skidmore, A.K. and Hussin, Y.A. (2009). Allometric Equations for Estimating aboveground Biomass in tropical lowland Dipterocarp forests. *Forest Ecology and Management* **257**: 1684-1694.

Bekele Lemma, Kleja, D. B., Olsson, M., & Nilsson, I. (2007). Factors controlling soil organic carbon sequestration under exotic tree plantations: A case study using the CO_2Fix model in southwestern Ethiopia. *Forest Ecology and Management* **252**: 124-131.

Belay Melese, Ensermu Kelbessa and Teshome Soromessa (2014). Forest Carbon Stocks in Woody Plants of Arba Minch Ground Water Forest and its Variations along Environmental Gradients. *Sci. Technol. Arts Res. Journal* **3**(2): 141-147.

Berg, B. and Cortina, J. (1995). Nutrient dynamics in some decomposing leaf and needle litters in a *Pinus sylvestris* forest. *Scandinavian Journal Forest Research* **10**:1–11.

Berg, B. and Laskowski, R. (2006). *Litter Decomposition: A Guide to Carbon and Nutrient Turnover*, Advances in Ecological Research vol. 38. Elsevier, San Diego, CA, 448p.

Berg, B. and McClaugherty, C. (2008). Plant Litter – decomposition, humus formation, carbon sequestration. Springer-Verlag, Berlin, Germany, 340p.

Berg, B. Johansson, M.B. and Meentemeyer, V. (2000). Litter Decomposition in a transect of Norway Spruce forests; Substrate quality and climate control. *Canadian Journal of Forest Research* **30**: 1136-1147.

Berg, B. (2000). Litter decomposition and organic matter turnover in northern forest soils. *Forest Ecology and Management* **133**: 13–22.

Berger, T.W. and Glatzel. G. (1994). Deposition of atmospheric constituents and its impact on nutrient budgets of oak forests (*Quercus petraea and Quercus robur*) in Lower Austria. *Forest Ecology and Management* **70**:183-193.

Bernhard-Reversat and Loumeto, (2002). The Litter System in African Forest tree Plantations. Science Publishers, Inc. USA.

Bernhard-Reversat, F., Main, G., Holl, K., Loumeto, J. and Ngao, J. (2003). Fast disappearance of the water-soluble phenolic fraction in eucalypt leaf litter during laboratory and field experiments. *Applied Soil Ecology* **23**: 273-278.

Bale Eco-Region Sustainable Management Program (BERSMP) (2010). Bale Mt's REDD project: Structuring, marketing and development preparation phase, FARM Africa / SOS Sahel Ethiopia.

Bhattarai, K.R., Vetaas, O.R. and Grytnes, J.A. (2004). Relationship between plant species richness and biomass in an arid Sub -alpine grassland of the central Himalayas, Nepal. *Folia Geobotanica* **39**: 57–71.

Birhanu Kebede (2010). Floristic Composition and Structural Analysis of Gedo Dry Evergreen Montane Forest, West Shewa Zone of Oromia National Regional State, Central Ethiopia. An Msc Thesis (Unpublished), Addis Ababa University, Ethiopia.

Blair, J.M., Parmelee, R.W. and Beare, M.H. (1990). Decay rates, nitrogen fluxes and decomposer communities of single- and mixed-species foliar litter. *Ecology* **71**:1976-1985.

Blair, J.M. (1988). Nitrogen, sulphur and phosphorus dynamics in decomposing deciduous leaf litter in the southern Appalachians. *Soil Biology and Biochemistry* **17**: 827-830.

Blankinship, Joseph C, Pascal A Niklaus, and Bruce a Hungate (2011). A Meta-Analysis of Responses of Soil Biota to Global Change. *Oecologia* **165** (3): 553–565.

Bockheim, J.G., Jepson, E.A. and Helsey, D.M. (1991). Nutrient dynamics in decomposing leaf litter of four tree species in northern Wisconsin. *Canadian Journal of Forest Research*, **21**: 267–286.

Bohm, W. (1979). *Methods of Studying Root Systems*. Springer-Verlag. New York, USA.

Bohra, C.S. Tewari, S.K. and Bhatt, M.D. (2014). Trends in Soil Carbon Stockpile of Three Major Forests along an Altitudinal Gradient in Indian Central Himalaya. *Ecoprint* **21:** 7-13.

Bolstad, P.V., Swank, W.T. and Vose, J. (1998). Predicting Southern Applachian Overstorey Vegetation with Digital terrain data. *Landscape ecology* **13**: 271-283.

Bolstad, P.V. and Vose, J.M.(1998). The effects of terrain position and elevation on the soil carbon in the southern Appalachians. Assessment Methods for Soil Carbon. Lewis Publishers, New York.

Bonnefille, R. and Hamilton A.C. (1986). Quaternary and late Tertiary history of Ethiopia Vegetation. *Symb. Bot. Ups*. **26**:48-63.

Bonnefille, R. (1983). Evidence for a cooler and drier climate in the Ethiopian uplands towards 2.5 Myr. ago. *Nature* **303**: 487 – 491.

Bosatta, E. and Staaf, H. (1982). The control of nitrogen turnover in forest litter. *Oikos* **39**:143–151.

Braun-Blanquet, J. (1932). Plant Sociology: The Study of Plant Communities. (Translated by

G.D. Fuller and H.S. Conard.) McGraw-Hill, New York and London.

Brinson, M.M. (1977). Decomposition and nutrient exchange of litter in an alluvial swamp forest. *Ecology* **58**: 601-609.

Brown , S. and Lugo, A. (1982). The storage and production of organic matter in tropical forests and their role in the global carbon cycle. *Biotropica* **14**: 161-187.

Brown, S. and Lugo, A.E. (1984). Biomass of tropical forests: a new estimate based on forest *Science* **223:** 1290–1293.

Brown, S., Sayant, J., Cannell, M. and Kauppi, P.E. (1996). Mitigation of carbon emissions to the atmosphere by forest management. *Commonwealth Forestry Review* **75:** 80-89.

Brown, S. (2002). Measuring carbon in forests: current status and future challenges *Environmental Pollution* **116:** 363–372.

Brown, S., Gillespie, J.R. and Lugo, A.E. (1989). Biomass estimation methods for tropical forests with application to forest inventory data. *Forest Science* **35**(4): 881–902.

Brown, S. (2002). Measuring carbon in forests: current status and future challenges *Environmental Pollution* **116:** 363–372.

Brown, S. (1997). Estimating biomass and biomass change of tropical forests, a prime*r*. FAO Forestry paper 134, FAO, Rome.

Bruijnzeel, L.A. and Veneklaas, J (1998). Climate Conditions and Tropical Montane forest Productivity: The fog has not lifted yet. *Ecology* **79**: 3-9.

Bubb, K.A., Xu, Z.H., Simpson, J.A. and Saffigna, P.G. (1998). Some nutrient dynamics associated with litterfall and litter decomposition in hoop pine plantations of southeast Queensland, Australia*. Forest Ecology and Manage*ment **110**: 343–352.

Burgess, N.D., Butynski, T.M., Cordeiro, N.J., Doggart, N.H., Fjeldså, J., Howell, K.M., Kilahama, F.B., Loader S.P., Lovett J.C., Mbilinyi B., Menegon M., Moyer D.C., Nashanda E., Perkin A., Rovero, F., Stanley, W.T. and Stuart, S.N. (2007). The biological importance of the Eastern Arc Mountains of Tanzania and Kenya. *Biological Conservation* **134**: 209-231.

Cairns, M.A, Brown, S., Helmer, E.H. and Baumgardner, G.A. (1997). Root biomass allocation in the world' s upland forests. *Oecologia* **111:** 1–11.

Canadell, J.G., Raupach, M.R. and Houghton, R.A. (2009). Anthropogenic CO_2 emissions in Africa. *Biogeosciences* **6**: 463–468.

Cannell, M.G.R. and Milne, R. (1995). Carbon pools and sequestration in forest ecosystems in Britain. *Forestry* **68**: 361 – 378.

Cao, M.K., Zhang, Q.F. and Shugart, H.H. (2001). Dynamic responses of African ecosystem carbon cycling to climate change. *Climate Research* **17**:183-193.

CCB-AR-PDD (2009). (CCB-AR-PDD). Climate, community and biodiversity standards project design document form for afforestation and reforestation project activities. Humbo Ethiopia Assisted Natural Regeneration Project, Ethiopia.

Chapman,C.A. and Chapman, L.J.(1997). Forest regeneration in logged and unlogged forests of Kibale National Park, Uganda. *Biotropica* **29***: 396–412.*

Chave, J., Mechain, M., Burquez, A. Chidumayo, E., Colgan, M., Delitti, W.B.C., Duque, A., Eid, T., Fearnside, P. and Goodman, R.C (2014). Improved allometric models to estimate the aboveground biomass of tropical trees. *Global Change Biology* **20**: 3177-3190.

Chave, J., Navarrete, D., Almeida, S. Alvarez, E., Aragao, L.E.O.C., Bonal, D., Chatelet, P., Silva-Espejo, J.E. and Malhi, Y.(2010). Regional and seasonal patterns of litterfall in tropical South America. *Biogeosciences* **7** (1): 43–55.

Chave, J., Coomes,D., Jansen, S. (2009). Towards a worldwide wood economicsspectrum. *Ecology Letters* **12**: 351–366.

Chave, J., Cairns, M.A., Andalo, C., Brown, S., Chambers, J.Q., Eamus, D., Fölster, H.,Fromard, F., Higuchi, N., Kira, T., Lescure, J.P., Nelson, B.W., Ogawa, H., Puig, H., Riéra, B. and Yamakura, T. (2005). Tree allometry and improved estimation of carbon stocks and balance in tropical forests. *Oecologia* **145**, 87–99.

Chave, J., Condit, R., Aguilar, S., Hernandez, A., Lao, S. and Perez, R. (2004). Error propagation and scaling for tropical forest biomass estimates. *Phil. Trans. R. Soc.* B **359:**409–420.

Chave, J., Riéra, B. and Dubois, M. A. (2001). Estimation of biomass in a neotropical forest of French Guiana: spatial and temporal variability. *Journal of Tropical Ecology* **17** (01): 79-96.

Christensen, T. (1975). Wood litter fall in relation to abscission, environmental factors and the decomposition cycle in Danish Oak forest. *Oikos* **26**: 187-195.

Clark, D.B. and Clark, D.A. (2000). Landscape-scale variation in forest structure and biomass in a tropical rain forest. *Forest Ecology and Management* **137**: 185-198.

Clark, D.B. and Clark D.A.(1996). Abundance, Growth and Mortality of very large trees in Neotropical lowland Rainforest. Forest Ecology and Management 80: 235-244.

Clarke, K.R. (1993). Non-Parametric Multivariate Analayses of Changes in Community Structure. *Australian Journal of Ecology* **18**:117-143.

Clark, J.S. (1991). Disturbance and population structure on the shifting mosaic landscape. *Ecology* **72**: 1119–1137.

Conner, W.H. (1994). Effect of forest management practices on southern forest wetland productivity. *Wetlands* **14**:27-40.

Cortez, J. (1998). Field Decomposition of Leaf litters: relationships between decomposition rates and soil moisture, soil temperature an earthworm activity. *Soil Biology and Biochemistry* **30**: 783-793.

Cotler, H. and Ortega-Larrocea, M.P. (2006). Effects of land use on soil erosion in a tropical dry forest ecosystem, Chamela watershed, Mexico. *Catena* **65**:107 117.

Coulis, M, Hättenschwiler, S., Fromin, N., and David, J.F. (2013). Macroarthropod-microorganism interactions during the decomposition of Mediterranean shrub litter at different moisture levels. *Soil Biology and Biochem*istry: 1–8.

Couteaux, M.M., Bottner, P. and Berg B. (1995). Litter decomposition, climate and litter quality. *Trends in Ecology and Evolution* **10**: 63–66.

Cuevas, E. and Lugo, A.E. (1998). Dynamics of organic matter and nutrient return from litterfall in stands of ten tropical tree plantation species. *Forest Ecology and Management* **112**: 263-279.

Culmsee, H., Leuschner, C., Moser, G. and Pitopang, R. (2010). Forest aboveground biomass along an elevational transect in Sulawesi, Indonesia, and the role of Fagaceae in tropical montane rain forests. *Journal of Biogeography* **37**: 960-974.

Curtis, J.T. and McIntosh, R.P. (1951). An upland Forest Continuum in the prairie forest border region of Wisconsin. *Ecology* **32**: 476-479.

Curtis, J.T. (1959). *The vegetation of Wisconsin*. Madison, University of Wisconsin Press. 657pp.

Das, K. and Ramakrishnan, P. S. (1985). Litter dynamics in Khasi pine (Pinus kesiya Royle ex Gordon) of north-eastern India. *Forest Ecology and Management* **10**: 135-153.

Daws, M.I., Mullins, C.E., Burslem, D.F.R.P., Paton, S.R. and Dalling, J.W. (2002).

Topographic position affects the water regime in a semi deciduous tropical forest in Panama. *Plant and Soil* **238**:79–90.

DeFries, R. S., Houghton, R.A., Hansen, M.C., Field, C.B., Skole, D. and Townshend, J. (2002). Carbon emissions from tropical deforestation and regrowth based on satellite observations for the 1980's and 1990's. *Proceedings of the National Academy of Sciences* **99**: 14256-14261.

Delang, C.O. and Li, W.M. (2013). Ecological Succession on Fallowed Shifting Cultivation Fields, Springer Briefs in Ecology, DOI: 10.1007/978-94-007-5821-6_2.

Delaney, M. Brown, S., Lugo, A.E., Torres-Lezama, A. and Bello Quintero (1997). The distribution of organic carbon in major components of forests located in five life zones in Venezuela. *Journal of Tropical Ecology* **13**: 697–708.

Demel Teketay and Tamrat Bekele (1995). Floristic composition of Wof-Washa natural forest, Central Ethiopia: implications for the conservation of biodiversity. *Feddes Repertorium* **106**: 127–147.

Demel Teketay (1997). Seedling population and Regeneration of woody species in Dry afromontane forests of Ethiopia. *Forest Ecology and Management* **98**: 149-165.

Demel Teketay (1996). Seed Ecology and Regeneration in Dry Afromontane Forests of Ethiopia. Swedish University of Agricultural Sciences, Doctoral thesis. Umeä.

Dereje Denu (2006). Floristic Composition and Ecological Study of Bibita Forest (Gura Ferda), Southwest Ethiopia. MSc Thesis (Unpublished), AAU, Ethiopia.

Desalegn Tadele, Ermias Lulekal, Destaw Damtie, Adane Assefa (2013). Floristic diversity and regeneration status of woody plants in Zengena Forest, a remnant montane forest patch in northwestern Ethiopia. *Journal of Forestry Research*. DOI 10.1007/s11676-013-0420-3.

Devi, A.S. and Yadava, P.S. (2007). Wood and Leaf Decomposition of *Dipterocarpus tubuculatus* roxb. in a tropical deciduous forest ecosystem of Manipur, North East India. *Current Science* **93**(2): 243-246.

Dezzeo, N., Chacón, N., Sanoja, E. and Picón, G. (2004). Changes in soil properties and vegetation characteristics along a forest-savanna gradient in southern Venezuela. *Forest Ecology and Management* **200**: 183-193.

Díaz, S., Hodgson, J.G., Thompson, K., Cabido, M., Cornelissen, J.H.C. and Jalili, A. (2004).

The plant traits that drive ecosystems: evidence from three continents. *Journal of Vegetation Science* **15**: 295–304.

Dinkissa Beche (2011). Floristic Composition, Diversity And Structure of Woody Plant Species in Menagesha Suba State Forest, Central Ethiopia. An Msc Thesis (Unpublished), Addis Ababa University, Ethiopia.

Dirham, R.K.(1998). Altered leaf-litter decomposition rates in tropical forest fragments. *Oecologia* **116**: 397–406.

Dixon, R.K., Houghton, R.A., Solomon, A.M., Trexler M.C. & Wisniewski J. (1994). Carbon pools and flux of global forest ecosystems. *Science* **263**: 185-190.

Djomo, A.N., Knohl, A. and Gravenhorst, G. (2011). Estimations of total ecosystem carbon pools distribution and carbon biomass current annual increment of a moist tropical forest. *Forest Ecology and Management* **261**: 1448–1459.

Djomo, A.N., Ibrahima, A., Saborowski, J. and Gravenhorst, G. (2010). Allometric equations for biomass estimations in Cameroon and pan moist tropical equations including biomass data from Africa. *Forest Ecology and Management* **260**:1873–1885.

Dossa, G.G.O., Paudel, E., Fujinuma, J., Yu, H. and Chutipong, W. (2013). Factors Determining Forest Diversity and Biomass on a Tropical Volcano, Mt. Rinjani, Lombok, Indonesia. *PLoS ONE* **8** (7): e67720. doi:10.1371/journal.pone.0067720.

Duchaufour, P. (1984). *Pedology of the Short: Soil Vegetation and Environment*. Masson, Paris, 220p.

Du, B., Kang, H., Zhu, J.P.P., Yin, S., Zou, Q., Kong, Z.W.F. and Liu, C. (2014). Soil organic carbon stock and chemical composition along an altitude gradient in the Lushan Mountain, subtropical China. *Ecological Research* **29**: 433–439.

EFAP (1994). Ethiopian Forestry Action Program (EFAP). EFAP, Addis Ababa.

EMSA (2016). Ethiopian Meteorological Service Agency, Addis Ababa, Ethiopia.

Ensermu Kelbessa and Teshome Soromessa (2008). Interfaces of regeneration, structure, diversity and uses of some plant species in Bonga Forest: A reservoir for wild coffee gene pool. *SINET: Ethiopian Journal of Science* **31(2):** 121-134.

Ensermu Kelbessa, Sebsebe Demissew, Zerihun Woldu and Edwards, S. (1992). Some threatened Endemic Plants of Ethiopia. *NAPRECA Monograph, Series* **2:**35-55.

Ermias Aynekulu (2011). Forest diversity in fragmented landscapes of northern Ethiopia and implications for conservation, PhD Thesis, Bonn University, Germany.

Ermias Lulekal, Ensermu Kelbessa, Tamirat Bekele and Haile Yineger (2008). Plant species composition and structure of the Mana Angetu Moist Montane Forest, South-Eastern Ethiopia. *Journal of East African Natural History* **97**(2): 165–185.

Eshetu Yirdaw (2002). Restoration of the native woody-species diversity, using plantation species as foster trees, in the degraded highlands of Ethiopia, PHD Thesis, University of Helsinki, Finland.

Everitt, B.S. (1998). *The Cambridge Dictionary of Statistics*. Cambridge University Press, Cambridge. 360p.

Everitt, B. S. (1978). Graphical techniques for multivariate data. London, Heinemann Educ. Books Ltd.

Ewel, J.J. (1976). Litterfall and leaf decomposition in a Tropical Forest Succession in Eastern Guatemala. *The Journal of Ecology* **64** (1): 293-308.

FAO (2011). State of the world's forests. Rome. 179p.

FAO (2010). Global forest resources assessment. Rome. 340p.

FAO (2005). Global Forest Resources Assessment, Progress towards sustainable forest management. Retrieved august, 2014, from www.fao.org.

FAO (1998). *The soil and terrain database for northeastern Africa*. Land and Water Digital Media Series 2. FAO, Rome.

FAO (1997). Estimating biomass and biomass change of tropical forests: primer, Rome, Italy: FAO Forestry Paper No.134.

FAO (1996). Ethiopia: Country report to the FAO International Technical conference on Plant Genetic Resource. Lepzig, Gemany.

Fauset, S., Johnson, M.O., Gloor, M., Baker,T.R., Monteagudo M.A., Roel J.W. Brienen, R.J.W., Feldpausch, T.R (2015). Hyperdominance in Amazonian forest carbon cycling. *Nature Communications* **6**:6857 doi: 10.1038/ncomms7857 (2015).

Fearnside, P.M. (1997). Greenhouse gases from deforestation in Brazilian Amazonia: net committed emissions. *Climatic Change* **35**: 321-360.

Federal Democratic Republic of Ethiopia (2011). Ethiopia's Climate Resilient Green Economy Strategy**.** Addis Ababa, Ethiopia, 199p.

Fekadu Gurmessa, Teshome Soromessa and Ensermu Kelbessa (2013). Floristic Composition and Community Analysis of Komto Afromontane Moist Forest, East Wollega Zone, West Ethiopia. *STAR journal* **2**(2): 58-69.

Feldpausch, T.R., Lloyd, J., Lewis, S.L., Brienen, R.J.W., Gloor, M., Monteagudo Mendoza, A., Lopez-Gonzalez, G., Banin, L., Abu Salim, K., Affum-Baffoe, K. (2012). Tree height integrated into pantropical forest biomass estimates. *Biogeosciences* **9**: 3381–3403.

Fentaw Yimer, Ledin, S. and Abdelkadir, A. (2006). Soil organic carbon and total nitrogen stocks as affected by topographic aspect and vegetation in the Bale Mountains, Ethiopia. *Geoderma* **135**:335–344.

Feyera Abdena (2010). Floristic Composition and Structure Of Chato Natural forest in Horo Guduru Wollega Zone of Oromia Region, West Ethiopia. An MSc Thesis (Unpublished). Addis Ababa University, Ethiopia.

Feyera Senbeta (2006). Biodiversity and Ecology of Afromontane Rainforests with Wild *Coffea arabica* L. Populations in Ethiopia*. Ecology and Development Series No. 38.* Center for Development Research, University of Bonn, Germany.

Feyerea Senbeta and Manfred, D. (2006). Effects of wild coffee management on species diversity in the Afromontane rainforests of Ethiopia*. Forest Ecology and management* **232**: 68-74.

Field, C.B., Behrenfeld, M.J., Randerson, J. and Falkowski (1998). Primary production of the biosphere: integrating terrestrial and oceanic components. *Science* **281**:237–240

Foaham, B. and Jonkers, W.B.J. (1992). *A programme for Tropenbos research in Cameroon*. Final Report, Tropenbos-Cameroon programme, Phase 1. The Tropenbos Foundation, Wageningen, The Netherlands. 181p.

Friis I. and Sebsebe Demissew (2001). Vegetation maps of Ethiopia and Eritrea. A review of existing maps and the need for a new map for the flora of Ethiopia and Eritrea. **In**: *Biodiversity Research in the Horn of Africa Region*, pp. 399-439 (Friis, I. and Ryding, O., eds). Proceedings of the 3rd International symposium on the Flora of Ethiopia and Eritrea at the Carlsberg Academy, Copenhagen.

Friis, I., Sebsebe Demissew and Breugel, P.V. (2011). Atlas of the Potential Vegetation of Ethiopia. The Royal Danish Academy of Sciences and Letters, Copenhagen, Denmark, 306p.

Friis, I., Edwards, S., Ensermu Kelbessa and Sebesbe Demissew (2001). Diversity and endemism in the flora of Ethiopia and Eritrea—what do the published flora volumes tell us? *Biol. Skr.* **54**, 173–193.

Friis, I. (1992). Forest and Forest Trees of Northeast Tropical Africa: Their natural habitats and distribution pattern in Ethiopia, Djibouti and Somalia. *Kew. Bull. Add. Ser.* **15**, 396 pp.

Fu, X.L., Shao, M.A., Wei, X.R. and Horton, R. (2010). Soil organic carbon and total nitrogen as affected by vegetation types in Northern Loess Plateau of China. *Geoderma* **155**: 31–35.

Gairola, S., Sharma, C.M., Ghildiyal, S.K. and Suyal, S. (2011). Live tree biomass and carbon variation along an altitudinal gradient in moist temperate valley slopes of the Garhwal Himalaya (India). *Journal of Current science* **100**(12): 1862-1870.

Garay, I., Nazoa, S. and Abbadie, L. (1986). Study of a mixed forest litter charm and oak. I: Decomposition of organic matter and dynamics of biogenic elements. *Oecol. Gen.,* **7**: 151-169.

Garten J.R. and Hanson, P.J. (2006). Measured Forest Soil Carbon stocks and estimated turnover times along an elevational graidient. *Geoderma* **136**: 342-352.

Gartner, T.B. and Cardon, Z.G (2004). Decomposition Dynamics in Mixed-Species Leaf Litter**.** *Oikos* **104**(2): 230-246.

Gaston, G., Brown, S., Lorenzini, M. and Singh, K.D. (1998). State and change in carbon pools in the forests of Tropical Africa. *Global Change Biology* **4**: 97-114.

Gatzweiler, F.W. (2007). Deforestation of Ethiopia's Afromontane rainforests; Reasons for concern, ZEF Policy Brief No. 7.

Gauch, H.G. and Whittaker, R.H. (1981). Hierarchical classification of community data. *Journal of Ecology* **69**: 537-557.

Gauch, H.G.J. (1982). Multivariate Analysis in Community Structure. Cambridge University Press, Cambridge.

Gawali, A. (2014). Litterfall and carbon, nutrient returns in stands of *Ceiba pentandra* (L.) Gaertn. Subhumid tropics of Eastern India. *African Journal of Wood Science and Forestry* **2**(2): 104-110.

Getachew Tesfaye (2007). Structure, biomass and net primary production in a dry tropical afromontane forest in Ethiopia. Addis Ababa University, Ethiopia; Institute of Biodiversity Conservation, Ethiopia; and Ethiopian Tree Fund Foundation, Ethiopia.

Gibbon, A., Silman, M.R., Malhi, Y., Fisher, J.B., Meir, P., Zimmermann, M., Dargie, G.C., Farfan, W.R. and Garcia, K.C. (2010). Ecosystem carbon storage across the grassland–forest transition in the high Andes of Manu National Park, Peru. *Ecosystems* **13**:1097–1111.

Gibbs, H.K., Brown, S., Niles, J.O., & Foley, J.A. (2007). Monitoring and Estimating Tropical Forest Carbon Stocks: Making REED a Reality, *Environ. Res. Lett.* **2**: 1 – 13.

Gillespie, A.J.R., Brown, S. and Lugo, A.E. (1992). Tropical forest biomass estimation from truncated stand tables. *Forest Ecology and Management* **48**: 69-87.

Gillespie, A.J.R., Grijalva, A. and Farris, C.N. (2000). Diversity, composition, and structure of tropical dry forests in Central America. *Plant Ecology* **147**: 37–47.

Girardin, C. A. J., Malhi, Y., Aragão, L. E. O. C., Mamani, M., Huaraca Huasco, W., Durand, L., Feeleys, K. J., Rapp, J., SilvaEspejo, J. E., Silman, M., Salinas, N. and Whittaker, R. J. (2014). Net primary productivity allocation and cycling of carbon along a tropical forest elevational transect in the Peruvian Andes. *Global Change Biology* **16**: 3176–3192.

Girardin, C.A.J., Malhi, Y., Aragão, L.E.O.C., Mamani, M., Huaraca Huasco, W., Durand, L., Feeleys, K.J., Rapp, J., SilvaEspejo, J. E., Silman, M., Salinas, N. and Whittaker, R. J. (2010). Net primary productivity allocation and cycling of carbon along a tropical forest elevational transect in the Peruvian Andes. *Global Change Biology* **16**: 3176–3192.

Gonzalez, E. (2012). Seasonal patterns of litterfall in the floodplain forest of a large Mediterranean river. *Limnetica* **31**(1): 173-186.

Goodale, C.L., Apps, M.J. and Birdsey, R.A. (2002). Forest carbon sinks in the northern hemisphere. *Ecological Applications* **12**:891–899.

Goodall, D.W. (1954). Objective methods for the classification of vegetation. III. An essay in the use of factor analysis. *Australian Journal of Botany* **2**:304-324.

Gorte, R.W. (2009). Carbon Sequestration in Forests. *CRS Report for Congress*, Congressional Research Service.

Gosz, J.R., Likens, G.E. and Bormann, F.H. (1976). Organic matter and nutrient dynamics of the forest and food floor in the his board book forest. *Oecologia* **22**: 305-320.

Green, R.H. (1979). *Sampling design and statistical methods for environmental biologists*. Wiley-Interscience, New York, Chichester, Brisbane, Toronto.

Greenway, P.J. (1973). A classification of the vegetation of East Africa. *Kirkia* **9**: 1-68.

Griffiths, R.P., Madritch , M.D. and Swanson, A.K. (2009). The effects of topography on forest soil characteristics in the Oregon Cascade Mountains (USA): Implications for the effects of climate change on soil properties. *Forest Ecology and Management* **257**: 1–7.

Gross, K.L., Willig, M.R., Gough, L., Inouye, R. and Cox, S.B. (2000). Patterns of species density and productivity at different spatial scales in herbaceous plant communities. *Oikos* **89**: 417–427.

Grubb, P.J., Lloyd, J.R., Pennington, J.D. and Whitmore, J.C. (1963). A comparison of montane and lowland rain forest in Ecuador. I. The forest structure, physiognomy and floristics. *Journal of Ecology* **51**: 567-601.

Guo, L. and Sims, R.E.H. (2002). Eucalypt litter decomposition and nutrient release under a short rotation forest regime and effluent irrigation treatments in New Zealand: II. Internal effects. *Soil Biology and Biochemistry* **34** (7): 913–922.

Hamere Yohannes, Teshome Soromessa and Mekuria Argaw (2015). Carbon Stock Analysis along Altitudinal Gradient in Gedo Forest: Implications for Forest Management and Climate Change Mitigation. *American Journal of Environmental Protection* **4**(5): 237-244.

Harmon, M.E., Nadelhoffer, K.J. and Blair, J.M. (1999). Measuring decomposition, nutrient turnover, and stores in plant litter. **In**: *Standard soil methods for long-term ecological research,* pp. 202-240 (G.P. Robertson, C.S. Bledsoe, D.C. Coleman, P. Sollins, eds.). Oxford University Press, UK.

Harmon, M.E., Baker, G.A., Spycher, G. and Greene, S.E. (1990). Leaf litter decomposition in the Picea/Tsuga forests of Olympic National Park, Washington, USA. *Forest Ecology and Management* **31**: 55–66.

Harmon, M.E., Franklin, J.F., Swason, F.J., Sollins, P., Gregory, S.V., Lattin, J.D., Anderson, N.H.(1986). *Ecology of Coarse woody debris in Temperate Ecosystem*. Advances in ecological Research, volume 15, 132p.

Harris, R. F. (1981). Effect of water potential on microbial growth and activity. **In**: *Water Potential Relations in Soil Microbiology*, pp.23–95 (Parr, J.F., Gardner, W.R. and Elliott, L.F. eds.). Madison, WI: American Society of Agronomy.

Hasanuzzaman, M. and Hossain, M. (2014). Leaf Litter Decomposition and Nutrient Dynamics Associated with Common Horticultural Cropland Agroforest Tree Species of Bangladesh**.**

International Journal of Forestry Research. http://dx.doi.org/10.1155/2014/805940

Hasegawam, M, and Takeda, H. (1997). Carbon and nutrient dynamics in decomposing pine needle litter in relation to fungal and faunal abundances. *Pedobiologia*, **40**: 171–184.

Heal, O.W., Anderson, J.M. and Swift, M.J. (1997). Plant litter quality and decomposition: an historical overview. **In**: *Driven by nature: plant litter quality and decomposition,* (Cadisch, G. Giller, K.E. eds.). CAB International, Wallingford, UK.

Henry, M. (2010). C stocks and dynamics in Sub Saharan Africa. PhD Thesis, The University of Tuscia.

Henry, M., Besnard, A., Asante, W., Eshun, J., Adu-Bredu, S., Valentini, R., Bernoux, M. and Saint-André, L. (2010). Wood density, phytomass variations within and among trees, and allometric equations in a tropical rainforest of Africa. *Forest Ecology and Management* **260**: 1375–1388.

Henry, M., Picard, N., Trotta, C., Manlay, R.J., Valentini, R., Bernoux, M. & Saint-André, L. (2011). Estimating tree biomass of sub-Saharan African forests: a review of available allometric equations. *Silva Fennica* **45**(3B): 477–569.

Hill, J.L. *and* Curran, P.J. (2003) Area, shape and isolation of tropical forest fragments: effects on tree species diversity and implications for conservation. *Journal of Biogeography* **30***:* 1391–1403.

Hill, M. O. (1979). TWINSPAN - a FORTRAN program for arranging multivariate data in an ordered two-way table by classification of the individuals and attributes. Cornell University, Ithaca, NY.

Hill, M. O. (1973). Reciprocal averaging: an eigenvector method of ordination. *Journal of Ecology* **61**:237-249.

Holdridge, L.R. (1967). *Life Zone Ecology*. Tropical science center, San Jose, Costa Rica.

Hoque, M.M., Kamal, A.H.M., Idris, M.H., Ahmed, H.O., Hoque, A.T.M.. and Masum Billah, M. (2015). Litterfall production in a tropical mangrove of Sarawak, Malaysia. *Zoology and Ecology*: DOI: 10.1080/21658005.2015.1016758.

Hossain, M.R.H., Siddique, M.S., Rahman, M. Z., Hossain, and Hasan, M.M. (2011). Nutrient dynamics associated with leaf litter decomposition of three agroforestry tree species (*Azadirachta indica*, *Dalbergia sissoo* and *Melia azedarach*) of Bangladesh. *Journal of Forestry Research* **22** (4): 577–582.

Houghton, R.A. and Hackler, J.L. (2006). Emissions of carbon from land use change in sub-Saharan Africa. *Journal of Geophysical Research* **111*:*** G02003, doi:10.1029/2005JG000076.

Houghton, R.A., Hall, F. and Goetz, S.J. (2009). Importance of biomass in the global carbon cycle. *Journal of Geophysical Research* **114**: G00E03.

Houghton, R.A. (2005). Aboveground forest biomass and the global carbon balance. *Global Change Biology* **11**: 945- 958.

Houghton, R.A. (2003). Revised estimates of the annual net flux of carbon to the atmosphere from changes in land use and land management 1850-2000. *Tellus Series B-Chemical and Physical Meteorology* **55**: 378-390.

Houghton, R.A. (1999). The annual net flux of carbon to the atmosphere from changes in land use 1850–1990. *Tellus* B**51:** 298–13.

Houghton, R. A. (2007). Balancing the global carbon budget. *Annual Review of Earth and Planetary Sciences* **35**:313–347.

Hoyle, M.C. (1973). Nature and properties of some forest soils in White Mountains of New Hemisphere. USDA Forest Service Research Paper N. E. 260, 18p.

Hughes, R.F., Kauffman, J.B. and Jaramillo, V.J. (1999). Biomass, carbon, and nutrient dynamic of secondary forests in a humid tropical region of Mexico. *Ecology* **80**: 1892–1907.

Hulme, M., Doherty, R., Ngara, T. and New, M. (2005): Global warming and African climate change:a reassessment. In: *Climate Change and Africa*, pp. 29–40 (P. Sum Low, ed), Cambridge University Press, Cambridge, UK.

ICBP, (1992). Putting biodiversity on the map: priority areas for global conservation. International Council for Bird Preservation (ICBP) (Bird Life International), Cambridge, United Kingdom.

IPCC (2007). Summary for Policymakers. Climate Change 2007. The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, Cambridge University Press, Cambridge, United Kingdom and New York, USA.

IPCC (2006). Intergovernmental Panel on Climate Change Guidelines for National Greenhouse Gas Inventories. **In**: *Institute For Global Environmental Strategies* Prepared by the

National Greenhouse Gas Inventories Programme. (Eggleston H. S, Buendia L, Miwa K, Ngara T, and Tanabe, K. eds). Tokyo, Japan.

IPCC (2003). Good practice guidance for land use, land use change and forestry. Institute for Global Environmental Strategies (IGES), Hayama.

IPCC (2001). Climate Change 2001: Working Group I: The Scientific Basis. Cambridge University Press, UK.

IPCC (2000). Land Use, Land-Use Change, and Forestry. Edited by Watson R.T, Nobel I.R, Bolin B, Ravindranath N.H, Verardo D.J, Dokken D.J, Cambridge University Press, Cambridge.

Iverson, L.R., Brown, S., Grainger, A., Prasad, A. and Liu, D. (1993). Carbon sequestration in tropical Asia: an assessment of technically suitable forest lands using geographic information systems analysis. *Climate Research* **3**: 23-38.

Jackson, R.B., Canadell, J., Ehleringer, J.R., Mooney, H.A., Sala, O.E. and Schulze, E.D. (1996). A global analysis of root distributions for terrestrial biomes. *Oecologia* **108**: 389-411.

Jamaluddin, V. and Kumar, B.M. (1999). Litter of multipurpose trees in Kerala, India: variations in the amount, quality, decay rates and release of nutrients. *Forest Ecology and Management* **115**:1–11.

Jenny, H., Gessel, S.P. and Bingham, F.T. (1949). Comparative study of decomposition of organic matter in temperate and tropical regions. *Soil Science* **68**: 419–432.

Jiregna Gindeba, Olsson, M. and Fisseha, I. (2004). Nutrient Composition and Short-term release from *Croton macrostachyus* Del. and *Millettia ferruginea* (Hochst.) Baker leaves. *Biology and fertility of soils* **40** (6): 1-5.

Johansson, M.B. (1995). The chemical composition of needle and leaf litter from Scots Pine, Norway Spruce and White birch in Scandinavian forests. *Forestry* **68** (1): 49-63.

Jongman, R.H., ter Braak, C.J.F. and Van Tongeren, O.F.R. (1987). *Data analysis in community and landscape ecology.* Cambridge University Press, Cambridge.

Kapos, V., Wandelli, E., Camargo, J.L. and Ganade, G. (1997). Edge-related changes in environment and plant responses due to forest fragmentation in Central Amazonia. **In**: *Tropical forest remnants: ecology, management, and conservation of fragmented communities,* Pp. 33–44 (Laurance W. F. and Bierregaard Jr. R. O., eds.). University of

Chicago Press, Chicago, Illinois.

Karsenty, A., Blanco, C. and Dufour, T. (2003). Forest and climate change: instruments related to the United Nations Framework Convention on Climate Change and their potential for sustainable forest management in Africa. Food and Agriculture Organization of the United Nations, Rome.

Keith, H., Jacobsen, K.L. and Raison, R.J. (1997). Effect of soil phosphorus availability, temperature and moisture on soil respiration in *Eucalyptus pauciflora* forest. *Plant and Soil* **190**: 127-141.

Kennard, D.K., Gould, K., Putz, F.E., Fredericksen, T.S. and Morales, F. (2002) Effect of disturbance intensity on regeneration mechanisms in a tropical dry forest. *Forest Ecology and Management* **162**: 197-208.

Kent, M. and Coker, P. (1992). *Vegetation Description and Analysis. A practical approach.* John Wiley and Sons, New York, 363p.

Kenzo, T., Ichie, T., Hattori, D., Itioka, T. and Handa, C. (2009). Developement of allometric relationships for accurate estimation of above- and below-ground biomass in tropical secondary forests in Sarawak, Malaysia. *Journal of Tropical Ecology* **25**: 371-386.

Ketterings, Q.M., Coe, R., Noordwijk, M., Ambagau, Y. and Palm, C.A. (2001). Reducing uncertainty in the use of allometric biomass equations for predicting aboveground tree biomass in mixed secondary forests. *Forest Ecology and Management* **146**: 199-209.

Killham, K., Amato, M. and Ladd, J.N. (1993). Effect of substrate location in soil and soil pore-water regime on carbon turnover. *Soil Biology and Biochemistry* **25**:57–62.

Kimmins, J.P. (1987). *Forest ecology*. New York, Macmillan. 530p.

Kitayama, K. and. Aiba, S.I. (2002). Ecosystem structure and productivity of tropical rain forests along altitudinal gradients with contrasting soil phosphorus pools on Mount Kinabalu, Borneo. *Journal of Ecology* **90**:37–51.

Kituyi, E., Wandiga, S.O., Andreae, M.O. and Helas, G. (2005). Biomass burning in Africa: role in atmospheric change and opportunities for emission mitigation. **In**: *Climate Change and Africa,* pp. 79-89 (Sum Low, P., ed). Cambridge University Press, Cambridge.

Klemmedson, J.O. (1987). Influence of oak in pine forests of central Arizona on selected nutrients of forest floor and soil. *Soil Science Society of America Journal* **51**:1623-1628.

Knoepp, J.D., Coleman, D.C., Crossley, D.A.Jr. and Clark, J.S. (2000). Biological indices of soil quality: an ecosystem case study of their use. *Forest Ecology and Management* **138**: 357-368.

Köhl, M., Baldauf, T., Plugge, D. and Krug, J. (2009). Reduced emissions from deforestation and forest degradation (REDD): A climate change mitigation strategy on a critical track. Carbon Balance Management **4**:1–10.

Kononova, M.M. (1975). Humus of virgin and cultivated soils. **In:** *Soil components*, pp.475-526 (J.E. Gieseking, ed). Vol.1, Springer-Verlag, New York.

Koppad, A.G and Tikhile, P. (2015). Effect of Aspect and Altitude on Organic Carbon Pool in forest Soils of Ankola Taluka. *The Indian Forester* **141**(5):

Korner, C. (2007). The use of altitude in ecological research. *Trends in Ecology and Evolution* **22**: 569–574.

Kumar, S. and Tewari, L.M. (2014). Pattern of litter fall and litter decomposition in a *Quercus leucotrichophora* A. Camus forest in Kumaun Himalaya. *International Journal of Biodiversity and Conservation* **6** (1): 108-114.

Kuper, W., Sommer, J.H., Lovett, J.C., Mutke, J., Linder, H.P.,Beentje, H.J., van Rompaey, R., Chatelain, C., Sosef, M. and Barthlott, W. (2004). Africa's hotspots of biodiversity redefined. *Annals of the Missouri Botanical Garden* **91**: 525–535.

Kwabiah, A.B., Stoskopf, N.C., Voroneyz, R,P. and Palm, C.A. (2001). Nitrogen and Phosphorus Release from Decomposing Leaves under Sub-Humid Tropical Conditions. *Biotropica* **33**(2): 229-240.

Lal, R. (2010). Manageing Soils and Ecosystems for Mitigating anthropogenic carbon emissions and Advancing Global Food security. *BioScience* **60**: 708–721.

Lal, R. (2008). Carbon Sequestration. *Phil. Trans. R. Soc. B.* **363**: 815–830.

Lal, R. (2006). World crop residues production and implication of its use as a biofuel. *Env. Intl.* **31**: 575-586.

Lal, R. (2005). Forest Soils and Carbon sequestration. *Forest Ecology and Management* **220**: 242-258.

Lal, R. (2004). Soil Carbon Sequestration Impacts on Global Climate Change and Food Security. *Science* **34**: 1623-1627.

Lal, R. (2001). Potential of desertification control to sequester carbon and mitigate the greenhouse effect. *Climate Change* **51**: 35-72.

Latter, P.M., Howson, G., Howard, D.M. and Scott, W.A. (1998). Long-term study of litter decomposition on a Pennine peat bog: which regression? *Oecologia* **113**: 94-103.

Laurance, W.F., Ferreira, L.V., Rankin-de Merona, J.M., Laurance, S.G., Hutchings, R.W.and Lovejoy, T.E. (1998). Effects of forest fragmentation on recruitment patterns in Amazonian tree communities. *Conservation Biology* **12**:460–464.

Laurance, W.F., Camargo, J.L.C., Luizão, R.C.C., Laurance, S.G., Pimm, S.L., Bruna, E.M. (2011). The fate of Amazonian forest fragments: A 32-year investigation. *Biological Conservation* **144**: 56-567.

Laurance, W.F. (2008). Theory meets reality: how habitat fragmentation research has transcended island biogeographic theory. *Biological Conservation* **141**: 1731–1744.

Laurance, W.F., Fearnside, P.M., Laurance, S.G., Delamonica, P., Lovejoy, T.E., Rankin-de Merona, J.M., Chambers, J.Q. and Gascon, C. (1999). Relationship between soils and Amazon forest biomass: a landscape scale study. *Forest Ecolology and Management* **118**:127–138.

Lavelle, E., Blanchart, E., Martin, A., and Martin, S. (1993) A hierarchical model for decomposition in terrestrial ecosystems: application to soils of the humid tropics. *Biotropica* **25**:130–150.

Leemans, R.(1989). Description and simulation of stand structure and dynamics in some Swedish forests. University of Uppsala, Uppsala, Sweden

Lemessa Kumsa (2010). Floristic Composition and Structure of Gura-Lopho Moist Afromontane Forest, Horo-Guduru Wollega Zone, Oromia National Regional State, West Ethiopia. MSc Thesis (Unpublished), Addis Ababa University, Ethiopia.

Lewis, S. L., Sonke, B., Sunderland, T., Begne, S.K., LopezGonzalez, G., van der Heijden, G.M.F., Phillips, O.L., AffumBaffoe, K., Baker, T.R., Banin, L., Bastin, J.F., Beeckman, H., Boeckx, P., Bogaert, J., De Canniere, C., Chezeaux E. (2013). Above-ground biomass and structure of 260 African tropical forests. *Phil. T. R. Soc. B*, 368, 1625, doi:10.1098/rstb.2012.0295.

Lewis, S.L., Lopez-Gonzalez, G., Sonké, B. Votere, R., Wöll, H. (2009). Increasing carbon storage in intact African tropical forests. *Nature* **457**: 1003–1006.

Lieberman, D., Lieberman, M., Peralta, R. and Harthorn, G.S. (1996). Tropical forest structure and composition on a largescale altitudinal gradient in Costa Rica. *Journal of Ecology* **84**, 137–152.

Likens, G.E. and Bormann, F.H. (1995). Biogeochemistry of a forested ecosystem. Springer-Verlag, New York, 162p.

Lin, Y. M., Liu, J. W., Xiang, P., Lin, P., Ding, Z. H. and Sternberg, S. (2007). Tannins and nitrogen dynamics in mangrove leaves at different age and decay stages (Jiulong River Estuary, China). *Hydrobiologia* **583** (1): 285–295.

Lindner, M., Maroschek, M., Netherer,S., Kremer, A., Barbati, A., Garcia-Gonzaloa, J., Seidl, R., Delzond, S., Coronae, P., Kolstroma, M., Lexer, M.J. and Marchetti, M. (2010). Climate change impacts, adaptive capacity, and vulnerability of European forest ecosystems. *Forest Ecology and Management* **259**: 698–709.

Lisanework Nigatu and Michelsen, A. (1994). Litterfall and nutrient release by decomposition in three plantations compared with a natural forest in the Ethiopian highland. *Forest Ecology and Management* ***65***:149-164.

Liski J, Nissinen A, Erhard M. and Taskinen, O.(2003). Climate effects on litter decomposition from arctic tundra to tropical rainforest. *Global Change Biology* **9**:575–84.

Liu, Q., Peng, S. L., Bi, H., Zhang, H. Y., Li, Z. A., Ma, W. H. & Li, N. Y. (2005). Decomposition of leaf litter in tropical and subtropical forests of southern China. *Journal of Tropical Forest Science* **17** (4): 543—556.

Liu, L. (2012). Patterns of litterfall and nutrient return at different altitudes in evergreen hardwood

forests of Central Taiwan. *Annals of Forest Science* **69** (8): 877-886.

Lodge, D.J., Scatena, F.N., Asbury, C.E., Sánchez, M.J. (1991). Fine litterfall and related nutrient inputs resulting from Hurricane Hugo in subtropical wet and lower montane rain forests of Puerto Rico. *Biotropica* **23**: 336–342.

Logan, W.E.M. (1946). An Introduction to the forests of Central and Southern Ethiopia. Imperial Forestry Institute, University of Oxford. Inst. Paper No. 24, 58 pp.

Lonsdale, W. (1988). Predicting the amount of litterfall in forests of the world. *Annals of Botany* **61**:319 –324.

Lu, D. (2005). Aboveground biomass estimation using Landsat TM data in the Brazilian

Amazon. *Int. J. Remote Sensing* **26** 2509–25.

Lugo, A. and Brown, S. (1992). Tropical forests as sinks of atmospheric carbon. *Forest Ecology and Management* **54**: 239-255.

Luo, Y. and Zhou, X. (2006). *Soil respiration and the Environment*. Elsevier Inc., 316p.

Magurran, A.E. (2004). *Measuring biological diversity*. Blackwell Publishing, Oxford.

Mahmood, H., Siddique, M.R.H. and Abdullah, S.M.R. (2014). Nutrient dynamics associated with leaching and microbial decomposition of four abundant mangrove species leaf litter of the Sundarbans, Bangladesh. *Wetlands* **34** (3): 439–448.

Mahmood, H., Saberi, O., Misri, K. and Japar Sidik, B. (2007). Nutrients dynamics associated with leaf litter degradation of *Bruguieria parviflora* (Whight and Arnold) at Kuala Selangor Mangrove Forest, Malaysia. *Indian Journal of Forestry* **30**: 325–330.

Malhi, Y., Silman, M., Salinas, N., Bush, M., Meir, P. and Saatchi, S. (2010). Elevation gradients in the tropics: laboratories for ecosystems ecology and global change research. *Global Change Biology* **16**: 3171–3175.

Malhi, Y., Aragão, L.E.O.C., Metcalfe, D.B., Paiva, R., Quesada, C.A., Almeida, S., Anderson, L., Brando, P., Chambers, J.Q., Da Costa, A.C.L., Hutyra, L.R., Oliveira, P., Patiño, S., Pyle, E.H., Robertson, A.L. and Teixeira, L.M. (2009). Comprehensive assessment of carbon productivity, allocation and storage in three Amazonian forests. *Global Change Biology* **15**: 1255–1274.

Malhi, Y., Meir, P. and Brown, S. (2002). Forests, Carbon and Global Climate. *Phil. Trans. R. Soc. Lond*. **360**: 1567–1591.

Malhi,Y., Baldocchi, D.D. and Jarvis, P.G. (1999). The carbon balance of tropical, temperate and boreal forests. *Plant, Cell and Environment* **22**:715–740.

Mamo Kebede, Eshetu Yirdaw, Luukkanen, O. and Mulugeta Lemenih (2013). Plant Community Analysis and Effect of Environmental factors on the diversity of woody species in the moist Afromontane forest of Wondo Genet, South Central Ethiopia. *Biodiversity Research and Conservation* **29**: 63-80.

Måren, I.E., Karki, S., Prajapati, C., Yadav, R.K. and Shrestha, B.B. (2015). Facing north or south: Does slope aspect impact forest stand characteristics and soil properties in a semiarid trans-Himalayan valley?

Maren, I.E. and Vetaas, O.R. (2007). Does Regulated Landuse allow Regeneration of Keystone species in the Annapurna conservation area, Central Himalaya. *Mountain Research Development* **27**(4): 345-351.

Marshall, A.R., Willcock, S., Platts, P.J., Lovett, J.C., Balmford, A., Burgess, N.D., Latham, J. E., Munishi, P.K.T., Salter, R., Shirima, D.D. and Lewis, S.L. (2012). Measuring and modelling above-ground carbon and tree allometry along a tropical elevation gradient. *Biological Conservation* **154**:20–33.

McKinley, D.C., Ryan, M.G., Birdsey, R.A., Giardina, C.P., Harmon, M.E. (2011). A synthesis of current knowledge on forests and carbon storage in the United States. *Ecological Applications* **21**:1902–1924.

Meentemeyer, V. (1978). Macroclimate and lignin control of litter decomposition rates. *Ecology* **59**: 465-472.

MEFCC (2015). Ministry of Environment, Forest and Climate Change, Addis Ababa, Ethiopia.

Megonigal, P.J. and Day, Jr. (1988). Organic matter dynamics in four seasonally flooded forest communities of the Dismal Swamp. *American Journal of Botany* **75**:1334-1343.

Mekonnen Biru (2003). An Ecological Study of Biteyu Forest, Gurage Zone, Southern Nations, Nationalities Peoples Region. MSc Thesis (unpublished), Addis Ababa University, 88p.

Melillo, J.M., McGuire, A.D., Kicklighter, D.W., Moore, B., Vorosmarty, C.J. and Schloss, A.L. (1993). Global climate change and terrestrial net primary production. *Nature* **363**:234–240.

Mesfin Sahle (2011). Estimating and Mapping of Carbon Stocks based on Remote Sensing, GIS and Ground Survey in the Menagesha Suba State Forest, Ethiopia. MSc. Thesis (Unpublished), Addis Ababa University, Ethiopia.

Mesfin Tadesse (1993). The Physiognomy and Floristic Composition of the Vegetation on some degraded hill sides in Southern Wello, Ethiopia. *Opera Botanica* **121**:61-70.

Millington, A.C., Critchley, R.W., Douglas, T.D. and Ryan, P. (1994). Evaluation de la Biomasse Ligneuse en Afn'que Subsaharienne, World Bank, Washington, D.C., 215p.

Million Bekele (2001). Forestry Outlook Studies in Africa (FOSA), Ethiopia. 39p.

MOA and EPA (2013). Terms of Reference for Developing Capacities for a national Measuring, Monitoring, Reporting and Verification System to support REDD+ participation of Ethiopia. Background, Capacity Assessment and Roadmap, 37p.

Mohammed Gedefaw, Teshome Soromessa and Belliethathan, S. (2014). Forest Carbon Stocks in Woody Plants of Tara Gedam Forest: Implication for Climate Change Mitigation. *J.STAR* **3**(1): 101-107.

Mokany, K., Raison, R..and Prokushkin, A.S. (2006). Critical analysis of root: shoot ratios in terrestrial biomes. *Global Change Biology* **12**(1): 84-96.

Montagnini, F. And Jordan, C.F.(2002). Reciclagem de nutrientes. **In:** *Ecología y conservación de bosques neotropicales*. LUR, Cartago, p.167-191.

Montagu, K.D., Düttmer, K., Barton, C.M. and Cowie, A.L. (2005). Developing general allometric relationships for regional estimates of carbon sequestration-an example using *Eucalyptus pilularis* from seven contrasting sites. *Forest Ecology and Management* **204**: 115-129.

Montès, N., Gauquelin, T., Badri, W., Bertaudiere, V. and Zaoui, E.H. (2000). A non destructive method for estimating above-ground forest biomass in threatened woodlands. *Forest Ecology and Management* **130**: 37-46.

Moore, T.R., Trofymow, J.A., Prescott, C.E., Fyles, J. and Titus, B.D. (2006). Patterns of carbon, nitrogen and phosphorus dynamics in decomposing foliar litter in Canadian forests. *Ecosystems* **9**: 46–62.

Morgan, J.A., Follett, R.F., Allen, L.H., Del Grosso, S., Derner, J.S., Dijkstra, F., Franzluebbers, A., Fry, R., Paustian, K. and Schoeneberger, M. (2010). Carbon sequestration in agricultural lands of the United States. *Journal of Soil and Water Conservation* **65**: 6A–13A

Morrison, I.K. (1991). Addition of Organic matter and Elements to the Forest floor of an Old growth *Accer saccharium* forest in the annual litter fall. *Canadian Journal of Forest Research* **9**: 19-26.

Moser, G., Leuschner, C., Hertel, D., Graefe, S., Soethe, N. and Iost, S. (2011). Elevation effects on the carbon budget of tropical mountain forests (S Ecuador): the role of the belowground compartment. *Global Change Biology* **17**:2211–2226.

Moser, G., Roderstein, M., Soethe, N., Hertel, D. and Leuschner, C. (2008). Altitudinal changes in stand structure and biomass allocation of tropical mountain forests in relation to microclimate and soil chemistry. *Ecological Studies* **198**: 229–242.

Moser, G., Hertel, D. and Leuschner, C. (2007). Altitudinal change in LAI and stand leaf biomass in tropical montane forests: a transect study in Ecuador and a pan-tropical meta analysis. *Journal of Ecosystems* **10**: 924–935.

Mudrick, D.A., Hoosein, M., Hicks, R.R. and Townsend, E.C. (1994). Decomposition of leaf litter in an Appalachian forest: effects of leaf species, aspect, slope position, and time. *Forest Ecology and Management* **68**: 231–250.

Muller-Dombois, D. and Ellenberg, H. (1974). *Aims and Methods of Vegetation Ecology*. John Wiley and Jons inc. USA.

Muluken Nega, Teshome Soromessa and Eyale Bayable (2015). Carbon Stock in Adaba-Dodola Community Forest of Danaba district, West Arsi zone of Oromia Region, Ethiopia: An implication for Climate Change Mitigation. *Journal of Ecology and the Natural Environment* **7**(1): 14-22.

Munishi, P.K.T. and Shirima, D.D. (2009). Valuation, mapping and conservation of carbon stocks and other ecosystem services I: The eastern arc mountains of Tanzania. University of Leeds, UK and Sokoine University of Agriculture, Tanzania.

Muoghalu, J.I., Akanni, S.O. and Eretan, O.O.(1993): Litterfall and nutrient dynamics in a Nigerian rainforest seven years after a ground fire. *Journal of vegetation science* **4**: 323-328.

Murphy, P.G. and Lugo, A.E. (1986). Ecology of Tropical Dry Forest**.** *Annual Review of Ecology and Systematics* **17**: 67-88.

Mutuo, K.P., Cadisch, A., Albrecht, C.A., Palm and Verchot, L. (2005). Potential of agroforestry for carbon sequestration and mitigation of greenhouse gas emissions from soils in the tropics. *Nutrient Cycling in Agroecosystems* **71:** 43–54.

Mwakisunga, B. and Majule, A.E. (2012). The influence of altitude and management on carbon stock quantities in rungwe forest, southern highland of Tanzania. *Open Journal of Ecology* **2**(4).214-221.

Mwampamba, T.H. (2009). Forest Recovery and Carbon Sequestration Under Shifting Cultivation in the Eastern Arc Mountains, Tanzania: Landscape and Land-use Effects. PhD Thesis (Unpublished), University of California, USA.

Nair, P.K.R., Kumar, B. and Nair, V. (2009). Agroforestry as a strategy for carbon sequestration. *Journal of Plant Nutrition and Soil Science* **172**: 10-23.

Navár, J. (2009). Allometric equations for tree species and carbon stocks for forests of Northwestern Mexico. *Forest Ecology and Management* **257**: 427-434.

Nelson, B.W., Mesquita, R., Pereira, J.L.G., de Souza, S.G.A., Batista, G.T. and Couto, L.B. (1999). Allometric regressions for improved estimate of secondary forest biomass in the central Amazon. *Forest Ecology and Management* **117**:149–167.

NIACS (2015). Northern Institute of Applied Climate Science, http://www.nrs.fs.fed.us/niacs/carbon/forests/.

NMSA (2016). National Metereological Service Agency, Ethiopia.

Nsabimana, D. (2009). Carbon stock and fluxes in Nyungwe forest and Ruhande Arboretum in Rwanda. A PhD Thesis, University Of Gothenburg, Germany.

Odiwe, A.I. and Muoghalu, J.I. (2003). Litterfall dynamics and forest floor litter as influenced by fire in a secondary lowland rain forest in Nigeria. *Tropical Ecology* **44**: 241–248.

Okeke, A.I. and Omaliko, C.P.E. (1992). Leaf litter decomposition and carbon dioxide evolution of some agroforestry fallow species in southern Nigeria. *Forest Ecology and Management* **50**: 103–116.

Oksanen, J. (2012). Constrained Ordination: Tutorial with R and vegan, 27p. cc.oulu.fi/~jarioksa/opetus/metodi/sessio2.pdf (accessed on April 2016).

Olson, J.S. (1963). Energy storage and balance of producers and decomposers in ecological systems. *Ecology* **44**: 322–331.

O'Neill, E.G., Johnson, D.W., Ledford, J. (2003). Acute seasonal drought does not permanently alter mass loss and nitrogen dynamics during decomposition of red maple litter. *Global Change Biology* **9**:117–23.

Oostra, S., Majdi, H. and Olsson, M. (2006). Impact of tree species on soil carbon stocks and soil acidity in southern Sweden. *Scandinavian Journal of Forest Research* **21**:364–371.

Osono, T. and Takeda, H. (2004). Potassium, calcium, and magnesium dynamics during litter decomposition in a cool temperate forest. *Journal of Forestry Research* **9**:23–31.

Oziegbe, M.B., Muoghalu, J.I. and Oke, S.O. (2011). Litterfall, precipitation and nutrient fluxes in a secondary lowland rain forest in Ile–Ife, Nigeria. *Acta Botanica Brasilica* **25**(3): 664-671.

Page,S. E., Siegert, F., Rieley, J.O., Hans-Dieter, V.B., Jaya, A. and Limin, S. (2002).The amount of carbon released from peat and forest fires in Indonesia during 1997. *Nature* **420:** 61–65.

Palmer, M.W.(1993). Putting things in even better order: the advantages of canonical correspondence analysis. *Ecology* **74**:2215-2230.

Palmer, M.W. (1986). Pattern in corticolous bryophyte communities of the North Carolina piedmont: Do mosses see the forest or the trees? *Bryologist* **89**:59-65

Pan, Y., Birdsey, R.A., Fang, J., Houghton, R., Kauppi, P.E., Kurz, W.A., Phillips, O.L., Shvidenko, A., Lewis, S.L., Canadell, J.G., Ciais, P., Jackson, R..B., Pacala, S., Mcuire, A. D., Piao, S., Rautiainen, A., Sitch, S. and Hayes, D. (2011). A Large and Persistent Carbon Sink in the World's Forest. *Science* **333**: 988-993.

Pandey, R.R. Sharma, G. Tripathi, S.K. and Singh, A.K. (2007). Litterfall, litter decomposition and nutrient dynamics in a subtropical natural oak forest and managed plantation in Northeastern India. *Forest Ecology and Management* **240**(1–3): 96–104.

Parker, G.G. (1983).Throughfall and stemflow in the forest nutrient cycle. *Advances in Ecological Research* **13**:57–133.

Parsons, S.A., Valdez-Ramirez,V., Congdon, R.A. and Williams, S. E. (2014). Contrasting patterns of litterfall seasonality and seasonal changes in litter decomposability in a tropical rainforest region. *Biogeosciences* **11**: 5047–5056.

Parton, W.J., Schimel, D.S., Cole, C.V. and Ojima, D.S. (1987). Analysis of factors controlling soil organic matter levels in Great Plains grasslands. *Soil Science Society of America Journal* **51**: 1173–1179.

Paul, C. (1993). *Ecology* 2. John Wiley and Sons Inc. New York. Brisbane. Toronto. Singapore.

Pearson, T., Brown, S. and Birdsey, R. (2007). Measurement Guidelines for the Sequestration of Forest Carbon. United States Department of Agriculture, General Technical Report NRS-18, USA.

Pearson, T., Walker, S. and Brown, S. (2005). Sourcebook for Land Use, Land-Use Change and Forestry Projects, Winrock International, USA.

Peet, R.K. (1980). Ordination as a tool for analyzing complex data sets. *Vegetatio* **42**:171-4

Penman, J., Gytarsky,M., Hiraishi, T., Krug, T., Kruger, D., Pipatti, R., Buendia, L., Miwa, K., Ngara,T., Tanabe, K. and Wagner, F. (2003). Good practice guidance for land use, land-use change and forestry. IPCC National Greenhouse Gas Inventories Programme and Institute for Global Environmental Strategies, Kanagawa.

Peterjohn, W.T., Melillo, J.M., Steudler, P.A. and Bowles, F.P. (1994). Responses of trace gas fluxes and N availability to experimentally elevated soil temperatures. *Ecological Applications* **4**: 617–625.

Phillipson, J., Putman, R.J., Steel, J and Woodell, S.R.J. (1975). Litter input, litter decomposition and the evolution of carbon dioxide in a beech woodland-Wytham Woods, Oxford. *Oecologia* **20**: 203–217.

Pielou, E.C. (1984). *The Interpretation of Ecological Data*: A Primer on Classification and Ordination. Wiley, New York.

Pielou, E.C. (1977). *Mathematical ecology*. John Wiley and Sons, New York.

Plumptre, A. (1996). Changes following 60 years of selective timber harvesting in the Budongo Forest Reserve, Uganda. *Forest Ecology and Management* **89**: 101–113.

Ponce-Hernandez, R. (2004). Assessing carbon stocks and modelling win-win scenarios of carbon sequestration through land-use changes. FAO, Rome, Italy.

Poore, M.E.D. (1962). The method of Successive Approximation in descriptive ecology. *Advances in ecological Research* **1**: 35-38.

Post, W.M., Emanuel, W.R., Zinke, P.J. and Stangenberger, A.G.(1982). Soil carbon pools and world life zones. *Nature* **298**: 155–159.

Post, W.M., Peng, T.H. Emanuel, W.R. King, A.W. Dale, V.H. and DeAngelis, D.L. (1990). The global carbon cycle. *American Scientist* **78**: 310-326.

Prescott, C. E. (2005). Do rates of litter decomposition tell us anything we really need to know? *Forest Ecology and Management* **220**: 66-74.

Prescott, C.E., Blevins, L.L and Staley, C. L. (2000). Effects of clearcutting on decomposition rates of litter and humus in forests of British Columbia. *Canadian Journal of Forest Research* **30**: 1751–1757.

Proctor, J. (1987). Nutrient Cycling in primary and old secondary rainforests. *Applied Geography* **7**: 135-152.

Proctor, J. (1983). Tropical forest litterfall I: Problems of data comparison. **In**: *Tropical rain forest: Ecology and management*, pp. 267- 273(S.L. Sutton, T.C. Whitmore & Chadwick, A.C, eds). Blackwell, Oxford.

Qiu X.Z., Xie, S.C. and Liu, W.Y. (1998). *Studies on the Forest Ecosystem in Ailao Mountains, Yunnan, China*. Sciences and Technology Press, Kumming.

R Core Team (2013). A language and environment for statistical computing, R Foundation for

Statistical Computing, Vienna, Austria.

Rahbek, C. (2005). The role of spatial scale and the perception of large-scale species-richness patterns. *Ecology Letters* **8**: 224–239.

Rahbek, C. (1995). The elevational gradient of species richness: a uniform pattern? *Ecography* **18**: 200–205.

Rai S.N. and Proctor, J. (1986). Ecological studies on four rainforests in Karnataka, India: II. Litterfall. *Journal Ecology* **74**:455–463.

Raich, J.W., Russell, A.E., Kitayama, K., Parton, W.J. and Vitousek, P.M. (2006). Temperature influences carbon accumulation in moist tropical forests. *Ecology* **87**: 76–87.

Ramachandran, A., Jayakumar, S., Haroon, R.M., Bhaskaran, A. and Arockiasamy, D.I. (2007). Carbon sequestration: estimation of carbon stock in natural forests using geospatial technology in the Eastern Ghats of Tamil Nadu, India. *Current Science* **92**: 10-20.

Ranger, J., Gerard, F., Lindemann, M., Gelhaye, D. and Gelhaye, L. (2003). Dynamics of litterfall in a chronosequence of Douglas-fir (*Pseudotsuga menziesii* Franco) stands in the Beaujolais mounts (France). *Annual Forestry Science* **60**: 475-488.

Rapport, D.J., Regier, H.A. and Hutchinson, T.C. (1985). Ecosystem behavior under stress**.** *Am. Nat.* **125**: 617–640.

Ravindranath, N.H. and Ostwald, M. (2008). Methods for estimating above-ground biomass. **In:** *Carbon Inventory Methods: Handbook for greenhouse gas inventory, carbon mitigation and round wood production projects, pp. 113-128 (*N.H. Ravindranath, and M. Ostwald, eds). Springer Science, New York.

Ravindranath, N.H., Somashekhar, B.S. and Gadgil, M. (1997). Carbon flow in India forests. *Climatic Change* **35**: 297-320.

Reusing, M. (1998). Monitoring of forest resources in Ethiopia. Government of the Federal Democratic Republic of Ethiopia, Ministry of Agriculture, Natural Resources Management and Regulatory Department and German Agency for Technical Cooperation (GTZ), Addis Ababa.

Ribeiro, C., Madeira, M. and Araujo, M.C. (2002). Decomposition and nutrient release from leaf litter of Eucalyptus globules grown under different water and nutrient regimes. *Forest Ecology and Management* **171**: 31–41.

Richards, P.W. (1996). *The Tropical Rain Forest*. 2nd edition. Cambridge University Press, Cambridge.

Richter, D.D., Markewitz, D., Dunsomb, J.K., Wells, C.G., Stuanes, A., Allen, H.L., Ureego, B., Harrison, K. and Bonani, G. (1995). Carbon cycling in a loblobby pine forest: Implication for the missing carbon sink and for the concept of soil. **In**: *Carbon Forms and Function in Forest Soils*, pp. 223-251(W.W. McFee and J.L. Kelly eds). Soil Science Society of America, Madison, WI.

Ringius, L. (2002). Soil Carbon Sequestration and the CDM: Opportunities and Challenges for Africa**.** *Climatic Change* **54:** 471–495.

Robinson, C.H; Wookey, P.A., Parsons, A.N., Potter, J.A., Callaghan, T.V., Lee, J.A., Press, M.C., Welker, J.M. (1995). Responses of plant litter decomposition and nitrogen mineralisation to simulated environmental change in a high arctic polar semi-desert and a subarctic dwarf shrub heath. *Oikos* **74**(3): 503-512.

Roderstein, M., Hertel, D. and Leuschner, C. (2005). Above and below-ground litter production in three tropical montane forests in Southern Ecuador. *Journal of Tropical Ecology* **21**: 483–492.

Ryan, C.M., Williams, M. and Grace, J. (2011). Above and belowground carbon stocks in a Miombo woodland landscape of Mozambique. *Biotropica* **43**: 423-432.

Reyes, G., Brown, S., Chapman, J. and Lugo, A.E. (1992).Wood densities of Tropical Tree Species. General Technical Report SO-88. USDA Forest Service, Southern Forest Experiment Station, New Orleans, USA.

Saatchi, S.S., Harris, N.L., Brown S., Lefsky, M., Mitchard, E.T.A., Salas, W., Zutta, B. R., Buermann, W., Lewis, S. L., Hagen, S., Petrova, S., White, L., Silman, M. and Morel, A. (2011). Benchmark map of forest carbon stocks in tropical regions across three continents. *P. Natl. Acad. Sci.* **108**: 9899– 9904.

Saeed, S., Younus, M., Barozai, K., Ahmad, A. and Shah, S.H. (2014). Impact of Altitude on Soil Physical and Chemical Properties in Sra Ghurgai (Takatu mountain range) Quetta, Balochistan. *International Journal of Scientific and Engineering Research* **5** (3): 730-735.

Salinas, N., Malhi, Y., Meir, P., Silman, M., Roman Cuesta, R., Huaman, J., Salinas, D., Huaman, V., Gibaja, A., Mamani, M. and Farfan, F.(2011). The sensitivity of tropical leaf litter decomposition to temperature: results from a large-scale leaf translocation experiment along an elevation gradient in Peruvian forests. *New Phytologist* **189**:967–977.

Sanches, L., Valentini, C.M.A., Junior, O.B.P., Nogueira, J.S., Vourlitis, G.L., Biudes, M.S. da Silva, C.J., Bambi, P. and Lobo, F.A. (2008). Seasonal and interannual litter dynamics of a tropical semideciduous forest of the southern Amazon Basin, Brazil. *Journal of Geophysical Research* **113**: G04007, doi:10.1029/2007JG000593.

Sangha, K.K, Jalota, R.K and Midmore, D. J. (2006). Litter production, decomposition and nutrient
release in cleared Queensland, Australia. *Journal of Tropical Ecol*ogy **22**: 177-189.

Sanjit, L. and Bhatt, D. (2005). How relevant are the concept of Species Diversity and Species Richness? *Journal of Bioscience* **30**(5): 557-560.

Santantonio, D., Hermann, R.K. and Overton, W.S.(1997). Root Biomass Studies in forest ecosystems. *Pedobiologia* **17**: 1-31.

Sapkota, I.P., Tigabu, M. and Oden, P.C. (2010) Changes in tree species diversity and dominance across a disturbance gradient in Nepalese Sal (*Shorea robusta* Gaertn. f.) forests. *Journal of Forestry Research* **21**(1): 25-32.

Sariyildiz, T., Anderson, J.M. and Kucuk, M. (2005). Effects of Tree Species and Topography on soil chemistry, litter quality and decomposition in Northeast Turkey. *Soil Biology and Biochemistry* **37**: 1695 – 1706.

Schaeffer, D.J., Herricks, E.E. and Kerster, H.W. (1988). Ecosystem health: I. measuring ecosystem health. *Environmental Management* **12**: 445-455.

Scheer, M.B. (2009). Nutrient flow in rainfall and throughfall in two stretches in an Atlantic Rain Forest in Southern Brazil (in Portuguese). *Floresta* **39**: 117-130.

Schimel, D., Melillo, J., Tian, H., McGuire, A.D., Kicklighter, D., Kittel, T., Rosenbloom, N., Running, S., Thornton, P., Ojima, D., Parton, W., Kelly, R., Sykes, M., Neilson, R., Rizzo, B. (2000). Contribution of increasing CO_2 and climate to carbon storage by ecosystems in the United States. *Science* **287**: 2004–2006.

Schimel, D.S. (1995). Terrestrial ecosystems and the carbon cycle. *Global Change Biology* **1**:77- 91.

Schlesinger, W.H. (1997). *Biogeochemistry: an analysis of global change*. Academic Press. xiii, 588p.

Scholes, R..J., Kuper, W. and Biggs, R. (2006). Chapter 2: biodiversity. **In**: United Nations Environment Programme (UNEP), Africa Environment Outlook II. Earth print, Stevenage.

Schroeder, P. (1992). Carbon storage potential of short rotation tropical tree plantations. *Forest Ecology and Management* **50**: 31-41.

Schulze, E.D. (2002). The Carbon and Nitrogen Cycle of Forest Ecosystems. **In**: Carbon and nitrogen cycling in European forest ecosystems, pp. 3-13 (E.D. Schulze, ed). Springer, Berlin Heidelberg, New York.

Sebsebe Demissew, Cribb, P. and Rasmussen, F. (2004). *Field Guide to Ethiopian Orchids.* Kew Field guide, Royal Botanic Gardens, Kew.

Sebsebe Demissew, Melaku Wondafrash and Yilma Dellegne (1996). Ethiopa's Natural Base. **In**: *Important Bird Areas of Ethiopia: A First Inventory*, Pp. 36-53. Ethiopia Wildlife and Natural History Society, Addis Ababa, Ethiopia.

Segura, M. and Kanninen, M. (2005). Allometric models for tree volume and total aboveground biomass in a tropical humid forest in Costa Rica. *Biotropica* **37**: 2-8.

Shah, Z., Adams, W.A. and Haven, C.D.V. (1990). Composition and activity of the microbial population in an acidic upland soil and effects of liming. *Soil Biology and Biochemistry* **22**: 257–263.

Shambel Alemu (2011). Woody Species Composition, Diversity and Structural Analysis of Angada Forest in Merti Wereda, Arsi Zone of Oromia Region, Ethiopia, MSC Thesis (Unpublished), Addis Ababa University, Ethiopia.

Shannon, C. E. and Wiener, W. (1949). *The Mathematical Theory of Communication*. University of Illions.

Sheikh, M.A., Kumar, M. and Bussmann, R.W. (2009). Altitudinal variation in soil organic carbon stock in coniferous subtropical and broadleaf temperate forests in Garhwal Himalaya. *Carbon Balance and Management* **4**:6.

Shiels, A.B. (2006). Leaf litter decomposition and substrate chemistry of early successional species on landslides in Puerto Rico. *Biotropica* **38**:348–53.

Shiferaw Belachew (2010). Floristic Composition, Structure and Regeneration Status of Woody Plant Species of Sese Forest, Oromia National Regional State, Southwest Ethiopia. MSc. Thesis (Unpublished), Addis Ababa University, Ethiopia.

Sierra, C.A., del Valle, J.I., Orrego, S.A., Moreno, F.H., Harmon, M.E., Zapata, M., Colorado, G.J., Herrera, M.A., Lara, W., Restrepo, D.E., Berrouet, L.M., Loaiza, L.M. and Benjumea, J.F (2007). Total carbon stocks in a tropical forest landscape of the Porce region, Colombia. *Forest Ecology and Management* **243**:299–309.

Silver, W.L., Lugo, A.E. and Keller, M. (1999). Soil oxygen availability and biogeochemistry along rainfall and topographic gradients in upland wet tropical forest soils. *Biogeochemistry* **44**: 301–328.

Simmons, J.,A., Fernandez, I.J., Briggs, R.D. and Delaney, M.T. (1996). Forest floor carbon pools and fluxes along a regional climate gradient in Maine, USA. *Forest Ecology and Management* **84**:81–95.

Simpson, E.H. (1949). Measurement of Diversity. *Nature* 163: 688-688.

Singh, K.P., Singh, P.K. and Tripathi, S.K. (1999). Litterall, litter decomposition and nutrient release patterns in four native tree species raised on coal mine spoil at Singrauli, India. *Biol. Fertil. Soils* **29**: 371–378.

Singh, L. (1992). Dry matter and nutrient inputs through litterfall in dry tropical forest of India. *Vegetatio* **98**:129-140.

Singh, K.P. (1969). Studies in decomposition of leaf litter of important trees of tropical deciduous forest at Varanasi. *Tropical Ecology* **10**: 292-311.

Slovic, S. (1997). Tree physiology. **In**: *Magnesium deficiency in forest ecosystems*, pp. 101-214 (R.F. Huttl, and W. Schaaf, eds). Kluwer Academic, Dordrecht.

Sohngen, B. and Brown. S. (2008). Extending Timber Rotations: Carbon and Cost Implications. *Climate Policy* **8**: 435-451.

Sokal, R.R. (1974). The species problem Reconsidered. *Systematic Zoology* **22**(4): 360-374.

Songwe, N.C., Okali, D.U.U. and Fasehun, F.E. (1995). Litter Decomposition and Nutrient Release in a Tropical rainforest,Southern Bakundu Forest Reserve, Cameroon. *Journal of Tropical Ecology* **11**: 333-350.

Songwe, N.C., Fasehun, F.E. and Okali, D.U.U. (1988). Litterfall and productivity in a tropical rain forest, Southern Bankundu Forest, Cameroon. *Journal Tropical Ecology* **4**:25-37.

Sousa Neto, E., Carmo, J.B., Keller, M., Martins, S.C., Alves, L.F., Vieira, S.A., Piccolo, M.C., Camargo, P., Couto, H.T.Z., Joly, C.A. and Martinelli L.A. (2011). Soil-atmosphere exchange of nitrous oxide, methane and carbon dioxide in a gradient of elevation in the coastal Brazilian Atlantic forest. *Biogeosciences* **8**: 733–742.

Spies, T.A. (1998). Forest Structure: A Key to the Ecosystem. USDA, Forest Service, Pacific Northwest Research Station, Corvallis, Oregon, USA.

Staelens, J, Nachtergale, L., De Schrijver, A., Vanhellemont, M., Wuyts, K., Verheyen, K. (2011) Spatio-temporal litterfall dynamics in a 60-year-old mixed deciduous forest. *Annals of Forest Science* **68** (1):89-98.

Stephens, B., Gurney, K., Sweeney, C., Peters, W., Bruhwiler, L., Ciais, P., Ramonet, M. and Bousquet, P. (2007). Northern and strong tropical land carbon uptake from vertical profiles of atmospheric CO_2. *Science* **316**: 1732-1735.

Stevenson, F.J. (1986). *Cycles of soil carbon, nitrogen, phosphorus, sulphur, micronutrients*. John

Wiley and Sons, New York, USA.

Struhsaker, T.T. (1997). *Ecology of an African rain forest: logging in Kibale and the conflict between conservation and exploitation*. University Press of Florida, USA, 456p.

SUNARMA (2015). Sustainable Natural Resource Management, Butajira Office, Ethiopia

Sundarapandian, S.M. and Swamy, P.S. (1999) Litter production and leaf-litter decomposition of selected tree species in tropical forests at Kodayar in the Western Ghats, India. *Forest Ecology and Management* **123**: 231-244.

Swai, G., Ndangalasi, H.J., Pantaleo, K.T., Munishi, P.K.T. and Shirima, D.D. (2014). Carbon stocks of Hanang forest, Tanzania: An implication for climate mitigation**.** *Journal of Ecology and the Natural Environment* **6**(3): 90-98.

Swift, M.J. and Anderson, J.M. (1989). Decomposition. **In:** *Ecosystems of the World: Tropical Rain Forest Ecosystems*, pp.547-560 (H. Lieth, and M.J.A. Werger, eds). Amsterdam, Elsevier Science.

Swift, M.J., Heal, O.W. and Anderson, J.M. (1979). *Decomposition in terrestrial ecosystems*. Blackwell Scientific Publications, Oxford.

Tadesse Woldemariam, Borsch, T., Denich, M. and Demel Teketay (2008). Floristic composition and environmental factors characterizing coffee forests in southwest Ethiopia. *Forest Ecology and Management* **255**: 2138–2150.

Tadesse Woldemariam (2003). Vegetation ecology of the Yayu forest in SW Ethiopia: impacts of human use and implications for in-situ conservation of wild *Coffea arabica* L.

populations. Ecology and Development Series No. 10. Cuvillier Verlag, Gottingen.

Takyu, M., Aiba, S. and Kitayama, K. (2003). Changes in biomass, productivity and decomposition along topographical gradients under different geological conditions in tropical lower montane forests on Mount Kinabalu, Borneo. *Oecologia* **134**:397–404.

Tamrat Bekele (1994). Phytosociology and Ecology of Humid Afromontane Forest on the Central plateau of Ethiopia. *Journal of Vegetation Science* **5**:87-98.

Tamrat Bakele (1993). Vegetation and Ecology of Afromontane forests on the central plateau of shewa, Ethiopia, *Acta phytogeorgr. Suec*. 79.

Tan, Z.X., Lal, R., Smeck, N.E. and Calhoun, F.G. (2004). Relationships between surface soil organic carbon pool and site variables. *Geoderma* **21**:185-187.

Tanner, E.V.J., Vitousek, P.M. and Cuevas, E. (1998). Experimental investigation of nutrient limitation of forest growth on wet tropical mountains. *Ecology* **79**: 10–22.

Taylor, B.R., Parkinson, D. and Parsons, W.F.J.(1989). Nitrogen and lignin content as predictors of litter decay rates: a microcosm test. *Ecology* **70**:91–104.

ter Braak, C.J.F.(1994). Cannonical Community Ordination. Part I: Basic Theory and Linear Methods. *Ecoscience* **1**(2): 127-140.

ter Braak, C.J.F.(1987). The analysis of vegetation-Environment Relationships by Canonical correspondence Analysis. *Vegetatio*. **69**: 69-77.

Tesfaye Teklay (2007). Decomposition and nutrient release from pruning residues of two indigenous agroforestry species during the wet and dry seasons. *Nutrient Cycling in Agroecosystems* **77**(2): 115-126.

Teshome Soromessa and Ensermu Kelbessa (2013). Diversity and Endemicity of Chilimo Forest, Central Ethiopia. *Bioscience Discovery* **4**(1):01-04.

Teshome Soromessa, Demel Teketay and Sebsebe Demissew (2004). Ecological study of the vegetation in Gamo Gofa zone, southern Ethiopia. *Journal of Tropical Ecology* **45**: 209-221.

Teshome Soromessa (1997). Ecological study of lowland vegetation of Key-Afer Shal Luqa and Southwest of Lake Chamo. M.Sc. Thesis (Unpublished), Addis Ababa University.

Tewolde Berhan Gebre Egziabher (1988). Vegetation and environment of the mountains of Ethiopia: Implications for utilization and conservation. *Mountain Research Development* **8**: 211-216.

Tewksbury, C.E. and Miegroet, H. (2007). Soil organic carbon along a climatic gradient in a South Appalachian spruce-fir forest. *Canadian Journal of Forestry Research* **37**: 1161-1172.

Theng B.K.G., Tate, K.R. and Sollins, P. (1989). Constituents or organic matter in temperate and tropical soils. **In:** *Dynamics of soil organic matter in tropical ecosystems,* pp. 5-31 (D.C. Coleman, ed). University of Hawaii Press: HI.

Thomas, K., Jijeesh, C.M. and Seethalakshmi, K.K. (2014). Litter production, decomposition and nutrient mineralization dynamics of *Ochlandra setigera*: A rare bamboo species of Nilgiri Biosphere Reserve, India. *Journal of Forestry Research* **25** (3): 579-584.

Thompson, I.D., Mackey, B., McNulty, S. and Mosseler, A., (2009). *Forest resilience, biodiversity, and climate change. A synthesis of the biodiversity/resilience/stability relationship in forest ecosystems*. Technical Series no. 43. Montreal: Secretariat of the Convention on Biological Diversity.

Tian, G., Brussaard, L. and Kang, B.T. (1995). Breakdown of plant residues with contrasting chemical compositions under humid tropical conditions: Effects of earthworms and millipedes. *Soil Biology and Biochemistry* **27**(3): 277–280.

Tian G, Kang, B.T. and Brussaard, L. (1992). Effects of chemical composition on N, Ca, and Mg release during incubation of leaves from selected agroforestry and fallow species. *Biogeochemistry* **15**: 1–17.

Tibebu Yelemfrhat, Teshome Soromessa and Eyale Bayable (2014). Forest Carbon Stocks in Lowland Area of Simien Mountains National Park: Implication for Climate Change Mitigation. *STAR* **3**(3): 29-36.

Tibebu Yelemfrhat and Teshome Soromessa (2015). Carbon stock variations along altitudinal and slope gradient in the Forest Belt of Simen Mountains National Park, Ethiopia. American Journal of Environmental Protection **4**(4): 199-201.

Tisdale, S.L. Nelson W.L., Beaton, J.D. and Havlin J.L. (1993). *Soil Fertility and Fertilizers*. 5th ed., Prentice Hall, New Delhi, India.

Tolsma, D.J., Ernest, W.H.O., Verweij, R.A. and Vooijs, R. (1987). Seasonal Variation of Nutrient concentration in a semi arid savana ecosystem in Botswana. *Journal of ecology* **75**: 755-770.

Torreta, N.K. and Takeda, H. (1999). Carbon and Nitrogen dynamics of decomposing leaf litter in a tropical hill evergreen forest. *European Journal of Soil Biology* **35**: 57-63.

Triadiati, T., Tjitrosemito, S., Guhardja, E., Sudarsono, I., Qayim I., and Leuschner, C. (2011). Litterfall Production and Leaf-Litter Decomposition at Natural Forest and Cacao Agroforestry in Sulawesi, Indonesia. *Asian Journal of Biological Sciences* **4** (3): 221–234.

Tripathi, S.K. and Singh, K.P. (1992). Abiotic and litter quality control during decomposition of different plant parts in a dry tropical bamboo savanna in India. *Pedobiologia* **36**: 241–256.

Tripathi, S.K., Sumida, A., Shibata, H., Ono, K., Uemura, S., Kodama, Y. and Hara, T. (2006). Leaf litterfall and decomposition of different above and belowground parts of birch (*Betula ermanii*) trees and dwarf bamboo (*Sasa kurilensis*) shrubs in a young secondary forest in Northern Japan. *Biol. Fertil. Soil* **43**: 237–246.

Tsegaye Tadesse (2010). Bale Eco-region Sustainable Management Programme. http://www.pfmpfarmsos.org/Publication.html.

Tulu Tolla (2011). Estimation of Carbon Stock in Church Forests: Implications for Managing Church Forest for Carbon Emission Reduction. MSc Thesis (Unpublished), Addis Ababa University, Ethiopia.

UNFCCC (2011). Report of the Global Environment Facility to the Seventeenth Session Of the Conference of the Parties to the United Nations Framework Convention on Climate Change.113p.

UNFCCC (2007). Report of the conference of parties on its thirteenth session, Bali, Indonesia. United Nations framework convention on climate change, Geneva**.**

UN-REDD (2011). UN-REDD Programme Strategy 2011-2015. 28p.

Vagen, T.G., Lal, R. and Singh, B.R. (2004). Soil carbon sequestration in sub-saharan Africa: a review. *Land Degraded Develop* **16**: 54–71.

Van der Maarel, E. (1979). Transformation of cover abundance values in phytogeography and its effects on community similarity. *Vegetatio* **39**:97-114.

van der Werf, G. R., Randerson, J. T., Giglio, L., Collatz, G. J., Kasibhatla, P. S., and Arellano Jr., A. F. (2006). Interannual variability in global biomass burning emissions from 1997 to 2004. *Atmos. Chem. Phys*. **6**: 3423–3441.

van Voris, P., O'Neill, R. V., Emanuel, W. R., and Shugart, H.H. (1980). Functional complexity and ecosystem stability. *Ecology* **61**:1352-1360.

Vargas, E., Allen, M.F. and Allen, E.B. (2008). Biomass and carbon accumulation in a fire chrono-sequence of a seasonally dry tropical forest. *Global Change Biology* **14**: 109–124.

Verhoef, H.A. and Guandi, B. (2001). Decomposition Dynamics and Nutrient flow in Pine Forest Plantation in Central Java. **In**: *Management of Tropical Plantation-forests and their Soil litter System*, pp. 173-211 (Reddy, M.V. ed.). Science Publications inc., USA.

Vesterdal, L., Schmidt, I. and Callesen, I. (2008). Carbon and nitrogen in forest floor and mineral soil under six common European tree species. *Forest Ecology and Management* **255**:35–48.

Vieilledent, G., Vaudry, R., Andriamanohisoa, S.D., Rakotonarivo, O.S., Randrianasolo, H.Z., et al. (2012) A universal approach to estimate biomass and carbon stock in tropical forests using generic allometric models. *Ecological Applications* **22**: 572-583.

Vitousek, P. and Sanford, R.L.J. (1986). Nutrient cycling in moist tropical forest. *Annu Rev Ecol Syst* **17**:137–167.

Vitousek, P.M. and Reiners, W.A. (1975). Ecosystem succession and nutrient retention: a hypothesis. *Bioscience* **25**:376-381.

Vitousek, P.M., Turner, D.R., Parton, W.J. and Sanford, R.L. (1994). Litter decomposition on the Mauna Loa environmental matrix, Hawaii: Patterns, mechanisms and Models. *Ecology*, **75**: 418 429.

Vitousek, P. (1984). Litterfall, nutrient cycling and nutrient limitation in tropical forests. *Ecology* **65**:285–298.

Vitousek, P. and Sanford, R.L.J. (1986). Nutrient cycling in moist tropical forest. *Annu Rev Ecol Syst* **17**:137–167.

Vivero, J.L., Ensermu Kelbessa and Sebsebe Demissew (2005). *The Red List of Endemic Trees and Shrubs of Ethiopia and Eritrea*. Fauna and Flora International, Cambridge.

Vogt, K.A., Grier, C.C. and Vogt, D.J. (1986). Production, turnover, and nutrient dynamics of above and below-ground detritus of world forests. *Advances in Ecological Research* **15**: 303-377.

von Breitenbach, F. (1962). National Forestry Development Planning: A Feasibility and Priority Study on the examples of Ethiopia. *Ethiop. For. Rev.,* (**3**/**4**): 41-68.

von Breitenbach, F. (1961). Forest and woodland of Ethiopia: A Geo-Botanical Contribution to the knowledge of Principal Plant Communities of Ethiopia, With special Regard to Forestry. *Ethiopian Forestry Review* **1**: 5-16.

Walkley, A. and Black I.A. (1934). An examination of the Degtjareff method for determining soil organic matter and a proposed modification of the chromic acid titration method. *Soil Science* **37**:29–38.

Wang, H.Q., Hall, C.A.S., Scatena, F.N., Fetcher, N. and Wu, W. (2003). Modelling the spatial and temporal variability in climate and primary productivity across the Luquillo Mountains, Puerto Rico. *Forest Ecology and Management* **179**:69–94.

Wang, H.Q., Wang, S. and Huang, Y. (2008). Comparisons of litterfall, litter decomposition and nutrient return in a monoculture *Cunninghamia lanceolata* and a mixed stand in southern China. *Forest Ecology and Management* **255**: 1210–1218.

Waring, R.H. and Schlesinger, W.H. (1985). Forest ecosystems: concepts and management. Academic Press Harcourt Brace Jovanovich, Orlando, FL.

WBISPP (2005). A National Strategy plan for the biomass sector. Addis Ababa, Ethiopia.55p. .

Werneck, M.S., Pedralli, G. and Gieseke, L.F. (2001). Produção de serrapilheira em três trechos de uma floresta semidecídua com diferentes graus de perturbação na Estação Ecológica do Tripuí, Ouro Preto, MG. *Rev. Bras. Bot.* **24**:195-198.

White, F. (1983).*The Vegetation of Africa. A descriptive memoir to accompany the UNESCO/AETFAT/UNSO*. UNESCO, Paris. 356p.

White, F. (1981). The History of Afromontane archepelago and the scientific need for its conservation. *African Journal of Ecology* **19**: 33-54.

Whittaker, R.H. (1975). *Communities and Ecosystems*. 2nd ed. Machmillan, London.

Whittaker, R.H. (1972). Evolution and measurement of species diversity. *Taxon* **21**: 213–251.

Wilcke W, Oelmann Y, Schmitt A, Valarezo C, Zech W, Homeier J. (2008). Soil properties and tree growth along an altitudinal transect in Ecuadorian tropical montane forest. *Journal of Plant Nutrition and Soil Science* **171**:220–30.

Williams, S.T. and Gray, T.R.G. (1974). Decomposition of litter on the soil surface.In: *Biology of plant litter decomposition.* Volume 2*,* Pp.611-632 (Dickinson, C.H. and Pugh G.J.F., eds.). Academic Press, New York, USA.

Williams, M., Ryan, C.M., Rees, R.M., Sarnbane, E., Femando, J. and Grace, J. (2008). Carbon sequestration and biodiversity of re-growing miombo woodlands in Mozambique. *Forest*

Ecology and Management **254**: 145-155.

Williams, C.H., Hanan,N.P., Neff, J.C., Scholes, R.J., Berry, J.A., Denning, A.S. and Baker, D.F. (2007). Africa and the global carbon cycle. *Carbon Balance and Management* **2**:3.

Wood, T.E., Lawrence, D. and Clark, D.A. (2006). Determinants of Leaf Litter Nutrient Cycling in a Tropical Rain Forest: Soil Fertility Versus Topography. *Ecosystems* **9**: 700–710.

Wood, T.E., Lawrence, D, and Clark, D.A. (2005). Variation in leaf litter nutrients of a Costa Rican rain forest is related to precipitation. *Biogeochemistry* **73**:417–37.

Wood, T.G. (1974). Field Investigations on the decomposition of leaves of Eucalyptus delegatensis in relation to Environmental factors. *Pedoiology* **14**: 343-371.

Wright, S. J. (2005) Tropical forests in a changing environment. *Trends in Ecology and Evolution* **20**: 553–560.

Wu, J., Liu, Z. and Wang, X. (2011). Effects of understory removal and tree girdling on soil microbial community composition and litter decomposition in two Eucalyptus plantations in South China. *Functional Ecology* **25**:921–31.

WWF and IUCN (1994). Centers of Plant Diversity: A Guide and Strategy for Their Conservation, 3 volumes. IUCN Publications Unit, Cambridge

www.TerrAfrica.org.

Xu, X. and Hirata, E. (2005). Decomposition patterns of leaf litter of seven common canopy species in a subtropical forest: N and P dynamics. *Plant and Soil* **273** (1-2): 279–289.

Xuluc-Tolosa, F. J., Vester, H. F. M., Ramirez-Marcial, N., Castellanos-Albores, J. and Lawrence, D. (2003). Leaf litter decomposition of tree species in three successional phases of tropical dry secondary forest in Campeche, Mexico. *Forest Ecology and Management* **174**: 401–412.

Yalden, D.W. (1983). The extent of high-ground in Ethiopia compared to the rest of Africa. *SINET*: *Ethiopian Journal of Science* **6**: 35–39.

Yang, Y.S., Guo, J. F., Chen, G.S., Xie, J.S, Gao, R., Li, Z. and Jin, Z. (2005). Litter production, seasonal pattern and nutrient return in seven natural forests compared with a plantation in southern China. *Forestry* **78** (4): 403-415.

Yong, Y., Tai, S. and Bao, X. (2011). Forest soil organic carbon density and its distribution characteristics along an altitudinal gradient in Lushan Mountains of China. **22**(7):1675-81.

Yong, Y., Tai, S. and Bao, X. (2007). Forest litter decomposition and its responses to global climate change. *Dec* **18**(12):2889-2895.

Zanne, A.E., Lopez-Gonzalez, G., Coomes, D.A., Ilic, J., Jansen, S., Lewis, S.L., Miller, R.B., Swenson, N.G., Wiemann, M.C. and Chave, J. (2009). Global wood density database. Dryad [February, 2016]. http://hdl.handle.net/10255/dryad.235.

Zerihun Woldu (2016). *Comprehensive and Ecological Data Analysis*; Basics, Concepts and Methods. Lambert Academic Publishing, Germany, 409p.

Zerihun Woldu (1999). Forests in the vegetation types of Ethiopia and their status in the geographical context. **In**: *Forest Genetic Resources Conservation: Principles, Strategies and Actions*, pp. 1-38 (Edwards, S., Abebe Demmissie, Taye Bekele and Haase, G., eds.). Workshop Proceedings. Institute of Biodiversity Conservation and Research, and GTZ, Addis Ababa.

Zerihun Woldu, Feoli, E. and Lisanework Nigatu (1989). Partitioning an elevation gradient of vegetation from southeastern Ethiopia by probabilistic methods. *Vegetatio* **81**: 189–198.

Zhang, H., Yuan, W., Dong, W. and Liu, S. (2014). Seasonal patterns of litterfall in forest ecosystem worldwide. *Ecological Complexity* **20**: 240-247.

Zhang, D., Hui, D., Luo, Y. and Zhou G. (2008). Rates of litter decomposition in terrestrial ecosystems: global patterns and controlling factors. *Journal of Plant Ecology* **1**(2): 1–9.

Zhou, G., Guan, L., Wei, X., Zhang, D., Zhang, Q., Yan, J., Wen, D., Liu, J., Liu, S., Huang, Z., Kong, G., Mo, J. and Yu, Q. (2007). Litterfall production along successional and altitudinal gradients of subtropical monsoon evergreen broadleaved forests in Guangdong, China. *Plant Ecology* **188**: 77–89.

Zhu, B., Wang, X., Fang J., Piao, S., Shen, H., Zhao, S. and Peng, C. (2010). Altitudinal changes in carbon storage of temperate forests on Mt Changbai, Northeast China**.** *Journal of Plant Research* **123** (4):439-52.

Zianis, D., Muukkonen, P., Mäkipää, R. and Mencuccini, M. (2005). Biomass and stem volume equations for tree species in Europe. *Silva Fennica* **4**: 1-63.

Zimmermann M, Meir, P., Silman, M.R., Fedders, A., Gibbon, A., Malhi, Y., Urrego, D.H., Bush, M.B., Feeley, K.J., Garcia, K.C., Dargie, G.C., Farfan, W.R., Zamora, F. (2010). No differences in soil carbon stocks across the tree line in the Peruvian Andes. *Ecosystems* **13**: 62–74.

9. Appendices

Appendix I. List of Plant Species Collected from Bitey Forest

No.	Local name	Scientific Name	Family Name	Habit
1		*Acalypha psilostachya* Hochst.	Euphorbiaceae	Shrub
2		*Achyranthes aspera* L.	Amaranthaceae	Herb
3	Guticha	*Acmella caulirhiza* Del.	Asteraceae	Herb
4		*Adiantum thalictroides* Schtdl.	Adianthaceae	Fern
5		*Agrocharis melanantha* Hochst.	Apiaceae	Herb
6	Ematelit	*Ajuga integrifolia* Buch.-Ham.ex D. Don.	Lamiaceae	Herb
7	Yefur enzin	*Alchemilla abyssinica* Fresen.	Rosaceae	Herb
8		*Anagallis arvensis* L.	Primulaceae	Herb
9		*Anthospermum herbaceum* L. f.	Rubiaceae	Herb
10	Wondimu	*Apodytes dimidiata* E. Mey. ex Arn.	Icacinaceae	Tree
11		*Asparagusafricanus* Lam.	Asparagaceae	Herb
12		*Asplenium abyssinicum* Fee	Aspleniaceae	Fern
13		*Asplenium aethiopicum* (Burm. f.) Bech.	Aspleniaceae	Fern
14		*Asplenium monanthes* L.	Aspleniaceae	Fern
15		*Asplenium protensum* Schard.	Aspleniaceae	Fern
16	Kureta	*Bersama abyssinica* Fresen.	Melianthaceae	Shrub
17	Adey	*Bidens prestinaria* (Sch. Bip.) Cufod.	Asteraceae	Herb
18	Abeyi	*Brucea antidysenterica* J F. Mill.	Simaroubaceae	Shrub
19	Abre	*Buddleja polystachya* Fresen.	Loganiaceae	Shrub
20		*Calpurnea aurea* (Ait) Bent.	Fabaceae	Shrub
21		*Campanula edulis* Forssk.	Campanulaceae	Herb
22		*Canthium oligocarpum* Hier	Rubiaceae	Shrub
23		*Carania eminii* Schweinf	Campanulaceae	Epiphyte
24		*Carissa spinarium* L	Apocynaceae	Shrub
25		*Cerastium octandrum* A. Rich.	Caryophyllaceae	Herb
26		*Chiliocephalem tegetum* Mesfin	Asteraceae	Herb
27	Dendere	*Cirsium dender* Friis	Asteraceae	Herb
28		*Clausena anisata* (Willd.) Benth.	Rutaceae	Shrub
29		*Clematis hirsuta* Perr. & Guill.	Ranunculaceae	Climber
30	Yedibir hareg	*Clematis simensis* Fresen.	Ranunculaceae	Climber
31		*Clerodendrum myricoides* (Hochst.) Vatke	Lamiaceae	Herb
32		*Clutia abyssinica* Jaub. & Spach.	Euphorbiaceae	Shrub
33	Yefiyel kolo	*Colutea abyssinica* Kunth. & Bouche	Fabaceae	Shrub
34		*Commelina africana* L.	Commelinaceae	Herb
35		*Commelina benghalensis* L.	Commelinaceae	Herb
36		*Conyza pyrrhopappa* Sch. Bip. ex A. Rich.	Asteraceae	Herb

37	Nechach	*Conyza hypoleuca* A. Rich.	Asteraceae	Shrub
38		*Conyza steudelii* Sch. Bip. ex A. Rich.	Asteraceae	Herb
39		*Conyza tigrensis* Oliv. & Hiern	Asteraceae	Herb
40		*Conyza vernonioides* (Sch. Bip. ex A. Rich.) Wild	Asteraceae	Shrub
41		*Crassocephalum macropappum*(Sch. Bip. ex A.Rich.) S. Moore	Asteraceae	Herb
42		*Crassula alsinoides* (Hook. f.) Engl.	Crassulaceae	Herb
43		*Crotolaria quartiniana* A.Rich.	Fabaceae	Herb
44	Bisana	*Croton macrostachyus* Del.	Euphorbiaceae	Tree
45		*Cynoglossum amplifolium* Hochst ex A. DC.in DC.	Boraginaceae	Herb
46	Tibikit	*Cynoglossum coeruleum* Hochst ex A. DC.in DC.	Boraginaceae	Herb
47		*Diaphananthe schimperiana* (A. Rich.) Summerh.	Orchidaceae	Epiphyte
48		*Dichrocephala chrysanthemifolia* (Bl.) DC.	Asteraceae	Herb
49		*Dichrocephala integrifolia* (L. f.) Kuntze	Asteraceae	Herb
50		*Discopodium penninervium* Hochst.	Solanaceae	Shrub
51		*Dodonea angustifolia* L.f.	Sapindaceae	Tree
52		*Dombeya torrida* (J. F.Gmel) P. Bamps	Stereculiaceae	Tree
53		*Dovyalis abyssinica* (A. Rich.) Warb.	Flacourtiaceae	Shrub
54		*Drynaria volkensii* Hieron.	Polypodiaceae	Fern
55		*Dryopteris inaequalis* (Schltdl.) Kuntze	Dryopteridaceae	Fern
56		*Dracaena afromontana*Mildbr.	Dracaenaceae	Tree
57	Abetta	*Dregea schimperi* (Decne.) Bullock	Asclepiadaceae	woody climber
58	Yemar Dendere	*Echinops macrochaetus* Fresen.	Asteraceae	Herb
59	Wulel	*Ekebergia capensis* Sparrm.	Meliaceae	Tree
60	Enkoko	*Embelia schimperi* Vatke	Myrsinaceae	Climber
61		*Englerina woodfordioides* (Schweinf.) M. Gilbert	Loranthaceae	Parasite
62	Gedra	*Erica arborea* L.	Ericaceae	Shrub
63		*Erythrina brucei* Schweinf.	Fabaceae	Tree
64	Kelekel	*Euphorbia obovalifolia* A. Rich.	Euphorbiaceae	Tree
65		*Galinsoga quadriradiata* Ruiz & Pavon	Asteraceae	Herb
66	Anzolila	*Galiniera saxifraga* (Hochst.) Bridson	Rubiaceae	Tree
67		*Galium thunbergianum* Eckl. & Zeyh.	Rubiaceae	Herb
68	Ashehet	*Galium simense* Fresen.	Rubiaceae	Herb

69		*Geranium arabicum* Forssk.	Geraniaceae	Herb
70	Awura	*Gnidia lamprantha* Gilg	Thymelaeaceae	Shrub
71		*Habenaria petitiana* (A. Rich.) Th.Dur. & Schinz	Orchidaceae	Herb
72	Kebse	*Hagenia abyssinica* (Brace) JF. Gmel.	Rosaceae	Tree
73	Erorate Busel	*Halleria lucida* L.	Scrophulariaceae	Shrub
74		*Haplocarpha schimperi* (Sch. Bip.) Beauv.	Asteraceae	Herb
75		*Helichrypsum foetidum* (L.) Moench.	Asteraceae	Herb
76		*Helichrypsum odoratissimum* auct., non (L.) Sweet	Asteraceae	Shrub
77		*Heracleum abyssinicum* (Boiss.) Norman	Apiaceae	Shrub
78		*Hypericum peplidifolium* A. Rich.	Hypericaceae	Herb
79		*Hypericum quartinianum* A. Rich.	Hypericaceae	Shrub
80	Abeje	*Hypericum revolutum* Vahl	Hypericaceae	Shrub
81		*Hypoestes triflora* (Forssk.) Roem. & Schult.	Acanthaceae	Herb
82	Kechech	*Ilex mitis* (L.) Radlk.	Aquifoliaceae	Tree
83	Enshoshila	*Impatiens tinctoria* A. Rich.	Balsaminaceae	Herb
84		*Inula confertiflora* A. Rich.	Asteraceae	Shrub
85		*Jasminum abyssinicum* Hochst. ex DC.	Oleaceae	Climber
86	Mixo	*Jasminum stans* Pax.	Oleaceae	Shrub
87	Tede	*Juniperus procera* Hochst. ex Endl.	Cuppressaceae	Tree
88		*Kalanchoe densiflora* Rolfe	Crassulaceae	Herb
89	Endahula	*Kalanchoe petitiana* A. Rich.	Crassulaceae	Herb
90	Ambare	*Kniphofia foliosa* Hochst.	Asphodelaceae	Herb
91		*Lactuca inermis* Forssk.	Asteraceae	Herb
92	Yoyo bosha	*Laggera crispata* (Vahl) Hepper & Wood	Asteraceae	Herb
93	Yerte bosha	*Laggera tomentosa* (Sch. Bip. ex A. Rich.) Oliv. & Hiern	Asteraceae	Herb
94	Kesay	*Lippia adoensis* Hochst. ex Walp.	Verbenaceae	Shrub
95		*Lithospermum afromontanum* Weim.	Boraginaceae	Herb
96	Yegech Betir	*Lobelia giberroa* Hemsl.	Campanulaceae	Tree
97		*Lobelia holstii* Engl.	Campanulaceae	Herb
98		*Lotus corniculatus* L.	Fabaceae	Herb
99	Kelebuye	*Maesa lanceolata* Forssk.	Myrsinaceae	Tree
100		*Maytenus arbutifolia* (A. Rich.) Wilczek	Celastraceae	Shrub
101		*Maytenus senegalensis* (Lam.) Exell	Celastraceae	Tree
102	Shashamagna	*Maytenus addat* (Loes.) Sebsebe	Celastraceae	Tree

103		*Mikaniopsis clematoides* (Sch. Bip. ex A. Rich.) Milne-Redh.	Asteraceae	Climber
104	Deremo	*Mimulopsis solmsii* Schweinf.	Acanthaceae	Herb
105		*Monopsis stellarioides* (Presl.) Urb.	Lobeliaceae	Herb
106	Telota	*Myrica salicifolia* A. Rich.	Myricaceae	Tree
107	Kechemo	*Myrsine africana L.*	Myrsinaceae	Shrub
108	Gomira	*Myrsine melanophloeos* (L.) R. Br.	Myrsinaceae	Tree
109		*Nepeta azurea* R. Br.ex Benth.	Lamiaceae	Herb
110		*Nuxia congesta* R.Br. ex Fresen.	Loganiaceae	Tree
111		*Oldenlandia monanthos* (A. Rich.) Hiern	Rubiaceae	Herb
112	Bunna	*Olea europaea* L. subsp. *cuspidata* (Wall.ex G.Don.) Cif.	Oleaceae	Tree
113		*Olinia rochetiana* A. Juss.	Oliniaceae	Tree
114		*Ocimum lamiifolium* Hochst. ex Benth	Lamiaceae	Shrub
115	Karro	*Osyris quadripartita* Decne.	Santalaceae	Shrub
116		*Oxalis corniculata* L.	Oxalidaceae	Herb
117		*Pappea capensis* Eckl. & Zeyh.	Sapindaceae	Tree
118		*Pavetta abyssinica* Fresen.	Rubiaceae	Shrub
119	Musaber	*Pentas schimperiana* (A. Rich.) Vatke	Rubiaceae	Shrub
120		*Peperomia abyssinica* Miq.	Piperaceae	Parasite
121		*Peperomia fernandopoiana* C. DC.	Piperaceae	Herb
122		*Peucedanum mattirolii* Chiov.	Apiaceae	Herb
123		*Periploca linearifolia* Quart.-Dill. & A. Rich.	Asclepiadaceae	Climber
124		*Peucedanum petitianum*A. Rich.	Apiaceae	Herb
125		*Phagnalon abyssinicum* Sch. Bip. ex A. Rich.	Asteraceae	Herb
126	Endod	*Phytolacca dodecandra* L 'Herit.	Phytolaccaceae	Climber
127		*Pilea rivularis* Wedd.	Urticaceae	Herb
128		*Pimpinella oreophila* Hook. f.	Apiaceae	Herb
129	Umbalbula	*Pittosporum abyssinicum* Del.	Pittosporaceae	Tree
130		*Plantago palmata* Hook.f.	Plantaginaceae	Herb
131	Yeroratebusil	*Plectranthus assurgens* (Baker.) J.K. Morton	Lamiaceae	Herb
132		*Pluchea dioscoridis* (L.) DC.	Asteraceae	Herb
133	Zigba	*Podocarpus falcatus* (Thunb.) R.B.ex.Mirb.	Podocarpaceae	Tree
134		*Premna schimperi* Engl.	Lamiaceae	Shrub
135		*Primula verticillata* Forssk	Primulaceae	Herb
136	Gerbe	*Prunus africana* (Hook. f) Kalkm.	Rosaceae	Tree
137		*Pteridium aquilinum* (L.) Kuhn	Sinopteridaceae	Fern
138		*Pteris cretica* L.	Sinopteridaceae	Fern
139		*Pteris dentata* Forssk.	Sinopteridaceae	Fern

140	Fanfa	*Pycnostachys abyssinica* Fresen.	Lamiaceae	Herb
141		*Ranunculus multifidus* Forssk.	Ranunculaceae	Herb
142	Geshe	*Rhamnus prinoides* L' Herit	Rhamnaceae	Shrub
143	Jejee	*Rhamnus staddo* A. Rich.	Rhamnaceae	Shrub
144	Taala	*Rhus glutinosa* A.Rich.	Anacardiaceae	Shrub
145		*Rhus retinorrhoea* Oliv.	Anacardiaceae	Shrub
146	Taala	*Rhus ruspolii* Engl.	Anacardiaceae	Shrub
147	Engoche	*Rosa abyssinica* Lindley	Rosaceae	Shrub
148	Enjere	*Rubus steudneri* Schweinf.	Rosaceae	Shrub
149	Tumiye	*Rumex nepalensis* Spreng.	Polygonaceae	Herb
150	Habeshawa	*Rumex nervosus* Vahl	Polygonaceae	Shrub
151	Hamam	*Salvia nilotica* Jacq.	Lamiaceae	Herb
152		*Sanicula elata* Buch.-Ham. ex D. Don	Apiaceae	Herb
153	Yedibir tosign	*Satureja biflora* (Ham. ex Don) Briq.	Lamiaceae	Shrub
154	Bie'er	*Satureja punctata* (Benth.) Briq.	Lamiaceae	Herb
155	Bie'er	*Satureja paradoxa* (Vatke) Engl.ex Seybold	Lamiaceae	Herb
156		*Satureja simensis* (Benth.) Briq.	Lamiaceae	Herb
157		*Scabiosa columbaria* L.	Dipsacaceae	Herb
158	Getem	*Schefflera abyssinica* (Hochst. ex A. Rich.) Harms	Araliaceae	Tree
159	Angibo	*Schefflera volkensii* (Engl.) Harms	Araliaceae	Tree
160	Amotta	*Scolopia theifolia* Gilg	Flacourtiaceae	Tree
161		*Sebaea brachyphylla* Griseb.	Gentianaceae	Herb
162		*Senecio ragazzii* Chiov.	Asteraceae	Herb
163	Wogirtu	*Silene macrosolen* A. Rich.	Caryophyllaceae	Herb
164	Abita	*Smilax aspera* L.	Smilacaceae	Shrub
165		*Solanecio mannii* (Hook. f.) C. Jeffrey	Asteraceae	Shrub
166	Afechoma	*Solanum benderianum* Schimper ex Dammer	Solanaceae	Shrub
167	Kishishe	*Solanum indicum* L.	Solanaceae	Shrub
168	Awitiye	*Solanum nigrum* L.	Solanaceae	Shrub
169	Yedibir Awitiye	*Solanum vilosum* Mill.	Solanaceae	Herb
170		*Sonchus bipontini* Asch.	Asteraceae	Herb
171		*Sparmannia ricinocarpa* (Eckl. & Zehy.) O. Ktze	Tiliaceae	Shrub
172		*Stachys aculeolata* Hook. f.	Lamiaceae	Herb
173		*Stephania abyssinica* (Dillon & A. Rich.) Walp.	Menispermaceae	Climber
174		*Swertia abyssinica* Hochst.	Gentianaceae	Herb
175		*Swertia kilimandscharica* Engl.	Gentianaceae	Herb
176		*Syzygium guineense* subsp. *afromontanum* F. White	Myrtaceae	Tree

177		*Tagetes minuta* L.	Asteraceae	Herb
178		*Thalictrum rynchocarpum* Dill. & A. Rich.	Ranunculaceae	Herb
179		*Tacazzea conferta* N.E. Br.	Asclepiadaceae	Climber
180		*Tapinanthus heteromorphus* (A. Rich.) Danser	Loranthaceae	Parasite
181	Ararat	*Thunbergia alata* Boj. ex Sims	Acanthaceae	Climber
182		*Trifolium burchellianum* Ser.	Fabaceae	Herb
183	Tosign	*Thymus schimperi* Ronniger	Lamiaceae	Herb
184	Zewtir	*Urera hypselodendron* (A. Rich.) Wedd.	Urticaceae	Climber
185		*Verbascum sinaiticum* Benth.	Scrophulariaceae	Herb
186	Dingrita	*Vernonia rueppellii* Sch. Bip. ex Walp.	Asteraceae	Tree
187		*Veronica abyssinica* Fresen.	Scrophulariaceae	Herb
188		*Vigna oblongifolia* A. Rich.	Fabaceae	Climber
189		*Viola abyssinica* Oliv.	Violaceae	Herb
190		*Zehneria scabra* (Linn. f.) Sond.	Curcurbitaceae	Climber

Appendix II. Plant families with the respective number of species

No.	Family	No. species	Proportion
1	Acanthaceae	3	1.58
2	Adianthaceae	1	0.53
3	Amaranthaceae	1	0.53
4	Anacardiaceae	3	1.58
5	**Apiaceae**	**6**	3.16
6	Apocynaceae	1	0.53
7	Aquifoliaceae	1	0.53
8	Araliaceae	2	1.05
9	Asclepiadaceae	3	1.58
10	Asparagaceae	1	0.53
11	Asphodelaceae	1	0.53
12	Aspleniaceae	4	2.11
13	Asteraceae	29	15.26
14	Balsaminaceae	1	0.53
15	Boraginaceae	3	1.58
16	Campanulaceae	4	2.11
17	Caryophyllaceae	2	1.05
18	Celastraceae	3	1.58
19	Commelinaceae	2	1.05
20	Crassulaceae	3	1.58

21	Cuppressaceae	1	0.53
22	*Curcurbitaceae*	1	0.53
23	Dipsacaceae	1	0.53
24	Dracaenaceae	1	0.53
25	Dryopteridaceae	1	0.53
26	Ericaceae	1	0.53
27	Euphorbiaceae	4	2.11
28	Fabaceae	**7**	3.68
29	Flacourtiaceae	2	1.05
30	Gentianaceae	3	1.58
31	Geraniaceae	1	0.53
32	Hypericaceae	3	1.58
33	Icacinaceae	1	0.53
34	Lamiaceae	14	7.37
35	Lobeliaceae	1	0.53
36	Loganiaceae	2	1.05
37	Loranthaceae	2	1.05
38	Meliaceae	1	0.53
39	Melianthaceae	1	0.53
40	Menispermaceae	1	0.53
41	Myricaceae	1	0.53
42	Myrsinaceae	4	2.11
43	Myrtaceae	1	0.53
44	Oleaceae	3	1.58
45	Oliniaceae	1	0.53
46	Orchidaceae	2	1.05
47	Oxalidaceae	1	0.53
48	Phytolaccaceae	1	0.53
49	Piperaceae	2	1.05
50	Pittosporaceae	1	0.53
51	Plantaginaceae	1	0.53
52	Podocarpaceae	1	0.53
53	Polygonaceae	2	1.05
54	Polypodiaceae	1	0.53
55	Primulaceae	2	1.05
56	Ranunculaceae	4	2.11
57	Rhamnaceae	2	1.05
58	**Rosaceae**	**5**	2.63
59	Rubiaceae	**8**	4.21

60	Rutaceae	1	0.53
61	Santalaceae	1	0.53
62	Sapindaceae	2	1.05
63	Scrophulariaceae	3	1.58
64	Simaroubaceae	1	0.53
65	Sinopteridaceae	3	1.58
66	Smilacaceae	1	0.53
67	**Solanaceae**	**5**	2.63
68	Stereculiaceae	1	0.53
69	Thymelaeaceae	1	0.53
70	Tiliaceae	1	0.53
71	Urticaceae	2	1.05
72	Verbenaceae	1	0.53
73	Violaceae	1	0.53
	Total	190	100

Appendix III. Endemic taxa, IUCN red list category and its distribution in the Flora Area

No.	Endemic species	Family	Habit	IUCN Red list category	Distribution
1	*Cerastium octandrum*	Caryophyllaceae	Herb		
2	*Chiliocephalem tegetum*	Asteraceae	Herb		Afroalpine meadows with *Lobelia rynchopetalum*, outcrops of moss-covered rocks in *Erica-Hypericum* bushland; moist *Schefflera-Hagenia forest*; 2400-3600m. GD, GJ, SD, BA not known elsewhere.
3	*Cirsium dender*	Asteraceae	Herb		Glades in degraded *Juniperus-Podocarpus* forest and *Hagenia-Schefflera* forest, *Erica-Hagenia* scrub, *Arundinaria* bamboo thicket, often on steep slopes; 2200-2750 m. SU, IL, KF, GG not known elsewhere
4	*Clutia abyssinica*	Euphorbiaceae	Shrub	VU	Evergreen bushland or margins of *Juniperus* and *Podocarpus* forest, mostly in disturbed sites; 1450-2950m.TU,GD,WU,GJ,SU,AR, KF,SD,BA, HA.
5	*Crassocephalum macropappum*	Asteraceae	Herb		Common in moist places along margins of evergreen Forest, dry evergreen woodland and bushland, wasteland, and along road sides; 1600-3270m. GO, GJ, WU, SU, WG, IL, KF, GG, SD, BA, HA; not known elsewhere.
6	*Cynoglossum coeruleum*	Boraginaceae	Herb		Afroalpine grassland, rocky areas, field margins and fallow fields, openjuniperforest, woodland; 2700-3400m. GD GJ SU BA; not known elsewhere.
7	*Inula confertiflora*	Asteraceae	Shrub	NT	Margins of and clearings in *Juniperus-Podocarpus* forest. *Erica arborea* scrub,along stream banks in *Eucalyptus* plantations, montane grassland on slopes with scattered *Juniperus*, *Hagenia*, *Hypericum*, etc: 2500-3730 m. WU, SU, AR, BA, HA:not known elsewhere.
8	*Kalanchoe petitiana*	Crassulaceae	Succulent herb	LC	Forest margins and open evergreen bushland, often in disturbed areas; 2000-3000 m, EW,GO, WU, SU, AR, BA, HA
9	*Laggera tomentosa*	Asteraceae	Herb		Dry hill and mountain slopes; 2345-2950 m. TU, GO, GJ, WU, SU, HA; not known elsewhere.
10	*Lippia adoensis*	Verbenaceae	shrub	LC	
11	*Lotus corniculatus*	Fabaceae	Herb		Marshy places in grassland; 1900-

					2400 m, EW TU, HA Ethiopian plants are probably best referred and the description only applies to the Ethiopian material.
12	*Maytenus addat*	Celasteraceae	Tree	NT	It ranges throughout the highlands at altitudes of 2200-3000 m. Distributed in SU, AR, SD, GG floristic regions
13	*Mikaniopsis clematoides*	Asteraceae	Climber	LC	Margins of moist *Schefflera-Hagenia* forest. steep mountain slopes with remnants of *Juniperus* forest and scrub of *Maesa*, *Acacia*, etc. 2000-3300m. TU/GD, WU, SU, AR, KF, BA, HA: not known elsewhere.
14	*Peperomia fernandopoiana*	Piperaceae	Herb		
15	*Phagnalon abyssinicum*	Asteraceae	Herb		Rocky areas in montane forests, dry montane slopes with low shrubs and herbs; 1800-3660m. EW, TU, GO, GJ,WU,SU, AR, BA, HA; not known elsewhere.
16	*Rhus glutinosa* A. Rich. subsp. *glutinosa*	Anacardiaceae	Shrub	VU	A shrub of forest margins and evergreen scrub, growing at altitudes of 1800-3300m. Distributed in TU,GD,GJ
17	*Satureja paradoxa*	Lamiaceae	Herb	NT	Moist soil in open and shady grassland, forests with *Podocarpus,* rarely a weed in tea-plantations; 1350-3500 m. GO GJ SU AR WG IL KF GG SO BA HA; endemic to Ethiopia.
18	*Thymus schimperi* subsp. *schimperi*	Lamiaceae	Herb	LC	Open grassland, on bare exposed rocks, on slopes and tops of mountains, sometimes growing near ditches in afromontane and afroalpine vegetation zones; 2250-4000 m; EW TV GD WU SU AR SD BA HA
19	*Vernonia rueppellii*	Asteraceae	Shrub	LC	Forest margins, grassland with evergreen scrub, *Croton-Calpurnea* woodland on montane slopes;2150-3000m. EW, GD,TU, WU,SU,AR, KF, SD, BA, HA; not known elsewhere.
20	*Erythrina brucei* Schweinf.	Fabaceae	Tree	LC	Edges and open places of upland forests or woodlands;1400-2600 m. WU WG GJ SU BA HA IL KF GO GG SO; not known elsewhere.

Appendix IV: Synoptic Table of the three community types.

Scientific Name	C_I	C-II	C_III
Olea europaea subsp.*cuspidata*	2.27	0.57	1.25
Hagenia abyssinica	2.09	0.86	0.25
Myrica salicifolia	1.91	0	0.25
Erica arborea	1.45	0	0.17
Juniperus procera	1.36	0.14	0.42
Dombeya torrida	1.18	0.14	0.08
Buddleja polystachya	0.45	0	0.25
Phytolacca dodecandra	0.36	0	0.33
Rhus glutinosa	0.36	0	0.08
Syzygium guineense	0.36	0.29	0.33
Conyza hypoleuca	0.27	0	0.25
Croton macrostachyus	0.27	0	0.42
Hypericum revolutum	0.27	0	0.25
Jasminum stans	0.18	0	0.08
Lobelia giberroa	0.18	0	0.08
Rhamnus staddo	0.18	0.14	0
Solanum indicum	0.18	0.14	0
Clerodendrum myricoides	0.09	0	0
Clutia abyssinica	0.09	0	0.08
Dodenea angustifolia	0.09	0	0.08
Dovyalis abyssinica	0.09	0	0.08
Dracaena afromontana	0.09	0	0.08
Gnidia lamprantha	0.09	0	0
Inula confertiflora	0.09	0	0.08
Satureja punctata	0.09	0	0
Maesa lanceolata	2.82	7.00	3.33
Brucea antidysenterica	0	3.57	1
Ilex mitis	1.27	2.29	4.5
Galiniera saxifraga	0.27	1.14	0.92
Vernonia rueppellii	0.73	1.14	1
Ekebergia capensis	0	0.71	0.17
Prunus africana	0	0.71	0.08
Satureja biflora	0	0.29	0
Solanecio mannii	0	0.29	0
Dichrocephala integrifolia	0	0.14	0.08
Maytenus arbutifolia	0	0.14	0
Pavetta abyssinica	0	0.14	0

Plantago palmata	0	0.14	0.08
Plectranthus assurgens	0	0.14	0.08
Pluchea discoridis	0	0.14	0
Rhamnus prinoides	0	0.14	0
Smilax aspera	0	0.14	0
Solanum benderianum	0	0.14	0
Sparmannia ricinocarpa	0.09	0.14	0.08
Olinia rochetiana	5.27	1.57	5.67
Maytenus addat	1.91	1.14	5.50
Schefflera volkensii	0.36	0	4.92
Myrsine melanophloeos	0.91	0.43	4
Podocarpus falcatus	0.45	0.86	3.33
Bersama abyssinica	0.18	1.57	3
Nuxia congesta	1.36	0.92	2.25
Discopodium penninervium	0.73	1	1.33
Schefflera abyssinica	0.18	0.71	1.25
Mikanopsis clematoides	0	0	0.83
Osyris quadripartita	0.64	0.14	0.75
Myrsine africana	0.45	0	0.67
Pittosporum abyssinicum	0.18	0.14	0.67
Euphorbia obovalifolia	0.09	0	0.58
Rhus retinorrhoea	0	0	0.58
Rosa abyssinica	0.45	0	0.58
Canthium oligocarpum	0.27	0	0.5
Laggera tomentosa	0	0	0.5
Apodytes dimidiata	0.18	0	0.42
Clausena anisata	0	0.14	0.33
Scolopia theifolia	0	0	0.33
Embelia schimperi	0	0	0.25
Hypoestes triflora	0	0	0.25
Rhus ruspolii	0	0	0.25
Rubus steudneri	0.18	0	0.25
Rubus steudnerii	0	0	0.25
Dregea schimperi	0.09	0.14	0.17
Halleria lucida	0	0	0.17
Helichrypsum odoratissium	0.09	0	0.17
Pentas schimperiana	0	0.14	0.17
Pycnostachys abyssinica	0	0	0.17
Calpurnea aurea	0	0	0.08
Commelina africana	0	0	0.08
Heracleum abyssinicum	0	0	0.08

Kalanchoe petitiana	0	0	0.08
Maytenus senegalensis	0	0	0.08
Oxalis corniculata	0	0	0.08
Pappea capensis	0	0	0.08
Ranunculus multifidus	0	0	0.08
Rumex nervosus	0	0	0.08
Scabiosa columbaria	0	0	0.08
Thalictrum rynchocarpum	0	0	0.08
Thunbergia alata	0	0	0.08
Thymus schimperi	0	0	0.08

Appendix V: Environmental and Carbon data (t ha^{-1}) of Biteyu forest

P	A	S	Lat.	Long.	AGBC	BGBC	SoilC	LitterC	Live-stock	Stump
1	2869	45	08,13,51.8	038,20,23.3	5.69	2.22	59.29	0.43	7	87
2	2740	42	08,13,50	038,20,21.6	43.77	12.93	64.58	0.23	22	74
3	2744	43	08,13,47.1	038,20,17.3	52.09	14.43	57.46	0.37	19	86
4	2970	47	08,13,42.1	038,20,01.7	12.01	4.33	67.08	0.27	5	40
5	2888	45	08,13,39.2	038,20,05.3	126.72	35.55	59.04	0.34	5	75
6	2819	43	08,13,36.5	038,20,07.3	56.11	17.94	63.52	0.28	16	69
7	2762	36	08,13,31.7	038,20,10.8	184.45	45.8	45.9	0.28	16	58
8	2744	50	08,13,29.1	038,20,13.2	151.41	38.91	43.56	0.32	17	48
9	2897	55	08,13,31.6	038,19,56.1	91.68	27.27	62.64	0.24	7	50
10	2846	46	08,13,30.8	038,19,59.6	146.05	39.88	58.06	0.23	9	45
11	2806	44	08,13,29.4	038,20,01.9	148.44	40.26	53.06	0.29	14	52
12	2752	38	08,13,25.7	038,20,00.1	121.61	32.63	49.3	0.35	18	43
13	2726	33	08,13,25.5	038,20,03.9	155.3	36.91	65.74	0.28	15	38
14	2734	45	08,13,26.5	038,20,12.9	257.01	55.5	63.5	0.31	24	75
15	2914	52	08,13,12.6	038,19,50.2	22.71	7.2	50.81	0.21	5	37
16	2895	48	08,13,13.8	038,19,53.4	7.78	2.84	40.5	0.22	8	29
17	2878	46	08,13,14.5	038,19,56.2	12.41	3.87	43.06	0.24	6	68
18	2835	40	08,13,16.1	038,19,57.1	28.57	8.57	58.78	0.24	7	80
19	2796	35	08,13,19.1	038,19,58.1	102.09	25.8	65.93	0.22	15	105
20	2748	31	08,13,20.0	038,20,01.1	86.24	22.28	58.75	0.23	17	102
21	2725	31	08,13,20.5	038,20,05.0	58.69	15.92	63.48	0.23	21	98
22	2899	45	08,13,48.58	38,20,07.87	20.65	6.72	53.28	0.23	6	36
23	2857	42	08,13,44.69	38,20,09.88	20.18	6.49	72.31	0.24	8	78
24	2820	40	08,13,42.66	38,20,11.84	115.71	32.92	36.31	0.18	10	88
25	2784	38	08,13,40.28	38,20,13.73	108.7	29.92	54.86	0.16	18	107
26	2744	36	08,13,37.07	38,20,15.78	143.15	39.28	68.97	0.18	18	94
27	2703	33	08,13,33.82	38,20,18.04	182.58	45.54	61.94	0.28	16	102
28	2825	37	08,13,12.11	38,20,04.26	31.66	9.15	38.35	0.14	8	82
29	2805	33	08,13,15.04	38,20,06.21	70.63	19.11	51.55	0.28	15	102
30	2768	32	08,13,16.58	38,20,09.22	49.67	8.16	59.6	0.29	14	113

Note: P = plot, A = Altitude, S = slope (^{o}),

Appendix VI: Environmental and Carbon data (t ha^{-1}) of Boter-Becho forest

Plot No.	Altitude	Slope(^{o})	X (E)	Y (N)	AGBC	BGBC	SoilC	LitterC
Plot 1	2471	20	308238	926896	168.2674	44.3103	202.7616	0.301
Plot 2	2435	30	308441	926753	290.6038	65.9711	98.1475	0.2206
Plot 3	2406	26	308620	926542	191.0923	42.8264	120.9624	0.7791
Plot 4	2391	14	308800	926385	112.8554	28.2	141.9887	0.2108
Plot 5	2524	23	308263	924125	153.9811	40.0855	116.8552	0.576
Plot 6	2430	15	308426	924272	115.58	31.0646	139.7	0.3061
Plot 7	2335	6	308647	924331	109.0424	14.9645	166.9521	0.2538
Plot 8	2259	2	308898	924486	150.9005	36.5749	99.9182	0.3102
Plot 9	2346	17	311964	923869	122.8234	33.4716	106.1528	0.2812
Plot 10	2300	19	311896	924075	150.3122	41.4013	129.0343	0.2157
Plot 11	2209	15	311756	924261	39.6667	7.404	145.6646	0.1705
Plot 12	2105	8	311512	924576	91.2984	26.6002	118.4157	0.1576
Plot 13	2603	4	311811	927073	366.302	48.3897	119.9048	0.2364
Plot 14	2626	9	311489	927034	243.7145	35.3326	104.5046	0.1811
Plot 15	2564	21	311209	926834	316.7224	46.448	182.4161	0.1981
Plot 16	2394	23	310786	926378	173.761	25.6573	197.6764	0.2753
Plot 17	2685	27	308967	921620	432.4308	55.4043	422.6282	0.3594
Plot 18	2668	33	309084	921995	263.26	36.0694	183.24	0.2168
Plot 19	2422	29	309113	923100	325.2645	82.912	121.1776	0.141
Plot 20	2412	14	309148	923562	350.1898	89.9103	95.2537	0.1209
Plot 21	2502	28	309107	922370	213.6957	52.7225	333.3734	0.276
Plot 22	2550	30	309086	922676	495.3426	62.1285	117.5708	0.2624
Plot 23	2302	18	311216	923758	135.3755	38.657	152.1474	0.265
Plot 24	2344	9	311097	924138	109.6756	17.5404	145.2884	0.1542
Plot 25	2245	8	311059	924424	64.7787	17.9489	495.7676	0.502
Plot 26	2165	7	311037	924664	128.8258	36.4032	167.7663	0.3722
Plot 27	2141	8	311005	924873	109.1683	31.1682	125.5439	0.2972
Plot 28	2392	29	308089	925699	140.2237	40.6693	118.2002	0.2272
Plot 29	2512	16	308315	925635	170.8862	46.6535	155.7854	0.4566
Plot 30	2490	7	308502	925554	254.5772	64.6619	120.4132	0.343
Plot 31	2447	5	308753	925483	132.4161	38.6838	180.7966	0.3404
Plot 32	2332	5	308937	925369	103.3021	29.9363	129.8439	0.3776
Plot 33	2313	10	309121	923955	315.2335	43.2716	138.1512	0.3002
Plot 34	2296	8	309121	924268	242.3888	63.1403	118.3035	0.2767
Plot 35	2283	12	309644	923543	169.8348	47.4745	106.8163	0.3038

Plot 36	2278	11	309542	923766	91.75	26.0385	162.2134	0.2685
Plot 37	2252	12	309415	924033	209.6924	55.6776	132.067	0.188
Plot 38	2753	37	308663	930382	180.3866	47.9875	175.646	0.3298
Plot 39	2830	34	308623	929941	203.6317	52.0512	146.5259	0.2073
Plot 40	2932	45	308681	929569	8.3536	4.0061	151.0268	0.2597
Plot 41	2895	39	308845	929342	21.4601	8.3968	122.9276	0.2204
Plot 42	2890	37	309008	929177	160.6802	44.6724	124.671	0.2401
Plot 43	2454	12	308297	927956	131.1676	34.833	109.0403	0.2569
Plot 44	2467	18	308378	927647	206.2067	53.3061	205.0565	0.3276
Plot 45	2374	21	308490	927378	272.6176	68.5808	109.8596	0.3672
Plot 46	2354	31	308674	927106	218.1115	58.3517	98.3628	0.5794
Plot 47	2314	20	308872	926873	103.1994	29.1951	131.1609	0.3526
Plot 48	2517	22	308985	927720	84.4744	25.3025	195.3936	0.3089
Plot 49	2431	19	309114	927319	141.8233	38.6967	151.8093	0.1436
Plot 50	2388	8	309219	927039	155.1665	42.6506	146.315	0.1801
Plot 51	2322	8	309363	926795	218.3934	53.2059	114.7551	0.2385
Plot 52	2276	6	309510	926500	146.87	41.5867	164.7177	0.142
Plot 53	2565	13	312379	927218	396.6451	85.0457	199.8964	0.4036
Plot 54	2599	7	312095	927190	229.6266	58.677	151.64	0.3102
Plot 55	2348	34	307714	924108	387.9254	88.064	181.6189	0.4603
Plot 56	2371	27	307918	924386	160.1227	39.376	153.68	0.4396
Plot 57	2356	29	308055	924652	326.3206	76.0717	551.9033	0.4162
Plot 58	2366	28	308289	924993	91.5102	25.7026	116.7333	0.2439
Plot 59	2447	25	308260	924377	174.6096	43.1952	159.1962	0.2841
Plot 60	2296	7	308562	924691	198.1276	51.8689	126.56	0.3919
Plot 61	2362	17	310814	926064	184.6567	49.28	116.6	0.3751
Plot 62	2340	17	310771	925765	231.5146	57.5038	118.83	0.3674
Plot 63	2236	15	310641	925433	119.105	32.0683	123.7673	0.2445
Plot 64	2374	20	311247	925907	199.7744	54.6934	278.8	0.3521
Plot 65	2274	25	311176	925638	131.3405	40.2464	106.65	0.2801
Plot 66	2252	43	311159	925383	218.9535	59.0507	139.551	0.561
Plot 67	2212	42	311202	925184	65.9845	20.4656	76.184	0.2301
Plot 68	2452	39	309778	927244	172.2044	25.3738	148.93	0.3728
Plot 69	2392	20	309672	926996	128.6959	35.9922	153.794	0.2445
Plot 70	2323	10	309588	926802	122.2545	34.6611	75.896	0.283
Plot 71	2783	32	308723	930707	400.795	97.196	270.13	0.4663

Appendix VII. Litterfall (g m^{-2}) for two sites of the Botor Becho forest

A. Low Disturbed sites of the forest

	Plot 1		Plot 2		Plot 3		Plot 4		Plot 5	
	Trap 1	Trap2	Trap 3	Trap 4	Trap 5	Trap 6	Trap 7	Trap 8	Trap 9	Trap 10
M1	140.7	160.22	155.72	176.62	159.58	163.68	145.8	165.8	127.75	157.85
M2	81.41	100.56	56.6	78.92	69.9	89.9	71.23	91.43	78	98.73
M3	15.71	35.91	21.11	42.31	35.69	56.79	35.69	55.79	46.69	66.79
M4	19.66	31.86	59.4	69.6	39.75	50.76	36.8	48.92	43.45	57.35
M5	22.56	28.66	26.69	34.78	58.16	64.26	29.45	39.43	40.25	49.45
M6	11.86	21.96	27.79	32.69	40.25	53.28	27.79	37.89	32.34	36.34
M7	20.46	20.85	72.93	83.96	49.22	59.42	25.12	35.22	23.57	34.67
M8	30.72	44.82	79.42	75.64	67.7	78.7	42.55	82.65	29.22	30.28
M9	35.87	46.67	44.22	46.42	85.46	94.56	87.35	98.45	72.34	92.44
M10	82.00	102.82	198.28	176.38	132.5	141.6	110.52	115.62	137.63	146.83
M11	76.2	86.2	119.7	125.6	121.2	115.7	120.83	125.93	140.22	156.42
M12	34.1	74.1	75.5	86.8	62.3	83.8	153.24	144.34	146.34	165.44

B. High Disturbed sites of the forest

	Plot 1		Plot 2		Plot 3		Plot 4		Plot 5	
	Trap 1	Trap 2	Trap 3	Trap 4	Trap 5	Trap 6	Trap 7	Trap 8	Trap 9	Trap 10
M1	65.35	85.45	77.15	67.25	90.4	70.85	100.44	98.24	76.19	84.27
M2	29.74	69.65	49.79	48.29	80.73	76.93	69.55	56.67	33.04	40.08
M3	43.07	53.06	42.51	38.61	30.35	35.65	37.76	47.66	21.24	38.34
M4	26.72	46.62	26.18	36.28	20.85	24.86	28.93	38.84	25.52	32.69
M5	27.82	37.62	48.84	38.74	27.27	28.38	38.57	35.02	37.47	30.87
M6	22.59	32.69	23.92	25.93	34.47	32.56	30.52	32.44	25.33	28.63
M7	24.55	27.65	29.84	30.95	50.29	54.36	19.76	25.58	17.31	25.68
M8	38.15	68.15	48.51	50.61	47.27	43.38	18.51	24.43	24.84	25.14
M9	48.75	78.75	39.42	49.32	36.56	38.96	34.84	32.76	24.74	28.84
M10	52.1	66.65	101	106.54	109.9	113.8	116.7	100.73	32.5	34.82
M11	75.2	89.3	77	87.62	66.4	78.43	107.4	105.68	43.1	63.16
M12	47.6	97.8	53.4	63.22	74.5	73.55	40.1	50.1	26.3	78.86

Appendix VIII. Mean Monthly Decomposition constant (k), residence time, half-life t (0.5) and t (0.95) for the study forest (n=5 for each value)

Month	LD				HD			
	K	Residence time (Rt)	Half life (t 0.5)	$t_{0.95}$	K	Residence time (Rt)	Half life (t 0.5)	$t_{0.95}$
	(month^{-1})	(Month)	(Month)	(Month)	(month^{-1})	(Month)	(Month)	(Month)
Mar	0.05	49.92	32.51	140.76	0.16	8.88	6.16	26.66
Apr	0.16	7.63	5.28	22.9	0.12	10.54	7.3	31.62
May	0.13	9.3	6.45	27.9	0.13	7.8	5.41	23.42
Jun	0.10	11.08	7.68	33.24	0.11	8.85	6.13	26.6
Jul	0.10	11.17	7.74	33.51	0.10	9.66	6.7	29
Aug	0.10	11.03	7.64	33.09	0.10	10.13	7.02	30.4
Sep	0.09	11.54	7.99	34.6	0.10	9.77	6.77	29.3
Oct	0.09	11.6	8.03	34.78	0.11	9.3	6.42	27.8
Nov	0.11	9.84	6.82	29.53	0.10	9.67	6.7	29.02
Dec	0.11	9.56	6.63	28.68	0.13	8.01	5.56	24.05
Jan	0.11	9.36	6.48	20.8	0.13	7.8	5.41	23.43
Feb	0.13	7.93	5.49	23.77	0.13	7.89	6.16	23.69