UNIVERSITY OF GHANA

COLLEGE OF BASIC AND APPLIED SCIENCE

FOREST COVER CHANGE AND CARBON STOCK DYNAMICS IN DRY AFROMONTANE FOREST OF ETHIOPIA

BY

NEGASI SOLOMON GEBRU

(ID. NO. 10543858)

THIS THESIS IS SUBMITTED TO THE UNIVERSITY OF GHANA, LEGON IN PARTIAL FULFILMENT OF THE REQUIREMENT FOR THE AWARD OF PHD ENVIRONMENTAL SCIENCE DEGREE

INSTITUTE FOR ENVIRONMENT AND SANITATION STUDIES

(IESS)

DECEMBER 2018

DECLARATION

I declare that this thesis is the outcome of my own research work and has not been submitted before for any other degree elsewhere. All references to other materials have been duly acknowledged.

10/12/2018

Negasi Solomon
(Candidate)

Date

10/12/2018

Supervisor: Prof. I.K. Asante
(Principal Supervisor)

Date

10/12/2018

Dr. Ted Y. Annang
(Co-Supervisor)

Date

10/12/2018

Dr. Opoku Pabi
(Co-Supervisor)

Date

10/12/2018

Dr. Emiru Birhane
(Co-Supervisor)

Date

ABSTRACT

Forest ecosystems are the main wellsprings of livelihoods for many people and play a key part in regulating climate change by capturing and sequestering carbon. However, forests have been altered and converted into other land uses and thus negatively impacting land cover systems. The change in forest cover affects many ecological, social and economic well-being. This study aimed at quantifying the effects of forest cover change on carbon stocks and ecosystem services values in the dry Afromontane forest area in northern Ethiopia. Forest cover changes were analyzed using Landsat images of 1985, 2000 and 2016. Vegetation parameters, litter and soil samples were collected from the field to quantify the carbon stock potential of the forest. Total area coverage of each land cover class was multiplied by the carbon on hectare basis of each land cover to quantify the change in carbon stock due to land cover change. Changes in ecosystem service values were estimated using land cover data of the year 1985, 2000 and 2016 with their ecosystem service value coefficients derived from ecosystem service database of The Economics of Ecosystem and Biodiversity. Household interviews and focus group discussions were used to assess community perceptions of forest cover change and its drivers. Results from the study indicated that the dense forests and open forests increased by 8.2% and 32.3% respectively between 1985 and 2000 while it decreased by 10.4% and 9.8% respectively from 2000 to 2016. Grasslands and cultivated land decreased between 1985 and 2000 by 37.3% and 5.5% but increased between 2000 and 2016 by 89.5% and 28.5% respectively. The study further showed that fuelwood collection, cultivated land expansion, population growth, free grazing and drought were the major drivers of land cover change in the study area. Soil erosion, reduction in pollinating agent like honey bee due to a reduction in flower production, flooding and drought were the perceived major impacts of forest cover changes. Possible solutions to the current state of deforestation as indicated by respondents include strengthening of forest protection and monitoring systems, improving soil and water conservation mechanisms, enhancing afforestation, creating awareness about the importance of forest ecosystems and zero grazing campaigns. The estimated mean total carbon stock was 181.78±27.06 Mg ha^{-1} in the dense forest and 104.83±12.35 Mg ha^{-1} in the open forest. The carbon stock for grassland, cultivated land and bare land were 108.77±6.77 Mg ha^{-1}, 76.54±7.84 Mg ha^{-1} and 83.11±8.53 Mg ha^{-1} respectively. Soil organic carbon and above ground carbon stock contributed more to total carbon stocks across the land cover types compared to below ground carbon and litter carbon. There was a marginal increase in carbon stock between 1985 and 2000 while carbon stock between 2000 and 2016 marginally decreased. There was a significant relationship between above ground vegetation properties and soil organic carbon (adj. $R^2 = 0.59$, $p = 0.003$). The estimated total values of ecosystem services for the study area was US$ 16.6, 19 and 18.1 million in 1985, 2000 and 2016, respectively. Generally, the study revealed that forest cover change substantially affected carbon stock and ecosystem service values in Wujig Mahgo Waren forest in northern Ethiopia. The study, therefore, recommends that enhanced efforts at conserving the forest ecosystem by all stakeholders to ensure an improved forest structure and increased ecosystem services delivery be pursued.

DEDICATION

This work is dedicated to my family, particularly to my mother Alemtsahay Kebede, my father Solomon Gebru, my beloved wife Tigist Gebremeskel and my dear daughter Samantha.

ACKNOWLEDGEMENT

Firstly, I want to thank the Almighty God for helping me to complete this PhD thesis. Secondly, I wish to express my sincere appreciation to my supervisory team made up of Prof. I.K. Asante, Dr. Ted Y. Annang, Dr. Opoku Pabi and Dr. Emiru for their noble guidance, continuous encouragements, genuine and constructive criticism during my research from the very beginning to what is now.

My appreciation also goes to the University of Ghana for admitting me to study for my PhD degree at the institution. I am highly grateful to the Transdisciplinary Training for Resource Efficiency and Climate Change Adaptation in Africa II (TRECCAfrica II) project for awarding me the PhD scholarship. I am also grateful to Mekelle University and the Steps Toward Sustainable Forest Management with the Local Communities in Tigray, Northern Ethiopia (ETH 13/0018) funded by NORAD/NORHED for funding my research.

The contribution of Mr. Hadgu Hishe in the remote sensing and GIS work is highly appreciated. I also acknowledge Birhane Weldu, Angesom Shushay and farmers and local administrators of Wujig Mahgo Waren for their assistance during the fieldwork. My appreciation also goes to drivers Melaku Abay and Muez Kiros for their smooth transportation during data collection.

I am thankful for the support I got from the staff of Land Resources Management and Environmental Protection at Mekelle University. I would also like to extend my appreciation to my classmates, friends, staffs and students at the Institute for Environment and Sanitation Studies, the University of Ghana for their support and advice during my PhD stay in Ghana. Again, I extend many thanks to my family and all my friends who supported me during the study period, without them my research

and my PhD studies, in general, would not have been completed on time. I appreciate anyone else whom I have missed to mention but have contributed for the successful accomplishment of this work. Thank you all.

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LIST OF ABBREVIATION AND ACRONYMS

AGB	Above Ground Biomass
ANOVA	Analysis of Variance
BGB	Below Ground Biomass
CBD	Convention on Biological Diversity
CDM	Clean Development Mechanism
Cm	Centimeter
CO_2	Carbon dioxide
CRGE	Climate-Resilient Green Economy
CS	Coefficient of Sensitivity
CSA	Central statistical agency
CSA	Community Supported Agriculture
DBH	Diameter at Breast Height
EFAP	Ethiopian Forestry Action Plan
EFM	Environmental and Forest Ministry
ERDAS	Earth Resources Data Analysis
ESV	Ecosystem Services Valuation
ETM	Enhanced Thematic Mapper
FAO	Food and Agricultural Organization
FDRE	Federal Democratic Republic of Ethiopia
GDP	Gross Domestic Product
GIS	Geographic Information System
Gg	Gigagram
GPS	Global Positioning System
Gt	Gigaton
GTP	Growth and Transformation Plan
H	Height

Ha	Hectare
HSD	Honesty Significant Difference
IET	International Emissions Trading
IPCC	Intergovernmental Panel on Climate Change
ISODAT	Iterative Self-Organizing Data Analysis Technique
Kg	Kilogram
LC	Litter Carbon Stock
MEA	Millennium Ecosystem Assessment
MEFCC	Ministry of Environment, Forest and Climate Change
Mg	Megagram
Mm	Millimeter
MoA	Ministry of Agriculture
Mt	Megaton
OLI/TRIS	Operational Land Imager and Thermal Infrared Sensor
Pg	Petagram
REDD+	Reducing Emission from Deforestation and Forest Degradation
SAS	Statistical Analysis System
SOC	Soil Organic Carbon
SNNPR	Southern Nations, Nationalities and People Region
TEEB	The Economics of Eco-system and Biodiversity
Tg	Teragram
TM	Thematic Mapper
UN	United Nations
UNEP-DTU	United Nations Environmental Programme-Denmark Technical University
UNFCCC	United Nations Framework Convention on Climate Change
US	United States of America

USGS	United States Geographical Survey
WBISPP	Woody Biomass Inventory and Strategy Planning Project

CHAPTER ONE

1. INTRODUCTION

1.1. Background

Forest ecosystems are main wellsprings of livelihoods for many people and play a crucial role in socio-economic development of many countries (Agrawal *et al.*, 2013; Chao, 2012). A forest ecosystem is an area which consists of plants, animals and micro-organisms in interaction with non-living physical features of the environment (Sen, 2018). They are essential natural resources that furnish a wide-range of ecosystem services such as moderating atmospheric carbon balance and thus climate change (Kumar *et al.*, 2014).

Ecosystem services are the benefits that people get from ecosystem processes for survival and quality life, such as food, carbon sequestration, nutrient cycling, air and water filtration, and flood amelioration (Costanza *et al.*, 1997). Carbon sequestration is the capture and storage of carbon that would somehow be produced or stay in the atmosphere or terrestrial systems (Herzog & Golomb, 2004). Terrestrial systems especially plants represent an important carbon store, estimated globally at 638 Gigaton, of which 44% is present in plant biomass (FAO, 2015a).

Ecosystem conditions affect the flow of ecosystem services (de Groot *et al.*, 2002; Styers *et al.*, 2010). For instance, land cover dynamics can change ecosystem services values by increasing the supply of some services while decreasing in others (Hu *et al.*, 2008; Kreuter *et al.*, 2001; Polasky *et al.*, 2011). This might have an effect on the long-term sustainability of ecosystems.

Degradation of forest ecosystem has an effect on the composition of greenhouse gases in the atmosphere leading to global warming (Köhl *et al.*, 2015). Changes in land use including forest clearance for agriculture, settlement and industrial expansion have contributed about 136 (±55) Gt carbon or one-third of total anthropogenic emissions of carbon dioxide (CO_2) to the atmosphere over the past 150 years (Gómez *et al.*, 2006; Watson *et al.*, 2000). Carbon emissions from deforestation and forest degradation are the second largest source of anthropogenic carbon emissions (Le Quere *et al.*, 2009; van der Werf *et al.*, 2009). Studies indicate that land cover change has significant effects on carbon stock. For instance, land cover change significantly affected carbon stock by impacting the aboveground biomass and soil organic carbon in Malagasy rainforest, Madagascar (Andriamananjara *et al.*, 2016). On the other hand, changes in land cover from non-forest to forest ecosystems through exclosure, afforestation and reforestation activities are known to increase the carbon sequestration potential of an area. For example, Mekuria *et al.* (2009a) found that the introduction of exclosures on degraded free grazing lands increased carbon stocks in the lowlands of Tigray, Ethiopia.

In Ethiopia, forest resources provide goods and services of significant values to the economy, environment and society (Moges *et al.*, 2010). The vegetation types in Ethiopia are categorized into nine types including the Afromontane vegetation either dry or moist. The Afromontane vegetation covers more than 50% of the highlands in Ethiopia with a high diversity of plant species, therefore it is one of the prominent biodiversity haven in the country (Teketay, 1996). The dry Afromontane forests are composed of a number of indigenous tree species dominated by an association of Juniperus-Podocarpus or only Podocarpus species (Tesema *et al.*, 1993; Wubet *et al.*, 2003).

The dry Afromontane forests provide a range of ecosystem services including provision of diverse habitats for fauna and fodder for livestock, non-timber forest products, watershed protection including groundwater regulation, flood control and soil erosion prevention and control and carbon sequestration (Asfaw *et al.*, 2013; Price *et al.*, 2011; Solomon *et al.*, 2017). The dry Afromontane forest has not been managed sustainably and has undergone gradual degradation through human activities over a period of time. However, the Wujig Mahgo Waren forest is one of the remnants of the dry Afromontane forest in northern Ethiopia which continues to provide essential services for livelihoods of the people.

Forests are exposed to huge pressure due to increase in human population, widespread poverty, and overgrazing resulting in deforestation and degradation (FAO, 2011). For instance, an estimated 129 million hectares (size of South Africa) of forests have been lost since 1990 (FAO, 2015a). A hundred years ago, about 40% of the landmass of Ethiopia was covered by forest but currently, only about 3% forest cover is left (Babulo, 2007).

Generally, forest management which involves the participation of the forest fringe communities has been considered to yield sustainable outcomes (Bray *et al.*, 2009). However, in Ethiopia forest management practices over the last 50 years has not been community-based. For instance, local communities involvement in forest management was restricted thereby reducing access to the resources and use rights of the communities (Engida & Teshoma, 2012). This approach has negatively affected the forest resource through illegal and indiscriminate exploitation of the resources by the communities. Inquests of a solution, in 1990s different interventions have been introduced by the government and the local communities to tackle deforestation and forest degradation. Establishment of exclosures, construction of soil and water conservation practices and afforestation were among the main activities that have been introduced (Giday *et al.*, 2013; Mekuria *et al.*, 2009b).

Deforestation is an environmental problem in Wujig Mahgo Waren forest. However, a thorough understanding of forest cover dynamics and the impact of these dynamics on ecosystem service and values through empirical evidence is lacking. Furthermore, the perception of the farmers regarding forest cover change, their drivers, and possible solutions at landscape level has not been studied. Therefore, there is a need to study the change in forest cover, its effect on carbon stock and ecosystem services values and the perception of the community towards forest cover change, their drivers, and possible solutions in Wujig Mahgo Waren forest in the northern part of Ethiopia.

Exploring changes in forest cover and drivers, carbon stock and valuation of ecosystem services is fundamental to generate awareness and add to the emerging knowledge on sustainable management of natural resources. It also enhances decision making for allotment of limited capitals and formulates policies and delivers an impetus to safeguard the land cover type that offers the best-valued services.

1.2. Statement of the problem

The concentration of carbon dioxide in the atmosphere is increasing (Keeling, 2009). The increase in carbon dioxide in the atmosphere has a direct impact on climate change. Forests are considered as a natural solution for climate change through carbon sequestration (Kumar *et al.*, 2014; SILVA *et al.*, 2018). However, deforestation and forest degradation has affected the potential of forest to sequester carbon and produce other ecosystem services (Andriamananjara *et al.*, 2016).

The Wujig Mahgo Waren forest is one of the remnant Afromontane state forest patches in northern Ethiopia. Like many other parts of the country, the problem of deforestation poses an environmental threat in Wujig Mahgo Waren forest. The deforestation might have an impact on flow of ecosystem services. However, studies on the rate and degree of forest cover change and its impact on carbon stock and ecosystem service values is lacking regardless of its significance on climate change agenda and applicability for REDD+. In addition, while the community based forest management approach is said to be in place, its effectiveness was scantly studied. Empirical studies on the understanding of local community regarding forest cover change and its drivers were also limited to some areas. Thus, it is important to assess and monitor the status and drivers of forest cover change and impacts on communities' ecological dynamics. Further, it is important to assess the impact of forest cover change on ecosystem service provision and values.

1.3. Objectives

The main objective of this study is to assess forest cover change and its effect on the dynamics of carbon stock and ecosystem services values in Wujig Mahgo Waren forest in northern Ethiopia.

Specific objectives for the study are to:

i. quantify the magnitude and the percentage change of the Wujig Mahgo Waren forest.

ii. assess species composition and vegetation structure of the different land cover types in the study area.

iii. quantify levels of carbon stocked in different land cover types and estimate changes in carbon stock in the Wujig Mahgo Waren forest.

iv. evaluate the functional relation between soil organic carbon stock and above ground vegetation properties.

v. estimate the impact of land cover change on ecosystem service values over the past three decades.

vi. assess the perception of the community towards forest cover change, their drivers, impacts and possible solutions.

1.4. Research questions

- How much forest cover has changed over three decades?
- How is the species composition and vegetation structure varying across the different land cover types in the study area?
- What volume of carbon is stocked in the study area?
- Which carbon pool contributes most to the carbon stocked in the study area?
- What volume of the carbon stocked has changed due to forest cover change in the year of 1985, 2000 and 2016?
- How do soil organic carbon and above ground vegetation properties relate functionally?
- To what extent have ecosystem service values of the forest changed over the three decades?
- How do communities perceive forest cover change, the drivers of change and their impacts on community livelihood?

CHAPTER TWO

2. LITERATURE REVIEW

This chapter reviews existing literature on forest, status of forest cover change, drivers of forest cover change, impact of forest cover change, climate change, carbon sequestration, linkages between forest and climate change, linkages between forest and carbon sequestration, linkages between forest cover change and carbon sequestration, forest ecosystem services, linkages between land cover change and ecosystem services, methodological approaches and issues.

2.1. Forests as a resource

Forests are sources of many ecosystem services needed for our survival (Nkonya *et al.*, 2016). The Millennium Ecosystem Assessment categorized ecosystem services as provisioning services (food, freshwater, and fibre), cultural services (places for recreation and education), supporting services (nutrient cycling and soil formation) and regulating services (flood control and climate regulation) (Daily, 1997; MEA, 2005b).

Forests are a source of livelihood for more than 1.6×10^9 population of the globe in varying degrees (CBD, 2009). Many people obtain their daily food from the forest (Vira *et al.*, 2015). Forests are the sources of commercial products such as timber, clothing and pharmaceuticals (CBD, 2009). They are also a source of income (Babulo *et al.*, 2009; Shackleton *et al.*, 2007). For instance, Vedeld *et al.* (2007) revealed that on average, 22% of the total income in 17 countries were from forests. In Zambia, forest plays a key role in the income of people belonging to poor households (Kalaba *et al.*, 2013). The forestry sector is known to play an important role in employment by way of providing employment for 10×10^6 people (FAO, 2010a). The report of

Whiteman *et al.* (2015) showed that around 12.7 million individuals are working in the forestry sector.

Ecologically intact forests are known to store and purify drinking water (CBD, 2009). They are a means to water availability and cooling (Ellison *et al.*, 2012; Ellison *et al.*, 2017; Hesslerová *et al.*, 2013; Syktus & McAlpine, 2016). According to Calder *et al.* (2007), both upstream and downstream areas obtain water from forest to satisfy their domestic, manufacturing and environmental needs. The role of providing and protecting water resources has been attributed to forests for generations (Liniger & Weingartner, 1998). Cooling of environment depends on the ability of trees to catch and redistribute the sun's vitality (Pokorny *et al.*, 2010). Results from Li *et al.* (2015) reveal that cooling effect depends on the type of the forest; strong cooling effect in tropical forests throughout the year, moderate cooling in temperate forests in summer and net warming in boreal forests annually. Forests being used in watershed management technique plays a significant role in reducing the risk of pollution and creating and preserving healthy stream ecosystems (Escobedo & Nowak, 2009).

Furthermore, trees can initiate rainfall (Ellison *et al.*, 2017). Forests have the capacity to filter precipitation resulting in purified ground and surface water for communities (Neary *et al.*, 2009). Evapotranspiration plays a great role in the process of precipitation. For instance, 70% of rainfall in the region comes from evapotranspiration of the Amazon forest (van der Ent *et al.*, 2010).

Also, forests are the home of most plants and animals (CBD, 2009). For example, a comparison of species richness and frequency of occurrence between landscape types the Iberian Peninsula indicate that riparian woodlands are important habitats for many animals (Virgós, 2001). Moreover, the coastal forests in Tanzania are home to 50 endemic plant species, several rare mammals, seven bird species, reptiles and amphibians, an invertebrate fauna and many more (Burgess *et al.*, 1992). In addition, Duivenvoorden (1995) also found 1,077 tree species which he categorized into several genera and families in the middle Caquetá area, Colombian Amazonia. Wittmann *et al.* (2006) recorded more than 900 flood tolerant tree species in Amazonian várzea forests. Slik *et al.* (2015) found more than 53,000 tropical tree species and 124 tree species across temperate Europe. Bernard and Fenton (2002) captured 3,222 bats in the central Amazonian forest, Brazil. Beaudrot *et al.* (2016) also recorded 511 terrestrial mammals and bird population, including 244 species from 15 protected areas of tropical forest on three continents. Rovero *et al.* (2014) revealed that at least 32 mammal species exist in Udzungwa Mountains of Tanzania.

Forests have functioned responsibly in climate change mitigation by reducing carbon emissions. After oceans, forests are the world's largest storehouses of carbon. Soares-Filho *et al.* (2010) predicted that all protected areas in Brazilian Amazon have the capacity to elude 8.0 ± 2.8 Pg of carbon releases by 2050. Using ground data and durable carbon research in forest, Pan *et al.* (2011) estimated a total carbon sink of 2.4 ± 0.4 Pg carbon yr^{-1} globally for 1990–2007. Mangroves sequester 1,023 Mg carbon ha^{-1} placing them among the most carbon-rich forests in the tropics (Donato *et al.*, 2011). According to Bonan (2008), forests play an essential role in sinking anthropogenic carbon emissions by storing 45% of the terrestrial carbon in their biomass and soils. Keith *et al.* (2009) found that an average of 1,867 tons of carbon ha^{-1} in dead and living biomass in E. regnans forest in the O'Shannassy catchment of the central highlands of Australia.

Forests contribute significantly in mitigating flood and storm damage. For example, in India investments in forest management and restoration are essential to control flood damage and safeguard people and assets (Bhattacharjee & Behera, 2017). Zhang *et al.* (2015a) indicated that during heavy rains, tree roots play a crucial role in reducing the speed of the flow, consequently reducing soil erosion and material loss. They found that change in urban green spaces reduced runoff rate from 23% in 2000 to 17% in 2010 in Beijing, China. Brookhuis and Hein (2016) have also indicated that even a slight removal of forest can cause substantial rise in flood risks in Trinidad. According to Bathurst *et al.* (2011), forest cover offers substantial mitigation benefits for moderate rainfall events in Latin American environments.

Forests are important in soil erosion reduction. Abandonment of cultivated land in the 20th century improved vegetation cover resulting in a decrease of soil loss in mountain areas (García-Ruiz, 2010). Hartanto *et al.* (2003) indicated that soil surfaces are often protected due to surface roughness created by sapling and dead woods, thus preventing soil erosion. In 2015 about 25.1% of total forest area was allocated for the protection of soil and water resources across the globe (Miura *et al.*, 2015). This implies that the role forests play in soil and water conservation has received wider attention across the world.

Forests absorb a variety of airborne chemicals, such as nitrogen dioxide, carbon monoxide and sulfur dioxide (Beckett *et al.*, 1998). A study conducted in the United States revealed that urban trees improved air quality by removing huge amounts of air pollutants. However, the removal of pollutants differed among cities with total removed annual air pollutants estimated at 711,000 metric tons (Nowak *et al.*, 2006). Similar results were observed in the study of Selmi *et al.* (2016) where public trees managed by the city removed significant amounts of pollutants between 2012 and 2013 in Strasbourg city, France. Similarly, Jeanjean *et al.* (2017) showed that trees trap air pollutants by up to about 7% at the Marylebone monitoring station.

Furthermore, Yang *et al.* (2005) also found that trees absorbed about 1,261.4 tons of airborne contaminants in central part of Beijing as at 2002.

In addition, forests are sources of natural and artificial medicines. Tropical forests are composed of various plant species including those used for medicinal purposes. They are a source of new drugs and medicines for many people in the unindustrialized world (Balick *et al.*, 1996; Balick & Mendelsohn, 1992). Many important drugs can be obtained from natural plants, such as codeine, morphine, quinine and digitalis (Abelson, 1990). Kitula (2007) found 45 plant species used for healing about 22 human diseases in New Dabaga Ulongambi Forest Reserve, Tanzania.

Forests also have aesthetic values. A study in Georgia by Majumdar *et al.* (2011) showed that people readily volunteer to pay an average of US$11.25 for visiting urban forests. Based on their estimation, the annual value of urban forests can be estimated at a low of $81 million and a high of $167 million. Findings from Tanzanian also showed that US$ 10,000 were generated from the Amani Nature Reserve annually as revenue from eco-tourism (Shoo & Songorwa, 2013). Deng *et al.* (2010) indicated that urban forests can play a significant role in city beautification and attract tourists, which substantially contribute to tourism satisfaction.

From the review, we can understand that the contributions of forests to our survival are infinite. Without the services from forests, it will be unthinkable to consider human survival. However, the use of forests varies spatially. One country may use their forest resources for watershed protection and tourism while another may use theirs for paper production. Besides, it is also difficult to quantify the market price for some of the services provided by forests. A human being is exceptionally reliant on the flow of forest ecosystem services, yet this is subject to the way in which environments are influenced by human activities (Campos & Corrales, 2005). Generally, the nature and significance of these services have largely been overlooked to the point that their disturbance or loss has featured their significance (Daily, 1997).

2.2. Status of forests in the World

Forests currently cover around 4 billion ha (30%) of the total land of the Earth (MacDicken *et al.*, 2016). Globally, forest cover is decreasing due to the expansion of cultivated land to satisfy the continuously growing human population's food demand. However, a 50% reduction in the extent of forest loss has been observed (FAO, 2015a). Between 1990 and 2015, a 0.13% annual forest loss was recorded, making almost 129 million ha of total forest lost (Miura *et al.*, 2015). A net yearly forest loss of 3.3 million ha was recorded between 2010 and 2015, which resulted from 7.6 and 4.3 million ha of annual forest loss and gain respectively (FAO, 2015b).

Dynamics of forest cover and nature of forests vary among countries. For example, between 1990 and 2015, in tropical countries, especially Africa and South America, huge losses of forest cover were observed (MacDicken *et al.*, 2016). Similarly, Hansen *et al.* (2013) found the rate of forest loss to be increasing by 2,101 Km^2 in the tropics. According to their study, Brazil recorded a substantial reduction in deforestation while other tropical countries such as Indonesia, Malaysia, Paraguay, Bolivia, Zambia and Angola showed increasing forest loss. In Bhutan, where the total forest land covering of the country is 60%, Bruggeman *et al.* (2016) reported that the country's forest cover remained unchanged. Between 1995 and 2011, Mayes *et al.* (2015) observed a reduction of 7% in forest area in Miombo Woodlands, Tanzania. Findings from North Ethiopian Afro-Alpine forest showed severe deforestation with 63% reduction in forest cover between 1982 and 2010 (Jacob *et al.*, 2015). Stibig *et* *al.* (2014) also indicated a reduction of 32 million ha of forest cover between 1990 and 2010 in Southeast Asia. However, a net increase of forest cover was recorded from 1985 to 2012 in eastern Europe (Potapov *et al.*, 2015).

Between 1990 and 2015, a 10% (62 million ha) reduction of primary forest area was observed in tropical countries (Morales-Hidalgo *et al.*, 2015). Globally, from 1990 to 2015 a net reduction of 2.5% (31 million ha) of primary forest area was detected (Keenan *et al.*, 2015). However, an increase in forest cover was reported in temperate and boreal countries by 6 and 30 million ha, respectively (Keenan *et al.*, 2015). Since 1990, the temperate and tropics recorded a stable or an increasing rate of afforestation (Payn *et al.*, 2015). Generally, the review shows a net decline of forest cover around the globe though improvements can be observed in some countries.

2.3. Forest resources in Ethiopia

The new forest development, conservation, and utilization policy proclamation (No. 1065/2018) has defined forest as "land spanning at least 0.5 ha covered by trees (including bamboo) attaining a height of at least 2 m and a canopy cover of at least 20% of trees with the potential to reach these thresholds in situ in due course" (MEFCC, 2016).

Ethiopia has a total of 18 agro-ecological zones and 49 sub-zones (MoA, 2000) owing to varied elevations, landscapes, and climates. Due to the variation in agro-ecology, the country boasts of diverse vegetation types such as the tropical rain forest, cloud forest, mountain forest, desert scrubs and parkland agroforestry, and indigenous multi-strata agroforestry system (Moges & Tenkir, 2014; Negash, 2013; Teketay *et al.*, 2010).

The country is assessed as containing an estimated 6,500 to 7,000 higher plants species, of which 780 to 840 are unique to the country representing 12% of the flora (Teketay, 2000). The country also contains two biodiversity hotspot areas, namely the Eastern Afromontane and horn of Africa biodiversity hotspots (Lemenih & Woldemariam, 2010). Officially, nine vegetation types have been recognized in the country such as the dry and moist evergreen montane forest, Acacia-Commiphora deciduous forest, Combretum-Terminalia deciduous woodland, lowland semi-evergreen forest, desert and semi-desert scrub, wetland vegetation, evergreen scrub and Afroalpine and subafroalpine forests (Lemenih & Woldemariam, 2010). Later, the vegetation was classified into twelve comprising the desert and semi-desert bushland, Acacia-Commiphora woodland, wooded grassland, Combretum-Terminalia woodland and wooded grassland, dry and moist evergreen Afromontane forest and grassland complex, transitional rain forest, Ericaceous belt, Afroalpine belt, freshwater vegetation, riverine vegetation, and Salt-water lakes and pan vegetation (Friis *et al.*, 2010). Natural forests are dominant in the country, though some plantation forests also exist.

Although no formal study has been conducted on the actual high forest cover in the country, the assessments by WBISPP (2005) based on national inventory appears to be more dependable than the estimate of the other. For example, the estimates of natural high forest cover of FAO (2005b) were 3 times higher than WBISPP (2005) estimates. According to WBISPP (2005), natural high forests, plantations, woodlands and bamboo forests are considered as forest in Ethiopia which are estimated at 35.504 million ha, of which high forest accounts for 11.5%, 83% for woodland, 3% for plantation and 1.4 % for bamboo forest. If we include shrublands, the total vegetation cover of the country will be estimated at 61.91 million ha. FAO (2015b) estimates also showed that 40.63 and 12.5 million ha of woodland and forest land respectively, are available in Ethiopia.

In Ethiopia, forests perform key roles in the production, protection and conservation processes. They are the most important source of income for many households (Fikir *et al.*, 2016; Mamo *et al.*, 2007; Yemiru *et al.*, 2010). They supply most of the forest products consumed in the country (Teketay *et al.*, 2010). Forests are also a source of energy contributing about 78% of the total domestic energy consumed (Teketay *et al.*, 2010). Moreover, forests reduce runoff, soil erosion and conserve biodiversity (Haregeweyn *et al.*, 2012; Teketay *et al.*, 2010). They essentially help to mitigate climate change by storing carbon in its above ground biomass amounting to 2.7 billion tons (Moges *et al.*, 2010). They are also a source of great rivers (Teketay *et al.*, 2010). Generally, the forestry sector contributes 9.0% to the country's gross domestic product (Teketay *et al.*, 2010).

2.4. Status of forest in Ethiopia

In the 1900s, the high forest of the country was about 40% of the total area (EFAP, 1994). This figure was decreased to 16% in the 1950s with a rate of deforestation of about 200,000 ha per year (Davidson, 1989). According to Reusing (1998), who used satellite imagery to monitor forest cover change, the natural high forest cover drastically reduced to 4.75% between 1973 and 1976, and then, lessened to 3.93% between 1986 and 1990 with the annual forest loss growing to about 39,000 ha. In the 1990s, the forest cover further went down roughly to around 3% (Bishaw, 2001). The high rate of deforestation at the time was mostly ascribed to pulverization of state-possessed forests in and after the 1991 change of government (Bekele, 2003).

Regardless of the high deforestation rate, FAO (2010b) report shows that the forest cover rose to approximately 11% of the country's area.

Deforestation, mainly in the highland areas where most of the farming activities have taken place over a long period of time, has been an extreme and relentless practice in Ethiopia (Darbyshire *et al.*, 2003; Dessie & Christiansson, 2008; Eshetu & Högberg, 2000; Teketay, 2001). According to Place *et al.* (2006), the highland areas of Ethiopia are suitable for agricultural production making them attractive for settlement over extended periods of time. This is believed to be the main driver of forest loss in the country (McCann, 1997). At the national level information regarding forest cover change and its causes is lacking due to the lack of assessment on regular basis. In Ethiopia, most of the information regarding deforestation and forest degradation is existent in particular forest areas. Currently, the report on forest reference level submitted by the Ethiopian government to the United Nation Framework Convention on Climate Change (UNFCCC) included some information regarding deforestation at the national level (FDRE, 2016).

The WBISPP (2005) identified population growth and the concomitant upsurge in agricultural land as major causes of deforestation in Ethiopia. Between 1990 and 2014, about 1.24 million ha of natural high forest were deforested from areas like the Southern Nations, Nationalities, and Peoples' region (SNNP), Oromia region, and Gambela region for cultivated land expansion (WBISPP, 2005).

FAO estimates showed 0.93% annual rate of forest loss between 1990 and 2010 in Ethiopia (FAO, 2010b). The Afromontane forest in Ethiopia depicted a fluctuating trend. For instance, a study by Jacob *et al.* (2017) recognized an increase in forest in the western and eastern verge of the Semien mountain national park while the central area covered by dense forest decreased. Moreover, Kidane *et al.* (2012) found a stable Afromontane rainforest in the Bale mountains of Ethiopia while Hailemariam *et al.* (2016) showed a decrease in forest cover in the Bale mountain eco-region between 1985 and 2015.

The rate of forest loss and its related effects on forest cover in Ethiopia has shown a conflicting pattern over time. Some claim deforestation dates back to 5000 years ago with others contending that the biggest deforestation has been occurring within the past 150 years (Ayenew, 2014; Lemenih & Woldemariam, 2010; Reusing, 1998). According to the MEA (2005b)'s report, Ethiopia has lost 90% of its original forest cover. Based on the pattern of deforestation, the nation has been classified under the late forest transition phase (Noriko *et al.*, 2012). This suggests that a slowing down of the rate at which deforestation occurs in the little portion of the remaining forests will, in the long run, move the country's forest into the post-transition phase (Noriko *et al.*, 2012).

2.5. Drivers of deforestation

Deforestation can be initiated by different agents for different purposes. The forces that propel individuals to remove forests can be called *drivers* (Chakravarty *et al.*, 2012). Deforestation has been drilled by people for many years before the commencement of technological advancement. Deforestation has been practised by humans for so many years (Hance, 2008). The main objective of deforestation is to get open ecosystems in order to obtain suitable land for cultivation and grazing. This is done by clearing closed forests (Lanly, 2003). To save our forests, it is imperative to identify the direct and indirect causes of deforestation. Knowing the drivers of forest loss and degradation are essential for designing strategies and interventions that target change in the existing trend of forest activities toward a more environmentally sound practice.

Many studies have categorized the causes of deforestation as direct and indirect causes. Direct drivers of deforestation are relatively easy to detect but the indirect drivers which are commonly at the core of deforestation are challenging and difficult to quantify (Humphreys, 2014; Sands, 2013). Agricultural land expansion, fuelwood collection, natural and artificial fires, overgrazing, mining, urbanization and infrastructural development, war and air pollution are some of the direct causes of deforestation (Calvas *et al.*, 2013; Dewan *et al.*, 2012). However, overpopulation and poverty, land ownership systems, biased land sharing and resources, financial problems, economic programs, market proximity and demand, corruption, lack of strong forest management strategies and underestimation of the forests' value are some of the indirect causes of deforestation (Bewket, 2002; Chakravarty *et al.*, 2012).

According to Noriko *et al.* (2012), 73% of all forest loss in lower-income countries are caused by agricultural expansion. Out of the total deforested land, an agricultural expansion for both commercial and subsistence agricultural purposes caused 40% and 33% of deforestation respectively (Noriko *et al.*, 2012). The remaining 27% loss was caused by other drivers such as infrastructure (10%), urban expansion (10%) and mining (7%) activities. However, in developed countries such as Australia, Portugal, and Sweden, global market forces are listed as major drivers of deforestation (Beilin *et al.*, 2014).

Cultivated land expansion and unsustainable fuelwood utilisation are recorded as the two most prevalent drivers of deforestation and forest degradation in Ethiopia (Dessie & Christiansson, 2008; Dessie & Kleman, 2007; Jarno, 2016; Kindu *et al.*, 2013;

Zeleke & Hurni, 2001). Ensermu and Abenet (2011) reported that agricultural land increased from 9.44 to 15.4 million ha between 2001 and 2009 in Ethiopia. The change is essentially attributed to the expansion of agricultural land at the expense of forest land. According to the Climate Resilient Green Economy strategy (CRGE) report, the total cropland of the country is expected to reach 27 million ha, with business-as-usual annual growth rate of 3.9%, which is needed for a crop growth target of 9.5% per year to ensure food security and poverty alleviation according to the Growth Transformation Plan (GTP) (CRGE, 2011). These new agricultural lands mainly come from the forest.

Also, deforestation and forest degradation can be aggravated as a result of resettlement programs and migration. Resettlement program in some regions, for the sake of food security, have seen the conversion of large forest lands into agricultural lands. For instance, in the period between 2000 and 2004, a total of 220,000 households (1.2 million people) were resettled in Oromiya, SNNP, and Tigray mainly in natural forest areas where they cleared the forests for crop cultivation (Lemenih & Woldemariam, 2010). Furthermore, the ever-increasing population has resulted in increased demand for fuelwood. The total population of the country is predicted to grow to 134 million in 2030 at a growth rate of 2.62% per annum.

Other underlying factors attributed to deforestation have also been poverty, the lack of a sense of ownership and lack of clear legal policy framework for forest regulation, conservation, and utilization (Bishaw, 2001; Zeleke & Hurni, 2001). According to Reddy (2011) and Hurni (1993), demographic factors, especially an increase in local population is considered an important cause of land cover change in Ethiopia. Erratic rainfall, land degradation, drought, plantation expansion, housing, charcoal making, access to road, overgrazing and creation of market access are also described as major drivers of forest cover change in Ethiopia (Demissie *et al.*, 2017; Kindu *et al.*, 2013)

2.6. Impact of deforestation

Changes in the Earth's forest cover affect the capacity of ecosystems to provide important services and biodiversity. It also has an effect on biogeochemical cycles involving carbon, water and other nutrients (Foley *et al.*, 2005). Knowledge of these processes is essential to managing ecosystems sustainably. In spite of the fact that there are a variety of vegetative lands in Ethiopia, these assets are under huge pressure from deforestation, forest debasement, overexploitation, overgrazing, natural surroundings adversity, invasive species and pollution (Moges & Tenkir, 2014).

All over the world, the existence of human beings is being threatened by deforestation and forest degradation, henceforth, Ethiopia is not a special case. In Ethiopia, deforestation has resulted in multifaceted antagonistic results including the loss in biodiversity, soil erosion and decrease in soil richness, deficiency of forest products, drought and starvation, poverty, reduction in rural income, loss of wetlands, flooding, water scarcity, siltation of water bodies and desertification, just to mention a few (Bishaw, 2001; Gebre Egziabher, 1986; Teketay, 2004; Yemishaw *et al.*, 2008). This implies that deforestation has caused environmental, social and economic problems in Ethiopia.

Ethiopia is extremely susceptible to soil erosion because of the mountainous nature of the land. Yesuf *et al.* (2005) revealed that land degradation in the nation has resulted in serious land erosion. Therefore, crop and grass yield in the highlands of Ethiopia have decreased. The loss of soil fertility and land degradation prompt budgetary loss of around 2% of GDP in Ethiopia (EFAP, 1994). According to Zeleke and Hurni

(2001), soil erosion at the Ethiopian plateaus has a huge national and universal effect on water streams and sedimentation, specifically to the countries which utilize the Blue Nile such as Egypt and Sudan.

Woody plant species have decreased due to the act of cutting down trees and bushes for various purposes. *Juniperus procera, Podocarpus falcatus, Hagenia abyssinica, Cordia africana, Olea europaea subsp. cuspidata, Prunus africana* and *Combretum molle* are the most common highland tree species in Ethiopia. These have in recent times been decreased to scattered trees in mountainous zones for several reasons (Zegeye, 2017). The existence of *Boswellia papyrifera* and *Boswellia commiphora,* two most important gums and resin-bearing species are now under threat (Adefires, 2006; Eshete *et al.*, 2005; Gebrehiwot *et al.*, 2003), resulting in the loss of their economic value.

Deforestation is also a major agent of climate change. This is because deforestation influences wind flows, water vapour streams and the assimilation of sun oriented vitality which plainly affects nearby and global climates (Chomitz & Buys, 2007). For instance, about 20% of the global greenhouse gas emissions come from tropical deforestation. In Ethiopia, deforestation and forest degradation have brought about an emission of 65 Mt carbon dioxide (CO_2) equivalent, representing 40% of the aggregate emission from the nation (Moges & Tenkir, 2014). Climate change sequentially has adverse effects on the world.

The shortage and eventual extinction of forest products leading to serious loss of income is another consequence of deforestation. The immense contribution of the forestry sector to the Ethiopian economy has been curtailed due to the practice of misusing forest resources, an act which dates many years back (Zegeye, 2017).

Hansen (1997) reported a US$ 45 billion reduction in forestry income due to deforestation in tropical forests. All potential future incomes and businesses depend on the sustainable use of forest resources. Enbakom *et al.* (2017) opined, from their observation of the communities around Arba Minch Zuria Woreda, southern Ethiopia, that deforestation affects land productivity which has negative effects on people's livelihood. Tilahun and Fayera (2012) also observed the devastating effects of deforestation on rural livelihood in Kuyu Woreda, central Ethiopia.

Deforestation also interrupts the global hydrological cycle (Bruijnzeel, 2004). Usually, large amounts of water is transported from plants to the atmosphere through the process of transpiration. In Amazonia, a less intense water cycle was expected following a basin-wide scenario of deforestation (D'Almeida *et al.*, 2007). Werth and Avissar (2002) indicated that precipitation, evapotranspiration and cloudiness reduced due to deforestation in the Amazon. The impact of deforestation did not stop in the Amazon Jungle and its environs; it also reduced precipitation in other regions of the world. In the central Rift Valley of Ethiopia, Muluneh *et al.* (2017) showed that annual rainfall improved by 37.9 mm/decade whereas yearly precipitation on the highlands reduced by 29.8 mm/decade due to the decline of forest area from 44% in 1973 to less than 15% in 2009.

2.7. Climate change

The scientific community first started the concept of climate change when natural changes were observed in the 19th century (Luterbacher & Sprinz, 2001). However, the issue of climate change was boldly raised in the 1980s (Luterbacher & Sprinz, 2001) when the Intergovernmental Panel on Climate Change (IPCC) was established in 1988 to assess and report information on every single aspect (scientific, technical and socio-economic) of climate change including its possible impacts and alternatives for adaptation and mitigation (IPCC, 2014). Following the establishment of the IPCC, the UNFCCC was adopted in 1992 as an international environmental treaty aimed at reducing global warming by sinking greenhouse gas concentrations in the atmosphere (UN, 1992). Since then, the IPCC has been providing timely information by issuing a series of assessment reports regarding different aspects of climate change to improve on the understanding of the world with regards to the scientific and technical aspects of climate change.

Climate change can be defined in various ways. However, for this study, the definition of UNFCCC and IPCC were used. According to the UNFCCC's comprehensive definition, climate change is the variation in climate due to modification in the atmosphere's composition in addition to the natural climate variability observed within a short period of time which is ascribed directly or indirectly to human action (UN, 1992). On the other hand, the IPCC which exhorts government's on climate change describes climate change as any variation in the atmosphere's composition due to human activity or natural variability (Houghton, 1996).

Scientists have pointed out that the rise in greenhouse gas concentration in the atmosphere is the key driver of climate change. Anthropogenic factors such as the consumption of petroleum derivatives, deforestation and forest debasement have also caused a substantial rise in the atmospheric concentration of CO_2 (Sundquist *et al.*, 2008; US Global Change Research Program, 2009). Within the course of 250 years, the atmospheric concentration of CO_2 has increased from 280 to more than 401 parts per million instigating quantifiable global warming (IPCC, 2013). Likely antagonistic effects of climate change comprise sea-level rise, floods, repeated occurrences of wildfire, droughts, tropical storms, changes in rainfall pattern, runoff and disruption of coastal ecosystem and others (IPCC, 2014; Sundquist *et al.*, 2008).

However, there are individuals who insist that natural forces such as solar changes and changes in the Earth's orbit are major causes of climate change (Singer, 2008). Furthermore, they believe that computer models showing anthropogenic-based climate change are wrong or that the data themselves are wrong. They claim that there is no scientific evidence that demonstrates that human beings are the principal cause of the current change in climate and further criticize the scientific information given by the IPCC. Their conclusion is that natural causes are responsible for the current global warming being experienced (Singer, 2008).

2.8. Linkages between climate change and forests

Climate change influences almost every segment of the world's economy. It is unpredictably interlaced with other major ecological threats, such as population growth, desertification and land degradation, air and water contamination, loss of biodiversity, and deforestation. Climate change has an impact on the rates of photosynthesis and respiration which eventually affect forests (Kirschbaum, 2004; Law *et al.*, 2002). It also affects forest coverage by increasing the frequency of storms and wildfires, herbivory, and species migration (Gunderson, 2000). The change in global climate lessens plant species' biological resistance as well as forest biophysical processes which ultimately change forest ecosystems (Kellomäki *et al.*, 2008; Malhi *et al.*, 2008; Olesen *et al.*, 2007).

On the other hand, forests serve as home to and bolster the livelihood of a large number of poor and vulnerable individuals. They also provide imperative ecosystem services, from provisioning service to regulatory service. Forests are crucial in climate change adaptation as they can help prevent soil erosion and protect the coastal ecosystems against storm surges and hurricanes (Graham, 2011). The UNFCCC has paid special attention to the role forests play in dealing with climate change (UN, 1992). Forests are the most essential earthbound storehouses of carbon and play a crucial role in regulating our climate (FAO, 2010a). By 2030, estimates show that global forests will sequester nearly 13,800 Mt Co_2 per year (IPCC, 2007a). However, this figure is uncertain considering the variation in forest carbon sinks across regions and the high incertitude regarding the dynamics of these forest carbons in the future (Stinson *et al.*, 2011).

The dry Afromontane forests are important ecosystems for mitigating the effects of climate change (Kedir *et al.*, 2018; Munishi & Shear, 2004; Solomon *et al.*, 2017). However, they are heavily threatened by the adverse effects of climate change. In northern Ethiopia, climate-induced tree dieback affects dry Afromontane forest structure (Aynekulu *et al.*, 2011). In Kilimanjaro forest, the composition of species and forest structure have been altered as a result of fires caused by climate change (Hemp, 2009). Though climate change is proven to affect forests, empirical evidence detailing the magnitude of the direct impact of climate change in the dry Afromontane forests is lacking.

2.9. Forest cover change and ecosystem service values

Changes in the type of ecosystem prevalent in an environment have an effect on ecosystem services. Provision of services in a given ecosystem is determined by the type of ecosystem found in that particular area (Costanza *et al.*, 2014; de Groot *et al.*, 2012). Ecosystem dynamics, structure and functions across the world depend on land cover types (Wu & Hobbs, 2002). Consequently, ecological processes and services might be affected remarkably by the change in environmentally significant land cover types.

The ever-increasing change in land cover types, for instance, from forest to cultivated land or forest to grassland, cause a decline in supply of ecosystem goods and services to communities, alters functional processes and reduces biodiversity (Balvanera *et al.*, 2006; de Groot *et al.*, 2002; McIntyre & Lavorel, 2007). Various studies have indicated that the change in land cover, specifically from natural ecosystem to artificial land use has affected various ecosystem services such as nutrient cycling, climate regulation, genetic resources, soil fertility, erosion control, recreation and water supply (Kreuter *et al.*, 2001; Li *et al.*, 2007; Schröter *et al.*, 2005; Wang *et al.*, 2006; Zhao *et al.*, 2004).

Ecosystem service value can be affected by the change in land cover type as some ecosystem services increase while others decrease (Estoque & Murayama, 2013; Kindu *et al.*, 2016). For instance, between 1990 and 2000, an average annual decline of US$ 196.37 million ecosystem service value was recorded due to the loss of wetland in Chongming Island, China (Zhao *et al.*, 2004). Similarly, Li *et al.* (2010a) found that a decrease in wetlands and high-cover grassland in Zoige Plateau, China resulted in 5% loss of ecosystem service value between 1975 and 2005. In the Munessa-Shashemene landscape of the Ethiopian Highlands, Kindu *et al.* (2016) found that a reduction of US$ 19.3 million of ecosystem services between 1973 and 2012. In their study, they found that the ecosystem service value from croplands increased while value from forests and grasslands substantially decreased. Camacho-Valdez *et al.* (2014) noted an increase in the total ecosystem service value from 2000 to 2010, and attributed it primarily to the increase in coastal lagoon areas within the southern coast of Sinaloa State, northwest of Mexico. Wang *et al.* (2014) revealed a 22.7% increment in the overall ecosystem service value of the reserves in Ningxia between 2000 and 2010. In light of these, generalizing the outcomes of researchers to different areas may prompt wrong conclusions.

2.10. Linkage between carbon sequestration and forests

Carbon sequestration is a process of storing carbon released from different sources in different forms and preventing them from getting into the atmosphere. It is also the process of removing atmospheric carbon dioxide using different ways and storing it (Reichle *et al.*, 1999). Carbon can be stored in living biomass, litter, dead wood and soil, even though their storage capacity varies among the pools (Penman *et al.*, 2003). Out of the total above ground carbon in terrestrial ecosystems, forests hold more than 75% carbon, making it a main terrestrial reserve (Houghton, 2007; Watson, 2000).

Vegetation stores carbon through the process of photosynthesis where plants utilize daylight to convert supplements into sugars and carbohydrates. Although the extent of carbon varies, every plant stores carbon in their stem, branches, roots and leaves. Plant species type, plant age, growth pattern and density often affect the distribution of carbon among different plant parts (Gorte, 2009). The rate of photosynthesis affects carbon stock. This is because as the rate of photosynthesis increases, carbon stored in plant biomass also increases (Gorte, 2009).

Apart from the carbon stored in vegetation, carbon can also be stored in the soil. Organic carbon in the soil can be obtained from the decay of vegetation and microorganisms (Gorte, 2009). Soil carbon can increase when root biomass increases. The global soil carbon is estimated at 2500 Gt (Bhatti *et al.*, 2005). Bhatti *et al.*

(2005) have suggested that the carbon stored in soil pool is 4.5 times higher than the carbon stored in the biotic pool (560 Gt) and 3.3 times higher than the carbon in the atmosphere (760 Gt).

Carbon is also stored in fallen woody debris, standing dead trees and decaying wood. The amount of carbon in dead wood is affected by the time of disturbance, input, mortality rate, decay rate and management of dead wood (Spies *et al.*, 1988). In mature forests, a carbon equivalent of 10-20% of the above ground carbon is stored in dead wood (Achard *et al.*, 2002; Harmon & Sexton, 1996; Houghton *et al.*, 2001).

Litter carbon pool is also considered one of the main carbon pools that contribute significantly to the global carbon budget. Five per cent of the world's forest ecosystem's carbon stocks are stored in a litter (Domke *et al.*, 2016). Litter carbon can be affected by the annual amount of litterfall and the annual rate of decomposition (Fierer *et al.*, 2005; Prescott, 2010). Forest management practices such as bush burning, logging, and site clearing radically change litter properties (Binkley & Fisher, 2012).

In terrestrial ecosystems, forests are the most productive among their biotic components. These productive characteristics of forests make them outstanding in the process of mitigating climate change (Metz *et al.*, 2007). Forests represent an important carbon store, estimated globally to contain 638 Gt of carbon, of which 283 Gt carbon is present in above and below ground biomass alone (FAO, 2005a). The sequestration potential of forests varies greatly with forest type, site factors, and age (Wei *et al.*, 2013). Hence, for specific forests, there is a need to use specific approaches in carbon stock assessment.

In Ethiopia, woodlands (45.7%) and shrublands (34.4%) sequestered the largest amount of carbon (WBISPP, 2005). However, these forest ecosystems are largely ignored in forestry-related negotiations, including carbon discussions regardless of their great potential in sequestering carbon. In Ethiopia, there are different values of carbon reported by different researchers. For example, in 2005, a total of 2.5 billion tons of carbon was reported by Nune *et al.* (2010). On the other hand, Houghton (1999) and Gibbs and Brown (2007) reported a carbon stocks estimate of 153 million tons and 867 million tons respectively. Similarly, 101 tons of carbon ha^{-1} was reported for high forests in Ethiopia (Brown, 1997). Contrary to all these, Temam (2010) in his case study recorded around 200 tons of carbon ha^{-1} for high forests in the Bale mountains. The variation in the estimates of carbon might be due to the differences in agroecology, the approaches and tools used, as well as the doubts related with the approaches applied.

2.11. Linkages between forest cover change and carbon stock

Forests are among the key components of the carbon cycle in the terrestrial ecosystem (McMahon *et al.*, 2010). The change in forest cover affects the carbon cycle globally. Land cover change especially deforestation is the major contributor to the human-induced greenhouse gas emissions (IPCC, 2007b). About 20% of anthropogenic emissions come from the transformation of forests into other domains of land use (FAO, 2005a; Stern *et al.*, 2006). However, the emission from deforestation is not similar across regions. For example, in Africa, deforestation is the major contributor to greenhouse gas emissions contributing about 70% of the total emissions (Holly *et al.*, 2007).

Ecosystem maturity and degradation through natural processes can cause changes in biomass and soil carbon stocks (Ostle *et al.*, 2009) or through land cover change by human activities (Noble *et al.*, 2000). Forest land transformation into cultivated land, for instance, invariably causes emissions of an ample amount of CO_2 into the atmosphere and causes a prompt decrease of soil organic carbon stocks (Salinger, 2007).

Various studies have estimated a 20-50% soil carbon loss due to the cultivation of natural soil (Gregorich *et al.*, 2005; Guo & Gifford, 2002; Murty *et al.*, 2002; Post & Kwon, 2000). Similarly, Haghdoost *et al.* (2013) have also demonstrated that a change in land use had a substantial effect on carbon stock in Iran. A study by Kashaigili *et al.* (2013) in Pugu and Kazimzumbwi forest reserves, Tanzania also highlighted the effect forest cover change has on above ground carbon stock. In another study conducted in Yok Don National Park in central highlands of Vietnam, a 4.3 Megaton decline in total biomass between 2004 and 2010 was reported (Luong *et al.*, 2015). Between 2005 and 2010, an annual reduction of 0.5 Gt of biomass carbon stock was recorded and this is explained by the loss of forest area globally (Dalal-Clayton & Sadler, 2014).

The review concludes that forest cover change and carbon stocks are highly correlated subjects. As forest cover increases carbon stock increases or the other way round. Sustainable forest management, afforestation and reforestation are recommended practices to increase carbon stock in forests (Dalal-Clayton & Sadler, 2014).

2.12. Marketing forest carbon

Since 1992, the carbon market has received greater attention in the global market for ecosystem service. In 1992, 192 nations worldwide signed the UNFCCC agreement

(Jackson, 2011), the first-ever treaty went for settling greenhouse gas emissions and circumventing uncontrolled climate change. It incorporated an arrangement of underlying standards, including 'polluter pays' and 'common but differentiated responsibility', which perceived matters of verifiable obligation and equity in solving climate change (Reddy, 2011). The Kyoto Protocol, a universal and lawfully official consent to diminish greenhouse gas outflows globally was embraced in 1997 and brought into force in 2005.

The premise of Kyoto is trading of carbon for dealing with greenhouse gas credits between nations. It contains various policy instruments to assist industrialised countries to accomplish their emissions goals. It includes the Clean Development Mechanism (CDM), International Emissions Trading (IET), and Joint Implementation (JI) (Reddy, 2011).

As indicated by the World Bank carbon fund unit, carbon trading has become a huge industry worth over US$ 144 billion as at 2009 (Alexandre & Philippe, 2010). A total of $6.5 billion CDM based projects were implemented in 2008, even though this amount reduced to US$ 2.7 billion in 2009 (Reddy, 2011). This reduction was caused by the decline in economic situations, together with the fears about the post-2010 game plan when the Kyoto Protocol was terminated. Out of the total 7,645 CDM projects registered since 2015, 2,587 projects with 1.6 billion of certified emission reduction are already in operation, of which China, India and Brazil hosted more than 49%, 20.6% and 4.4% of total projects respectively (Alloisio *et al.*, 2015). Africa owned about 2.8% of valid CDM projects registered with the UN's climate change secretariat (UNEP DTU, 2015; World Bank, 2010).

The enthusiasm for carbon sequestration and marketing as an instrument for both poverty alleviation and natural security in emerging nations has expanded extensively in the most recent periods (Priyadarshan, 2011). The idea hinges on improving ecosystem services such as carbon sequestration and reducing poverty by paying landowners who own plots being used for carbon sequestration projects (Smith & Scherr, 2002). It is anticipated that productivity, income and food security benefits could increase for producers under CDM (Pagiola *et al.*, 2005; Tschakert, 2004; Tschakert *et al.*, 2004; Woomer *et al.*, 2004). CDM projects are also anticipated to boost the monetary value of the environmental benefits (biodiversity conservation and carbon sequestration) delivered by different land use types (Kremen *et al.*, 2000; Portela & Rademacher, 2001). In spite of the introduction of several projects on the carbon market, a lack of clear procedures has halted their development.

2.13. Reducing emissions from deforestation and forest degradation

Reducing emission from deforestation and forest degradation (REDD+) is very crucial in lowering the effect of climate change. The objective of REDD+ is to promote that increase forest cover by reducing deforestation and forest degradation at the local level, thereby reducing the amount of CO_2 that is released into the air (Baldo-Soriano *et al.*, 2012). The focus of the REDD+ idea is on providing financial inducements to support unindustrialized nations who have volunteered to decrease deforestation rates and related carbon releases below the baseline (Holly *et al.*, 2007). Countries that ensure reductions in emissions might have the opportunity to supply carbon stock to the global carbon market or somewhere else. The reduction in emissions perhaps, instantaneously helps fight climate change, safeguard plant and animal diversity and eventually improves ecosystem services (Holly *et al.*, 2007).

Since the protection of forests infers inescapable incomes, emission reduction from deforestation has its costs (Butler *et al.*, 2009). The REDD+ mechanism of incentives to decrease deforestation and forest degradation emissions looks to advance economic development and growth without putting an end to valuable natural resources (Streck, 2012). Under the REDD+ mechanism, countries also agree to improve their forest cover by reducing deforestation and forest degradation (Streck, 2012). However, REDD+ has several challenges on the ground. Mobilizing adequate universal and national funds to take care of the expenses of REDD+ approaches and measures, matching the necessities and requirements of stakeholders and different partners in low income nations with those of contributors or financiers in REDD+, and solidifying the institutions expected to execute policies and oversee REDD+ funds are some of the challenges (Streck, 2012).

Additionally, REDD+ has faced challenges in the distribution of benefits (Costenbader, 2011). Fair distribution of incentives from the REDD+ is very important for sustainable emission reduction by avoiding leakages which can negatively affect REDD+ projects (Peskett, 2011).

While the importance of emission reduction from deforestation and its related activities is recognized by many developing countries, clear incentives and other economic policies are lacking (Santilli *et al.*, 2005).

In Ethiopia, REDD+ has established under a program that supports land rehabilitation through afforestation and reforestation (Bekele *et al.*, 2015). Currently, the REDD+ policy appears to have been integrated with the Climate-Resilient Green Economy (CRGE) strategy, which cooperates with the growth and transformation plan (GTP) of Ethiopia. The objective of the CRGE is to provide pushy cross-sectorial strategies aimed at improving the economic development of the country and developing the country into a middle income one by 2025 without adding to the existing levels of greenhouse gas emissions. Notably, REDD+ is among the four noteworthy initiatives of CRGE chosen for quick execution (FDRE, 2011).

The REDD+ secretariat has been housed in the Environment and Forest Ministry (EFM) facility since 2013 (Bekele *et al.*, 2015). Ethiopia is making decent progress in establishing REDD+ based projects and integrating them with participatory forest management projects, including the Yayu & Gedo forests, Bale eco-region and Baro-Akobo forest located in the southwest of Ethiopia (Lemenih & Woldemariam, 2010).

Currently, activities of the REDD+ secretariat focus on REDD+'s Readiness phase II aimed at awareness creation regarding REDD+, organizational and individual capacity building at the national and regional levels in order to back REDD+, detailed studies of REDD+ issues (e.g. studies on identifying causes of deforestation, safeguards, institutional set-up, reference level, consultation and participation, capacity building and communication strategies), piloting REDD+ implementation strategies on forest/landscape, establishing regional REDD+ coordinators in consultation with regions, and working on developing a draft for REDD+'s national strategy (Bekele *et al.*, 2015).

2.14. Methodological approaches and issues

2.14.1. Geographic Information System (GIS) and remote sensing

GIS and remote sensing are currently providing new instruments for cutting-edge environment management practices. The use of remote sensing and GIS techniques allows to effectively investigate the extent of vegetation change (Lunetta, 1998).

Vegetation change detection is valuable in the assessment of land cover changes, rate of deforestation and wildlife extinction.

The application of remote sensing and GIS on forest cover change analysis is in recent times receiving wider acceptance among experts (Hosonuma *et al.*, 2012; Luong *et al.*, 2015). Synoptic coverage, global reach and readability, consistency in data, precision and maximum accuracy in data provision makes remotely sensed data preferable as compared to other methods (Hosonuma *et al.*, 2012).

The science of remote sensing and its application in societies have provided progressively solid and consistent information (Olofsson *et al.*, 2014). Remote sensing advancement can give objective, reasonable and financially savvy answers for improving REDD+ monitoring methods (Hosonuma *et al.*, 2012). The elucidation of Landsat images was observed to be the most advantageous, rapid appraisal system for surveying and assessing dynamic ecological changes and patterns (Larsson & Strömquist, 1991). Image interpretation and analysis provide accurate information on forest cover, forest type, forest condition, biomass and land potential for afforestation programs.

The utilization of satellite remote sensing in forest cover change identification has grown quickly in relation to the digital image analysis of earth resources satellite information (Yuan *et al.*, 2005). Considering the rapid changes in land cover universally, remote sensing technology is a fundamental instrument particularly useful in monitoring tropical forest conditions. The remote and unreachable nature of many tropical forest regions confines the practicality of ground-based data collection and observation for broad regions (Hayes & Sader, 2001). Currently, all natural resource assessment-related activities, especially land cover change analysis, are being relayed on remotely sensed data. A range of methods are being employed to identify forest cover changes from multi-temporal images (Hayes & Sader, 2001).

In view of this review, it can be concluded that satellite remote sensing is very useful in deriving quantitative data on forest cover change and presenting the results in an understandable form. Besides, digital image analysis applied to forest cover monitoring has the advantage of being fast and able to handle large quantities of data hence the increase in its use.

2.14.2. Forest biomass and carbon stock assessment

Biomass incorporates both above ground and below ground living masses of trees, shrubs, vines, and the dead mass of fine and coarse litter (Liang *et al.*, 2012). The accurate evaluation of a forest's biomass is critical for achieving accurate forest carbon stocks since one is obtained from the other. Assessment of biomass and carbon stock is an important tool in finding a sustainable way of utilizing forests and understanding the contribution of forest to the global carbon cycle. Furthermore, to adequately realize mitigating approaches and explore the REDD+ program of the UNFCCC (Chaturvedi *et al.*, 2011), nations require highly validated evaluations of forest carbon stocks. Consequently, knowing the methods of forest biomass and carbon stock assessment with their advantages and drawbacks is very essential.

Forest biomass and carbon stock can be measured using ground data from field inventory and remotely sensed data (Lu, 2006; Ravindranath & Ostwald, 2008). There are two forest biomass field measurement techniques namely destructive and non-destructive methods. The destructive method computes biomass directly by felling trees and quantifying the actual weight of its components (stem, leave, branch and root) (Kangas & Maltamo, 2006). The destructive method of biomass quantification is known to achieve an accurate and precise quantification of forest carbon stocks (Henry *et al.*, 2011). Though this approach is comparatively precise for a small area, it is time-consuming, costly, damaging, and not applicable to large areas of land. In addition, a destructive method is not appropriate for depleted forests containing endangered species (Montès *et al.*, 2000). However, this method is very important in developing allometric equations used for estimating biomass in large areas (Návar, 2009; Segura & Kanninen, 2005). Many scholars have used destructive methods to develop allometric equations based on site and species specificity (Birhane *et al.*, 2017b; Giday *et al.*, 2013; Hasen-Yusuf *et al.*, 2013; Solomon *et al.*, 2017; Tesfaye *et al.*, 2016b; Ubuy *et al.*, 2014).

Biomass can be measured by employing non-destructive methods using ground data. This method estimates tree biomass indirectly, without cutting the tree. It basically requires the measurement of a tree's vegetation parameters (dendrometric variables) such as the diameter at breast height, tree height, crown diameter, tree volume and wood density (Ravindranath & Ostwald, 2008) and computes biomass using biomass equations (Hughes *et al.*, 1999). Non-destructive methods are the most frequently used techniques for assessing forest biomass. In the meantime, when picking allometric equations, the applicability or the appropriateness of the method should be considered. Allometric equations are not recommended for use beyond their range of validity (Chave *et al.*, 2005). This method is considered to be time efficient (Peltier *et al.*, 2007).

The allometric relationship is often the preferred technique for evaluating woodland biomass as it is non-damaging, relatively less tedious and more affordable. However, the degree of accuracy is lower as compared to the destructive methods. In spite of the fact that field inventory is the most accurate method, it is expensive and time-consuming. Besides, it is relevant for only small-scale assessment. In this manner, remotely sensed data is relied upon to give an answer to the previously stated challenges.

The use of remote sensing data for biomass estimation is an evolving process even though it is being progressively used for forest biomass assessment. In the near future, biomass assessment using satellite data will become more available (Gibbs *et al.*, 2007). In any case, biomass cannot be estimated directly using remote sensing data (Vashum & Jayakumar, 2012). It just measures tree dendrometric variables such as tree height, crown size, forest density, forest type, forest volume, leaf area index, and so on, which are associated with biomass. It is important to note that remote sensing data needs to be supported by field data in order to estimate biomass. The field estimations are normally used to establish biomass models and confirm the outcomes of the remotely sensed data (Vashum & Jayakumar, 2012).

Forest biomass has been estimated using a combination of remote sensing and ground-based data in many studies (Boudreau *et al.*, 2008; Drake *et al.*, 2003; García *et al.*, 2010; Gleason & Im, 2012; Lefsky *et al.*, 2002; Lu, 2005; Zhao *et al.*, 2009; Zheng *et al.*, 2004). Boudreau *et al.* (2008) conducted a study to assess above ground dry biomass in the forested land of Québec, a province in eastern Canada. In their study, they used a combination of data from remote sensors and ground data collected using tools such as the Landsat ETM+ land cover map, airborne and space-borne light detection and ranging, Shuttle Radar Topographic Mission digital elevation model, ground inventory plots, and vegetation zone maps. Drake *et al.* (2003) examined the relationship between light detection, ranging metrics and forest structural characteristics which included above ground biomass. Two areas in central America were considered in the study using canopy height from airborne light detection, ranging data and basal area, mean stem diameter and above ground biomass from ongoing inventory data. They found that a strong correlation existed between light detection and ranging metrics and vegetation parameters such as mean stem diameter, basal area, and above ground biomass.

Some of the remote sensing data used to estimate biomass and carbon have very high-resolution aerial imagery, optical remote sensing data, light detection and ranging microwave or radar data (Gibbs *et al.*, 2007). In the coming decades, remote sensing-based carbon monitoring, reporting, and verification will be increasingly used alongside a validation of ground data (Asner *et al.*, 2010; Baccini *et al.*, 2012; Le Toan *et al.*, 2011; Saatchi *et al.*, 2011).

The benefit of remote sensing is that it provides data which would have usually been inaccessible or difficult to obtain. Additionally, remote sensing can be used for analyzing large area even at the global level. It is also one of the most cost-effective methods for getting information over a substantial zone.

2.14.3. Soil organic carbon estimation

Soil organic carbon is characterized as the portion of non-living natural compounds in the top one meter of soil, (humus is an example) which is imperative to soil quality and plant nutrition and is replenished by the decay of plant material (FAO, 2017). Soil organic carbon as explained by Paustian *et al.* (2006) involves dead and living fine roots in the soil as well as organic carbon in mineral soils to a particular depth. Considering the increasing enthusiasm for the carbon cycle around the world, there is the need to measure soil organic carbon using various methods. Dry combustion with automated analyzers, a wet chemical oxidation technique also known as the Walkley and Black technique and loss-on-ignition are methods generally used to analyze soil organic carbon.

The dry combustion technique quantifies CO_2 produced by the entire breakdown of organic material at a given (recommended) temperature of mostly 900 °C or above (Fernandes *et al.*, 2015). Currently, this method is regarded as reliable in comparison with other methods (Lettens *et al.*, 2007). Nonetheless, this method is costly, which restricts its applicability (Fernandes *et al.*, 2015). In addition, it applies acid pretreatment to remove carbonate in calcareous soils which may reduce the organic matter content of the soils (Byers *et al.*, 1978). Various studies have used dry combustion methods to measure soil organic carbon (Gelaw *et al.*, 2014; Li *et al.*, 2010b; Sato *et al.*, 2014; Wright & Bailey, 2001)

Alternatively, the Walkley and Black technique is considered one of the standard approaches to investigating soil organic carbon (Nelson & Sommers, 1996). Walkley and Black's method is used to determine the dry mass of organic matter in a soil sample. In this reaction, carbon is oxidized by the dichromate ion. This strategy is more affordable than dry ignition. However, it may prompt variable recuperation of soil organic carbon (Conyers *et al.*, 2011; De Vos *et al.*, 2007), and pose a danger to chromium (Cr) because of the utilization of dichromate. The Walkley and Black technique is applicable when the soil carbonate content of the soil is high. Hence, when conducting a soil organic carbon analysis in carbon-poor areas, it is recommended to use an adjustment factor of 1.30 (Gessesse & Khamzina, 2018; Gillman *et al.*, 1986; Nelson & Sommers, 1982). Several studies have been conducted to measure the soil organic carbon using Walkley and Black's oxidation method

(Andriamananjara *et al.*, 2016; Girmay & Singh, 2012; Mathew *et al.*, 2016; Sharma *et al.*, 2014; Tesfaye *et al.*, 2016b).

On the other hand, loss-on-ignition is a method which measures weight loss by combusting soil organic matter at high temperature (Goldin, 1987; Wang *et al.*, 2011). The loss-on-ignition technique is basic, more affordable relative than the dry combustion technique and chemical methods, as it requires simple materials (such as oven dry, muffle furnace, and scale) which are often readily available in many research centres (Konen *et al.*, 2002). However, this technique is mostly efficient when applied on fertile soils though it gives precise assessments of carbonate in sediments (Heiri *et al.*, 2001; Wang *et al.*, 2011). Several studies have been carried out using the loss-on-ignition technique to measure soil organic carbon (Ball, 1964; Ben-Dor & Banin, 1989; Craft *et al.*, 1991; Howard & Howard, 1990).

2.14.4. Valuation of ecosystem services

Ecosystems service valuation endeavours to allocate economic value to the entire range of ecosystem services, including those that have no known market values. The main objective of ecosystem services valuation is to comprehend the various benefits provided by ecosystems (Guo *et al.*, 2001). Having better knowledge of the ecosystem services' contribution highlights natural resources as critical components of inclusive wealth, prosperity, and sustainability (Costanza *et al.*, 2014). Supporting and improving human prosperity requires a good balance of the majority of our assets. Assessing the relative value of the role of ecosystems in the wellbeing of humankind has been an important tool for promoting the sustainable use of natural resources.

Since 1997 various investigations have been carried out focusing on the assessment of ecosystem services' value. The state of knowledge on ecosystem services ranges from studies on the value of ecosystem services at the global scale (Costanza *et al.*, 1997; Costanza *et al.*, 2014; de Groot *et al.*, 2012) to changes in ecosystem services' provision and value in response to land use changes at the landscape scale (Hu *et al.*, 2008; Kindu *et al.*, 2016). Costanza *et al.* (1997) estimated ecosystem service values for 16 biomes and provided an aggregate global value expressed in monetary units. de Groot *et al.* (2012) estimated the value of ecosystem services supplied by 10 main biomes using published materials of ecosystem service values across the world.

Various methods have been established to evaluate ecosystem services. Some of the approaches are market values, change in productivity, travel cost, hedonic price, avoided damage costs, replacement/substitute costs, costs of human capital, contingent valuation, choice modelling, and benefits transfer (Costanza *et al.*, 1997; Kroeger & Casey, 2007; Loomis, 2005; Loomis *et al.*, 2000; Perez-Verdin *et al.*, 2016). Many types of research have been done based on the above methods. For example, Costanza *et al.* (1997) valued the world's ecosystems based on the benefits transfer method. Kroeger and Casey (2007) assessed ecosystem services of agricultural lands with a market-based approach. Loomis *et al.* (2000) used the contingent valuation method to assess the total ecosystem services' value in an impaired river basin. Wilson and Carpenter (1999) reviewed the economic value of freshwater ecosystem services in the United States from 1971 to 1997 based on 30 published papers done using the hedonic pricing methods, travel cost methods and contingent valuation methods. Currently, many researchers have been adopting the value given by Costanza *et al.* (1997) to evaluate ecosystem service values. For example, Kindu *et al.* (2016), Kreuter *et al.* (2001) and Li *et al.* (2007) evaluated the impact of land cover changes on ecosystem services value in the Munessa-Shashemene landscape of the Ethiopian highlands, the San Antonio area of Texas and

Pingbian County in China, respectively. Similarly, many types of research have used the Economics of Ecosystems and Biodiversity (TEEB) database to evaluate the ecosystem service values of different ecosystems (Costanza *et al.*, 2014; de Groot *et al.*, 2012; Kindu *et al.*, 2016). This database has 1,310 data points collected from 267 different publications (Van der Ploeg & De Groot, 2010).

2.15. Conceptual framework

Human being and the ecosystem interacts dynamically (MEA, 2005a). Humans are among the key drivers of ecosystem change. This notwithstanding, there are natural forces that also influence ecosystems (Figure 1). The changes in ecosystems in turn cause changes in human wellbeing.

This research focuses on the linkages between land cover change and ecosystem services using carbon sequestration as a proxy. Ecosystem conditions affect the flow of ecosystem services (de Groot *et al.*, 2002; Styers *et al.*, 2010). For instance, land cover dynamics can change the ecosystem service values by increasing the supply of some services while causing a decrease in others (Hu *et al.*, 2008; Kreuter *et al.*, 2001; Polasky *et al.*, 2011). For example, changing land from forest to cultivated land can increase food supply. However, it can negatively affect other ecosystem services such as water supply, carbon sequestration, raw material production and erosion control. The change in land cover can be the result of various factors including cultivated land expansion, logging, species introduction, climate change, governance, demography, infrastructure and culture.

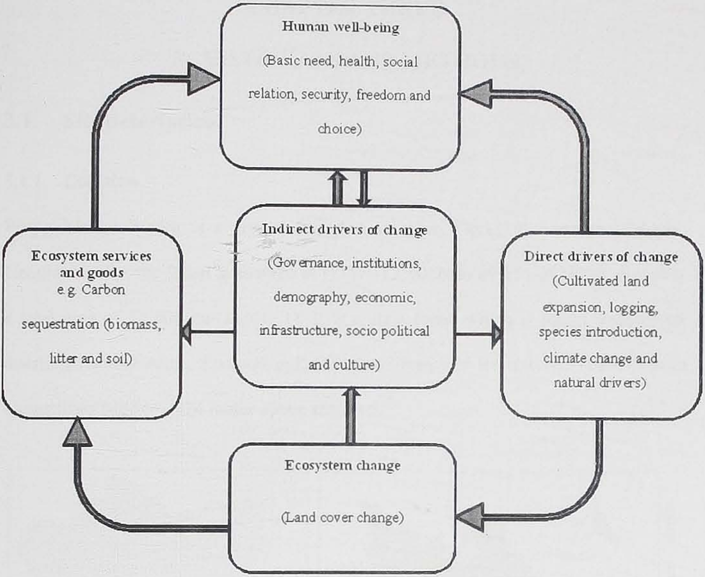

Figure 1: Conceptual framework modified from (MEA, 2005c)

The ecosystem can be directly affected by drivers such as cultivated land expansion and logging aimed at increasing economic gain (right corner of figure 1) or indirectly influenced by drivers such as demography, institutions, and economic activities (example, market) (center of figure 1). The resulting changes in the ecosystem (bottom of figure 1) cause some changes in ecosystem services such as carbon sequestration (left corner of figure 1) and thus, distract human well-being (top of figure 1).

CHAPTER THREE

3. MATERIALS AND METHODS

3.1. Site description

3.1.1. Location

Wujig Mahgo Waren is a natural forest in southern Tigray in northern Ethiopia. Geographically, the forest is situated at 12°47′–13°02′ N to 39°26′–39°39′ E, covering a total area of 17,000 ha (Figure 1). It is a state forest which is found within four districts, namely Alaje, Endamokeni, Hintalowejerat, and Rayaazebo. The elevation varies from 1404 to 3924 meter above sea level.

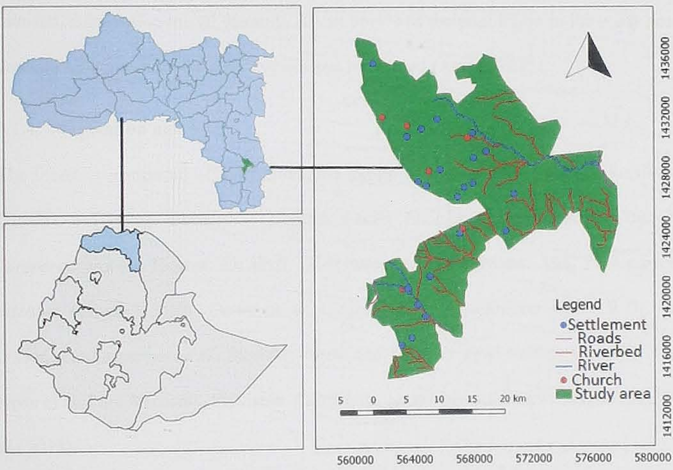

Figure 2: Map of Wujig Mahgo Waren forest in Tigray, northern Ethiopia.

3.1.2. Climate

The mean annual and monthly temperature of the study area ranges between 8 and 25 °C. The area receives mean annual rainfall of 837 mm (Amanuel *et al.*, 2015). The study area is characterized by semiarid climate. The area experiences two rainy seasons with the main season coming between June and September follows the short rainy season which lasts from February to May.

3.1.3. Population

Based on the 2013 population projection of Ethiopia, southern Tigray zone had a total population of 1,184,208 in 2017 of which 49% were men and 51% were women; 18% were urban inhabitants (CSA, 2013). The community in the study area practice rain-fed, subsistence-mixed farming. About 86.4% of the population in the study area depends on agriculture for employment and livelihood (Abrha, 2015).

3.1.4. Vegetation and soils

The forest is composed of indigenous and exotic species, mainly *Acacia abyssinica* [Hochst. ex] Benth, *Eucalyptus globulus* Labill, *Eucalyptus camaldulensis* Dehnh, *Juniperus procera* Hochst. Ex Endl, *Olea europaea ssp. africana Mill, Podocarpus falcatus* (Thunb.) Mirb, *Dodonea angustifolia* L.F., *Combretum molle* R.Br. ex G.Don, *Cadia purpurea* (G.Piccioli) Aiton and *Opuntia ficus-indica* (L.) Mill. The types of soil are Vertisols, Fluvisols, Cambisols, Leptosols and Regosols (Amanuel *et al.*, 2015).

3.2. Acquisition of satellite images

Cloud-free Landsat satellite images captured in the dry season for the years 1985, 2000, and 2016 were acquired from the U.S. Geological Survey's Earth Resources Observation and Science (USGS) (http://earthexplorer.usgs.gov/). The years of satellite images were based on major changes in political regimes in Ethiopia. The details of satellite images are presented in Table 1.

Table 1: Satellite data used in the study

No.	Satellite	Sensor	Date of acquisition	Pixel resolution (m)	No of bands used
1	Landsat	TM	1985	30	6
2	Landsat	ETM+	2000	30	6
3	Landsat	OLI/TRIS	2016	30	6

3.3. Ground truthing data

The Global Positioning System (Garmin GPS) was used to collect ground control points as was used by Hailemariam *et al.* (2016). More than 30 ground control points per land cover class were purposively assigned. The method for accuracy assessment was adopted from MacLean and Congalton (2012). Forty points per class were collected for accuracy assessment.

Ground control points were collected by interaction with community elders for classification and accuracy assessment of 1985 and 2000 images. Majority agreement was used to validate accuracy.

3.4. Land cover classification, accuracy assessment and change detection

The Earth Resources Data Analysis System (ERDAS) Imagine software (2014) was used for satellite image analysis and accuracy assessment after image data preprocessing. Both unsupervised and supervised image classification methods were employed. Unsupervised classification using the Iterative Self-Organizing Data Analysis Technique (ISODATA algorithm) was first carried out to identify a potential representative of the overall land cover clusters of pixels available. Maximum likelihood supervised classification was performed by using training samples.

Consequently, the accuracy of the image classification was assessed by using both Kappa and accuracy statistics.

Matrix analysis was adopted to determine and detect land cover change (Lu *et al.*, 2004). Areas that were converted from one class to any of the other classes were computed and the change directions were also determined. The levels of change were measured in hectares and percentages. The estimation of the percentage of change for the different covers was computed based on the following formula;

$$\% \text{ cover change} = \frac{\text{Area}_{\text{i year x+1}} - \text{Area}_{\text{i year x}}}{\sum_{i=1}^{n} \text{Area}_{\text{i year x}}} \text{x } 100\%$$

Where,

$\text{Area}_{\text{year x}}$ = area of cover *i* at the first date,

$\text{Area}_{\text{year x+1}}$ = area of cover *i* at the second date,

$\sum_{i=1}^{n} Area_{i\ year\ x}$ = total cover area at the first date.

3.5. Household survey and focus group discussion

Household survey using semi-structured questionnaires (Appendix A) and focus group discussions were conducted to understand perception of forest cover change, impact of deforestation on livelihood, drivers of deforestation, existing remedies and possible mitigating measures.

A stratified two-stage sampling procedure was used to select kebeles (the smallest administrative unit of Ethiopia) and farming households. The study area was stratified into the following three agroecological zones, following the classifications of the Ministry of Agriculture (MoA, 2000);

- Sub-humid highlands (1500 to 2300 m),
- humid highlands (2300 to 3200 m), and
- Cold highlands (above 3200 m).

The two-stage sampling technique was applied to select the respondents for the household survey. In the first stage, a purposive sampling method was employed to identify representative kebeles from each of the agro-ecological zones based on their distance to the forest. Two communities namely Tsigea and Ebo from the sub-humid highlands, one community (Hizba Teklahymanot) from the humid highlands, and one community (Ayiba) from the cold highlands were selected. A total of 150 respondents were randomly selected from the selected Kebeles in the second stage. In addition, twelve group discussion sessions were held with farmers, foresters and agricultural extension officers. Each discussion session involved 10 to 12 individuals.

The survey questionnaires covered issues regarding demographic characteristics of households, drivers of forest cover change, perception of the local people on forest cover change, and existing remedies and possible mitigating measures. Related questions were also discussed during group discussions. The consent of the participants was obtained preceding the data collection activities.

3.6. Analysis of household survey and focus group discussion

Descriptive statistics of simple frequency analyses were used to describe socioeconomic characteristics of households. Differences in perceptions among respondents in different kebeles concerning the forest cover change and drivers of forest cover change were analyzed using Pearson's chi-square test. Drivers of forest cover change were ranked using ranking index method adapted from Musa *et al.* (2006). The index was computed as:

$$\text{Index} = \frac{\text{Rn} * \text{C1} + \text{Rn} - 1 * \text{C2} \ldots + \text{R1} * \text{Cn}}{\Sigma\, \text{Rn} * \text{C1} + \text{Rn} - 1 * \text{C2} \ldots + \text{R1} * \text{Cn}}$$

Where, Rn = 5, Rn-1 = 4, R1 = 1; Cn = the count of the 5th rank, and the count of the 1st rank = C1.

Data collected through focus group discussions and observations were analyzed qualitatively using content analysis.

3.7. Vegetation and soil sampling design

Ten parallel line transects were laid with a distance of 1 km between them. Randomly selected sample plots of 20 m × 20 m, representing the main plots, were demarcated for trees and shrub assessment, and five 1 m x 1 m subplots within the main plot designated for litter and soil sampling. Sample plots along the transects were set at 400 m intervals. There were a total of 88 sample plots.

To determine the number of main plots to be sampled, Pearson *et al.* (2005) equation was used as follows;

$$n = \frac{\left(\sum_{i=1}^{n} N_i * S_i\right)^2}{\frac{N^2 * E^2}{t^2} + \left(\sum_{i=1}^{n} N_i * S_i^2\right)}$$

Where:

n = number of sample plots

E = allowable error computed from mean carbon stock by the desired precision (that is, mean carbon stock x 0.1, for 10 per cent precision),

t = t statistic for the 95 per cent confidence level; *t* is usually set at 2,

N_i = Area of land cover type *i* divided by plot size,

S_i = standard deviation of land cover *i*.

3.7.1. Species inventory

An inventory to determine trees and shrubs of the plots was carried out. All trees and shrubs were identified in the field and listed. A botanist supported by the local people was engaged to confirm scientific names and local names of the plant species.

3.7.2. Vegetation parameters measurement

Height (H) and diameter at breast height (DBH) (1.3 m) of all trees and shrubs with DBH $\geq$ 2 cm were measured using a 5 m pole graduated with 10 cm markings and measuring tape, respectively from each main plot. Clinometer was used to measure tree height for trees taller than 5 m.

3.7.3. Litter sampling

Litter samples were collected from five 1 m^2 subplot within the main plot. A composite sample of 100 grams litter was taken to the laboratory for litter carbon analysis.

3.7.4. Soil sampling

Soil samples were collected from 5 subplots within the main plot at a depth of 30 cm using a core sampler. All samples were put in paper bags with proper labels. From each plot, a composite sample of 100-gram soil was submitted to analyze bulk density and soil organic carbon.

3.8. Carbon quantification

3.8.1. Biomass carbon stock assessment

Above ground biomass (AGB) was estimated using the equation of Chave *et al.* (2014) provided below.

$$AGB\ (Kg) = 0.0673 * (\rho DBH^2 H)^{0.976}$$

Where,

DBH is diameter at breast height,

H is total tree height and

ρ is wood specific gravity = 0.58 g cm^{-3} the arithmetic mean for tropical Africa.

Site and species-specific allometric equations were also used for above ground biomass computation (Appendix B).

Below ground biomass (BGB) was estimated using the following regression model given by Cairns *et al.* (1977):

$$BGB\ (Kg) = \exp(-1.0587 + 0.883 * \ln AGB)$$

Where,

AGB = above ground biomass

For the conversion of biomass to carbon stocks, the following formula given by Pearson *et al.* (2005).

$$Carbon\ (Kg) = 0.5 * biomass$$

3.8.2. Litter carbon estimation

To estimate litter carbon 100 grams of composite fresh weight of litter was collected from the five subplot sample and oven dried at 105 °C. Litter biomass was estimated using Pearson *et al.* (2005) equation provided below.

$$Dry\ \text{mass} = \left[\frac{\text{dry mass of composite sample}}{\text{fresh mass of composite sample}}\right] * \text{X}$$

Where, X is a total fresh mass of the whole sample.

To estimate litter carbon stock the following formula was used.

$$\text{Litter carbon (Mg ha}^{-1}\text{)} = \text{dry mass} * \%\ \text{carbon}$$

% carbon is the carbon fraction of IPCC with a default value of 0.37.

3.8.3. Soil carbon stock assessment and texture analysis

Soil organic carbon (SOC) was calculated using Pearson *et al.* (2005).

$$\text{Soil organic carbon} = \text{bulk density} * \text{depth} * \%\text{carbon}$$

Where,

$$\text{Bulk density (g cm}^{-3}\text{)} = \frac{\text{oven dry mass (g)}}{\text{volume (cm}^3\text{)}}$$

%Carbon = determined in the laboratory following Walkley and Black (1934) method.

Soil texture analysis was performed using the hydrometer method.

3.8.4. Total Carbon Stock

The total carbon stock (C_t) was computed by adding the carbon stock values of the individual carbon pools of the land cover type using the following formula.

$$C_t\ \text{(Mg ha}^{-1}\text{)} = \text{AGC} + \text{BGC} + \text{LC} + \text{SOC}$$

Where,

AGC = above ground carbon stock,

BGC = below ground carbon stock,

LC = litter carbon stock and

SOC = soil organic carbon.

3.8.5. Carbon Mapping

Mapping of carbon stock by exponential semivariogram model was done to estimate the spatial distribution of carbon values (Du *et al.*, 2010).

3.9. Ecosystem service valuation

Ecosystem service value coefficients were obtained from ecosystem service valuation database (Van der Ploeg & De Groot, 2010) to determine ecosystem service value from the five land cover categories. The values were modified through a benefit transfer method. The modified ecosystem service values used in this study can be found in Appendix C and D.

3.10. Estimation of ecosystem service values

Values of ecosystem service (ESV) from each land cover type were estimated by using the following formula as in Kindu *et al.* (2016) and Gashaw *et al.* (2018):

$$\text{ESV (US\$ yr}^{-1}) = \sum (A_k \times VC_k)$$

Where,

A_k is the area (ha) for land cover type *k* and

VC_k the ecosystem service value coefficient (US \$ ha^{-1} yr^{-1}) for land cover type *k*.

The total ESV of the entire study area for a particular year was computed by adding the ESV from each land cover type.

Ecosystem service values of functions (ESV_f) were calculated using the following equation:

$$ESV_f\ (US\$\ yr^{-1}) = \sum (A_k \times VC_{fk})$$

Where,

A_k is the area (ha) for land cover type *k* and

VC_{fk} the value coefficient of function f (US$ ha^{-1} yr^{-1}) for land cover category k.

Changes in ecosystem service values were obtained by computing the difference between the estimated values in each year and presented as percentages.

$$\%\ ESV\ change = \frac{ESV_{final\ year} - ESV_{initial\ year}}{ESV_{initial\ year}} \times 100$$

Where, ESV is total estimated ecosystem service value.

The average ecosystem service value (ESVav) of the study area (US$ ha^{-1} yr^{-1}) was estimated for the reference years using the following formula as in Gashaw *et al.* (2018).

$$ESV_{av} = \frac{ESV_t}{A}$$

Where, ESV_t is the total ESV in each year, and A is the area of the study site in ha.

Coefficient of sensitivity (CS) was calculated to determine the percentage change in ecosystem service values using the following equation (Li *et al.*, 2010a).

$$CS = \frac{(ESV_j - ESV_i)/ESV_i}{(VC_j - VC_i)/VC_{ik}}$$

Where,

ESV is the estimated ecosystem service value,

VC is the value coefficient,

i and *j* represent the initial and adjusted values, respectively, and

k represents the land cover category.

3.11. Statistical analysis

The SAS 9.0 was used to perform one-way analysis of variance (ANOVA) to test for mean differences in vegetation parameters, carbon stock means across land covers and carbon pools. Tukey HSD test was performed to separate means.

The Minitab 16 statistical software was used to perform Pearson correlation and multiple linear regression analyses on the following parameters. Soil organic carbon stock, biomass carbon stock, average tree diameter, average tree height and tree density. The multiple regression was used to select predictor variables.

CHAPTER FOUR

4. RESULTS

Findings on land cover change, carbon stock and ecosystem service values from different land cover types in different periods are reported.

4.1. Accuracy classification

Table 2 shows the results for accuracy classification. The overall accuracies for the classified thematic maps ranged from 90% to 93% with the Kappa coefficient ranging from 0.87 to 0.89. While the lowest user's accuracies were recorded in 2016 for cultivated land which is 84%, the highest user's accuracies were recorded in 1985 for bare land which is 100%. The highest producer's accuracies were observed in 2000 for dense forest and the lowest was recorded in 1985 for the open forest.

Table 2: Accuracy assessment of classified images.

	Accuracy (%)					
Land cover types	1985		2000		2016	
	Producer's	User's	Producer's	User's	Producer's	User's
Open forest	77	95	85	90	81	91
Dense forest	97	91	98	89	94	93
Cultivated land	95	90	95	91	97	84
Bare land	85	100	79	95	69	90
Grassland	89	93	82	90	96	93
Overall Accuracy	93		90		90	
Kappa Coefficient	0.89		0.87		0.87	

4.2. Land cover classification and changes

Table 3 and Figure 2 provide details of the area and land cover changes of five land cover types for the years 1985, 2000, and 2016. In 1985, dense forest was the dominant land cover followed by bare land, open forest, cultivated land and grassland, respectively. In 2000, dense forest constituted the largest part followed by open forest, bare land, cultivated land and grassland, respectively. Dense forest continued to be the dominant land cover in 2016 followed by open forest, cultivated land, bare land and grassland.

There were an increase in dense forest and open forest and a reduction in cultivated land, grassland and bare land between 1985 and 2000. Between 2000 and 2016, a decrease in dense forest and open forest and an increase in cultivated land and grassland were recorded.

Table 3: Area and land cover change in Wujig Mahgo Waren forest in 1985, 2000, and 2016.

Land cover types	Land cover distribution						Land cover changes (%)		
	1985		2000		2016		1985–2000	2000–2016	1985–2016
	Area (ha)	% cover	Area (ha)	% cover	Area (ha)	% cover			
Dense forest	4469	26	4836	28	4335	25	8	−10	−3
Open forest	3629	21	4802	28	4337	25	32	−10	16
Grassland	1713	10	1074	6	2035	12	−37	90	16
Cultivated land	3211	19	3035	18	3902	23	−6	29	18
Bare land	3999	24	3272	19	2417	14	−18	−26	−65

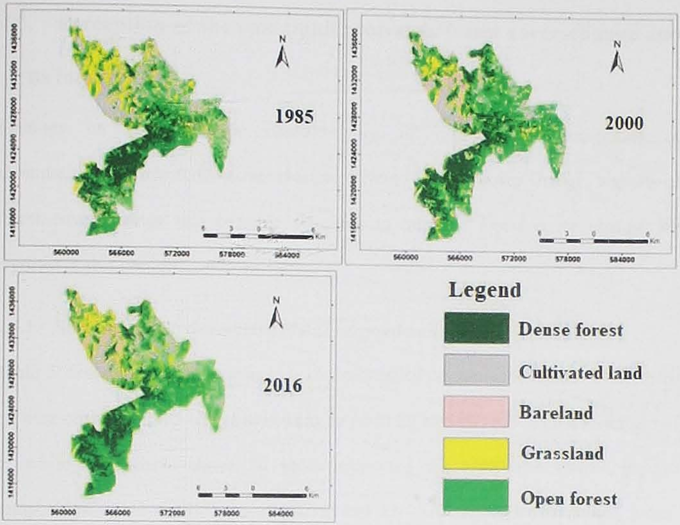

Figure 3: Classified satellite images showing the actual land cover change in Wujig Mahgo Waren forest in 1985, 2000 and 2016

Table 4 provides change matrix of land cover types between 1985 and 2016. Out of 3629 ha of open forest in 1985, 1320 ha remained unchanged. 2435 ha of dense forest remained unchanged, while 2035 ha were changed to other land cover types.

Table 4: Change matrix of land cover types between 1985 and 2016

	Land cover Type	Land cover state in 2016 (ha)					
		Open Forest	Dense Forest	Cultivated Land	Bare Land	Grassland	Total
Land cover state in 1985 (ha)	Open forest	**1320**	834	620	476	379	3629
	Dense forest	1263	**2435**	253	213	304	4469
	Cultivated land	572	251	**1627**	492	269	3211
	Bare land	928	705	1122	**766**	477	3998
	Grassland	248	110	279	470	**606**	1713
	Total	4332	4334	3902	2417	2035	17,019

4.3. Perception of the community towards forest cover change and its impact

Findings on socioeconomic characteristics of respondents, perceptions of communities towards forest cover change, drivers of forest cover change, impacts of forest cover change and possible solutions to improve forest cover change are reported in this section.

4.3.1. Socioeconomic characteristics of respondents

Table 5 describes the socioeconomic characteristics of the respondents. Sixty-three per cent of the sampled households were between 20 and 49 years with a mean age of 47 while respondents above 50 years accounted for 37%. The household size composition was 1–5 (51%), 6–10 (47%) and 11–15 (2.6%). The minimum, mean, and maximum household size was 1, 6, and 12 persons, respectively. Sixty-eight per cent of the respondents have a land size of less than 0.5 ha, with 28% owning between 0.75 and 1.25 ha. Four per cent of the respondents do not have land for cultivation. The mean household income for all the interviewed households was 7377 Ethiopian birr, which is equivalent to US$ 323 per year. Seventy-five per cent of the respondents were male, and only 29% of the respondents were literate. Approximately 97% were farmers and on!y 3% of the respondents depended primarily on other off-farm income sources like wood collection and trade.

Table 5: Sample household characteristics in the study landscape (N = 150)

Household attributes	Value
Gender (male, %)	75
Average household age (years)	47
Education (literate, %)	27
Household occupation (farming, %)	97
Mean household size (Number)	6
Mean land holding size (ha)	0.5
Mean household income (Birr[a]/year)	7377

[a] Ethiopian currency: at the time of the study, 1 US$ = 22.8 Birr

4.3.2. Perception of farmers towards forest cover change

The respondents indicated that there were differences regarding forest cover change. Seventy-three per cent of respondents perceived that the forest cover was decreasing. In contrast, 24% indicated that the forest was increasing in cover and 3% of the respondents stated that there had been no change in forest cover (Table 6). Significant differences were also found in perceptions among respondents in different kebeles concerning the forest cover change (Table 6). While 89% of the respondents from Hiziba Teklehimanot kebele perceived that the forest was declining; 50% from Tsgea kebele indicated the forest was increasing. This might be due to the location of communities and spatial differences of forest conditions within the study area. During the reconnaissance survey, it was observed that the forest cover in Tsgea Kebele was better compared to that of Hiziba Teklehimanot kebele.

Table 6: Farmer's perception of forest cover changes.

Forest cover change	Response by kebele												
	Hiziba T/himanot (N = 36)			Tsgea (N = 30)			Ebo (N = 46)			Ayiba (N = 38)			X^2
	1	2	3	1	2	3	1	2	3	1	2	3	
% of respondents	11	89	0	50	40	10	11	87	2	32	68	0	***

***Significant at $P < 0.001$, indicating that there is a significant difference in perception about the forest cover change between kebeles. 1 = increased, 2 = decreased and 3 = no change

4.3.3. Drivers of forest cover change

Table 7 presents the drivers of forest cover change as ranked by respondents. The respondents identified seven important factors responsible for land cover changes in the studied landscape. Fuelwood collection (71%) and the expansion of cultivated land (63%) were the most important drivers. In addition, some of the respondents reported house construction (51%), population growth (38%), pastoral use (35%), logging for income generation (34%) and drought (12%) as important causes for the observed land cover changes.

The focus group discussions in the different kebeles have indicated that;

i. Population pressure had a great impact on the forest dynamics.

ii. They depend on the sale of fuelwood as a source of income during decline or failure of crop production in drought years.

iii. Livestock grazing was also viewed as one of the important drivers for land cover changes.

Table 7: Drivers of forest cover change rankings by respondents.

S/n	Drivers	Rank					Weight	Percentage (%)	Rank
		1	2	3	4	5			
1	Cultivated land expansion	70	3	8	2	11	401	26.1	2
2	Fuel wood collection	35	66	4	1	0	453	29.5	1
3	Free grazing	0	18	32	2	0	172	11.2	4
4	Housing	4	12	46	14	1	235	15.3	3
5	Drought	4	8	5	1	0	69	4.5	7
6	Income generation	0	0	9	32	10	101	6.5	6
7	Population growth	1	1	0	38	17	102	6.6	5

Differences between respondent's perceptions of drivers of forest cover changes are presented in Table 8. Differences were found in perception among respondents in different kebeles regarding free grazing and income generation driving factors. While all of the respondents in Hiziba Teklehimanot perceived free grazing as a major driver of forest cover change, most farmers in Ayiba kebele did not consider it as a key driver of the change. Instead, the majority of the respondents from Ayiba kebele perceived income generation as a major driver of the observed changes.

Table 8: Farmer's perceptions of drivers of forest cover changes.

Forest cover Change Drivers	Response by Kebele								
	Hiziba T/himanot (N = 32)		Tsgea (N = 12)		Ebo (N = 40)		Ayiba (N = 26)		X^2
	yes	no	yes	no	yes	no	yes	no	
Cultivated land expansion	100	0	100	0	100	0	100	0	
Fuelwood collection	100	0	100	0	100	0	100	0	
Free grazing	100	0	92	8	100	0	56	44	***
Housing	100	0	100	0	100	0	100	0	
Drought	94	6	100	0	95	5	94	4	
Income generation	97	3	83	17	98	2	100	0	***
Population growth	97	3	100	0	98	2	100	0	
Wildfire	0	100	0	100	0	100	0	100	
Civil war and conflict	0	100	0	100	0	100	0	100	
Land tenure	0	100	0	100	0	100	0	100	

***Significant at $P < 0.001$, indicating that the location of households had a significant effect on the perception of farmers towards the drivers of land cover change.

4.3.4. Perception of farmers on impacts of deforestation

Figure 3 provides the perception of farmers regarding impacts of deforestation. Seventy-eight (78) out of a total of 150 respondents representing 52% indicated that soil erosion was a major impact of deforestation in the study area. Flooding and shortage of wood for fuel, housing and agricultural implements were each indicated by 35% of respondents as being caused by deforestation, whiles 11% and 12% of respondents respectively indicated drought and loss of honey bees were the impacts of deforestation.

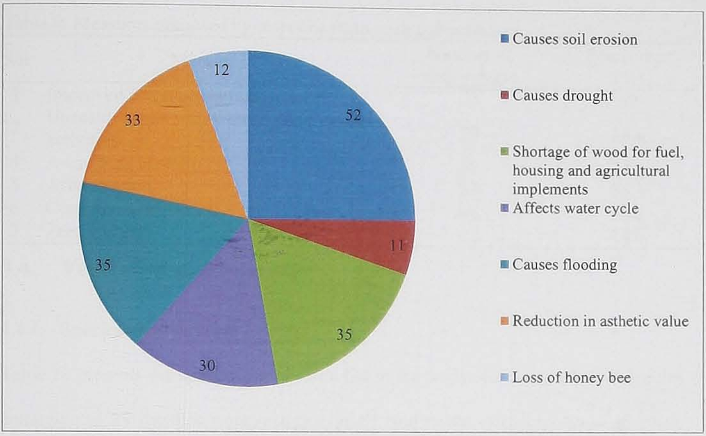

Figure 4: Impacts of deforestation as perceived by respondents.

4.3.5. Existing remedies and potential solutions for improving forest cover

The establishment of exclosures, formulation of bylaws, enrichment plantation and soil and water conservation structures were among the main activities that have been implemented. Respondents viewed that a portion of the forest land has been given to a church to be strongly preserved as a solution for the current deforestation.

Possible solutions including strengthening forest protection, improving soil and water conservation structures, awareness creation, enrichment planting, financing for added ecosystem services and introduction of zero grazing mechanisms were indicated by the respondents as mitigative measures to the current deforestation (Table 9).

Table 9: Measures proposed by respondents to curb deforestation.

S/n	Measures	Number of respondents	Percentage (%)
1	Improved forest protection	78	52
2	Upscale of soil & water conservation activities	58	38.6
3	Awareness creation	55	36.6
5	Afforestation	53	35.3
6	Compensation	46	30.6
7	Zero grazing	22	14.6

4.4. Vegetation characteristics

4.4.1. Species identification

Table 10 presents the species composition list in the study area. A total of 45 species belonging to 29 families were recorded in all land cover categories from all plots. *Cadia purpurea* (26%), *Dodonaea angustifolia* (13.7%), *Maytenus arbutifolia* (10.8), *Juniperus procera* (9.6%), *Calpurnia aurea* (5.6%), *Carissa spinarum* (3.2%) *and Acacia abyssinica* (3%) were the dominant species contributing 72% of the total species observed.

Table 10: Species composition list in the study area.

Scientific name	Family	Lifeform	Density (stems ha^{-1})
Abutilon longicuspe Hochst.exARich	Malvaceae	Shrub	3.6
Acacia abyssinica Hochst. ex Benth	Fabaceae	Tree	35.6
Acacia etbaica Sihweinf.	Fabaceae	Tree	4.6
Acacia seyal Del.	Fabaceae	Tree	0.7
Acacia tortilis (Forssk.) Hayne	Fabaceae	Tree	1.4
Acokanthera schimperi (A.DC.)Schweinf	Apocynaceae	Shrub & tree	2.5
Allophylus macrobotrys Gilg	Sapindaceae	Tree	1.05
Berberis holstii Engl.	Berberidaceae	Shrub	2.1
Bersama abyssinica Fresen.	Melianthaceae	Shrub & tree	1.8
Buddleja polystachya Fresen	Loganiaceae	Shrub	1.05
Cadia purpurea (Picc.)Ait	Fabaceae	Shrub	299.6
Calpurnia aurea (Ait.) Benth.	Fabaceae	Shrub & tree	65.1
Carissa spinarum L	Apocynaceae	Shrub	37.7
Celtis africana Burm.f	Ulmaceae	Tree	2.1
Clerodendron myricoides (Hochst.) Vatke	Verbenaceae	Shrub	1.8
Clutia abyssinica Jaub. & Spach.	Euphorbiaceae	Shrub	2.1

Cupressus lusitanica Miller	Cupressaceae	Tree	10.9
Discopodium penninervium Hochst.	Solanaceae	Tree	2.5
Dodonaea angustifolia L.f.	Sapindaceae	Shrub	158.8
Dovyalis abyssinica (A.Rich.)Warb.	Flacourtiaceae	Shrub & tree	4.6
Dovyalis verrucosa (Hochst.) Warb.	Flacourtiaceae	Shrub & tree	11.6
Ekebergia capensis Sparrm.	Meliaceae	Tree	15.1
Erica arborea L.	Ericaceae	Shrub & tree	13.0
Eucalyptus camaldulensis Dehnh	Myrtaceae	Tree	23.9
Eucalyptus globlus	Myrtaceae	Tree	18.7
Euclea racemosa Murr. subsp. Schimperi	Ebenaceae	Shrub	1.8
Juniperus procera Hochst.ex.Endl	Cupressaceae	Tree	112.3
Maytenus arbutifolia (A.Rich.)Wilczek	Celastraceae	Shrub	125.7
Maytenus undata (Thunb.) Blakelock	Celastraceae	Shrub & tree	11.6
Myrsine africana L.	Myrsinaceae	Shrub	8.5
Nuxia congesta R.Br.ex.Fresen	Loganiaceae	Tree	5.6
Olea europaea L. Subsp. *Cuspidate*	Oleaceae	Tree	52.5
Osyris quadripartita Decn.	Santalaceae	Shrub	15.1
Otostegia integrifolia Benth.	Lamiaceae	Shrub	0.7
Pittosporum viridiflorum Sims	Pittosporaceae	Shrub & tree	15.1
Podocarpus (Afrocarpus) falcatus (Thun) Mirb.	Podocarpaceae	Tree	12.7
Psydrax schimperiana (A.Rich.)Bridson	Rubiaceae	Shrub & tree	3.9
Pterolobium stellatum (Forssk.)	Fabaceae	Shrub	12.3
Rhus glutinosa A.Rich.	Anacardiaceae	Tree	20.4
Rhus natalensis Bernh. ex Krauss	Anacardiaceae	Tree	9.5
Rosa abyssinica Lindely	Rosaceae	Shrub	2.5
Rumex nervosus Steud.ex A.Rich.	Polygonaceae	Shrub	4.6
Solanum schimperianum Hochst	Solanaceae	Shrub	12.3
Teclea simplicifolia (Engl.) Verdoorn	Rutaceae	Shrub & tree	8.5
Toddalia asiatica (L.) Lam.	Rutaceae	Climber	1.1

4.4.2. Vegetation parameters

Table 11 indicates woody plant dendrometric variables and average number of stems under different land cover types. Stem density ranged from 196 to 1618, with the highest value recorded in dense forest and lowest in grassland. The mean stem diameter of dense forest was significantly higher than the diameter of open forest and grassland. Dense forest recorded highest average woody plant height followed by open forest and grassland.

Table 11: Mean values of woody plant dendrometric variables under different land cover types.

Land cover types	Dendrometric variables		
	DBH (cm)	H (m)	No of stems ha^{-1}
Dense forest	7.21±0.51^{a}	4.3±0.44^{a}	1618.3±93.4^{a}
Open forest	5.56±0.47^{b}	3.03±0.21^{b}	959.1±64.9^{b}
Grass land	2.96±0.17^{c}	1.89±0.10^{b}	196.9±19.7^{c}
p-value	0.0003	0.001	<0.0001

DBH = diameter at breast height, H = height. Values within a column with same letters are not significantly different (p > 0.05) according to Tukey's HSD test.

4.5. Soil characteristics

Soil physical properties and soil organic carbon in different land cover types are shown in Table 12. Percentage organic carbon concentration ranged between 2.0 to 3.1, with the highest occurring in dense forest, and lowest on bare land. In the dense forest, soils were higher in clay content than in the open forest, and the mean value for bulk density varied from 1.11 to 1.37 g cm^{-3}, with the highest content in bare land and lowest in dense forest.

Table 12: Mean values of three soil properties in different land cover types.

Land cover types	Particle size distribution			OC (%)	BD (g cm^{-3})
	Sand (%)	Silt (%)	Clay (%)		
Dense forest	33.1±3.3^{ab}	34.5±2.9^{a}	32.3±2.3^{a}	3.1±0.17^{a}	1.11±0.05^{a}
Open forest	27.2±2.6^{ab}	43.8±2.2^{a}	28.0±1.6^{a}	2.7±0.16^{a}	1.17±0.04^{a}
Grassland	29.2±8.1^{ab}	42.5±7.1^{a}	28.3±2.6^{a}	2.8±0.27^{a}	1.28±0.07^{a}
Cultivated land	19.7±3.6^{b}	48.4±4.4^{a}	31.8±3.2^{a}	2.2±0.33^{a}	1.31±0.04^{a}
Bare land	50.2±11.1^{a}	27.8±8.0^{a}	22.0±6.0^{a}	2.0±0.25^{a}	1.37±0.05^{a}
p-value	0.023	0.02	0.23	0.06	0.035

OC = Soil organic carbon, BD = Bulk density. Values within a column with same letters are not significantly different (p > 0.05) according to Tukey's HSD test.

4.6. Effect of land cover change on carbon stock

Carbon stocks, the contribution of carbon pools to total carbon stock, changes in carbon stocks are presented in this section.

4.6.1. Carbon stocks

Mean estimated values of carbon stock across the land cover types are presented in Table 13. Mean above ground biomass carbon stock ranged from 3.43 Mg ha^{-1} in grassland to 65.8 in dense forest. Highest below ground biomass carbon stock was recorded in dense forest with a mean value of 11.4 Mg ha^{-1}, while lowest was recorded in grassland with a mean value of 1.02 Mg ha^{-1}. The carbon content of litter biomass varied from 1.17 to 2.25 Mg ha^{-1}, with the highest value in dense forest and lowest in grassland. The carbon content of litter biomass was significantly higher under dense forest than grassland. Grassland recorded highest soil organic carbon with a mean value of 103.1 Mg ha^{-1}. The lowest soil organic carbon stock was recorded in cultivated land (76.54 Mg ha^{-1}). Total carbon stock ranged from 76.54±7.84 Mg ha^{-1} to 181.78±27.06 Mg ha^{-1}, with the highest value recorded in dense forest and lowest in cultivated land.

Table 13: Estimated carbon stocks (Mg ha^{-1}) across the land cover types.

	Land cover types					***p*-value**
Carbon pools	**Dense forest**	**Open forest**	**Grassland**	**Cultivated land**	**Bare land**	
AGC	65.81±18.50^{a}	12.67±2.22^{b}	3.43±0.33^{b}	-	-	<0.001
BGC	11.38±2.61^{a}	2.92±0.41^{b}	1.02±0.08^{b}	-	-	<0.0002
LC	2.25±0.27^{a}	1.68±0.20^{ab}	1.17±0.09^{b}	-	-	<0.0048
SOC	102.33±13.19^{a}	87.55±12.73^{a}	103.13±6.75^{a}	76.54±7.84^{a}	83.13±8.53^{a}	0.2712
Total	181.78±27.06^{a}	104.83±12.35^{b}	108.77±6.77^{b}	76.54±7.84^{b}	83.11±8.53^{b}	<0.0001

AGC = above ground carbon, BGC = below ground carbon, LC = Litter carbon, SOC = soil organic carbon. Values within a row with same letters are not significantly different ($p > 0.05$) according to Tukey's HSD test.

Figure 4 shows the spatial distribution of carbon stocks in the study area. The highest carbon stock was found in the southwestern part of the study area with a value of 167.6-187.1 Mg ha^{-1}.

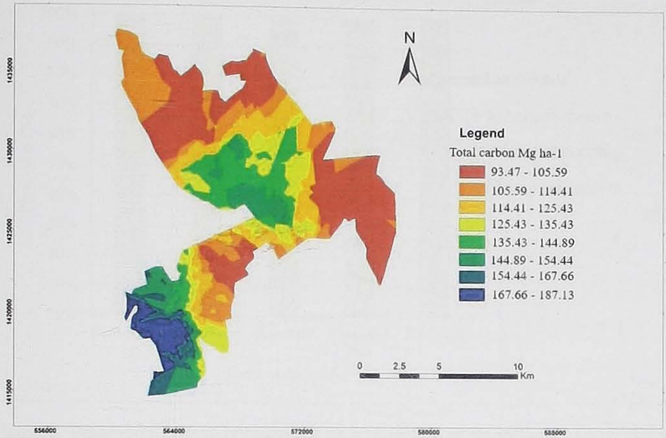

Figure 5: Spatial distribution of carbon stock in the study area for 2016.

4.6.2. Contribution of carbon pools

Figure 5 shows the percentage carbon contribution of four carbon pools in the five land cover types. Soil organic carbon contributed the highest (56%) to total carbon stock of dense forest, followed by above ground biomass carbon (36%). In the dense forest, the lowest carbon stock was recorded in litter carbon pool (1.2%). In open forest highest carbon stock was stored in soil (83.6%) followed by above ground carbon (12.1%), below ground carbon (2.7%) and litter carbon (1.6%). In Grassland 94.8% of the total carbon stock was recorded from soil organic carbon, followed by above ground biomass carbon (3.1%). Soil organic carbon accounted for 100% of total carbon stocks for cultivated land and bare land.

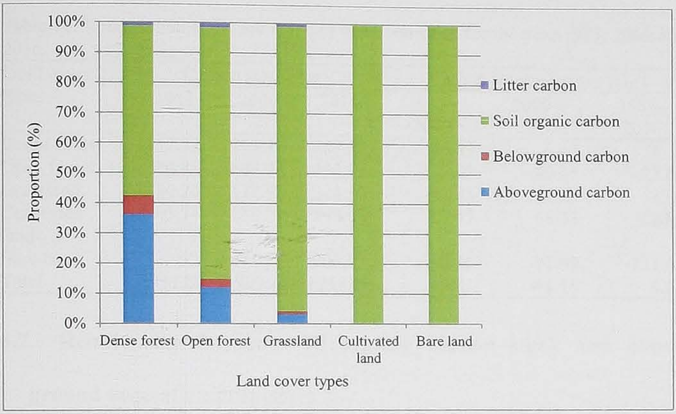

Figure 6: Percentage carbon contribution of four carbon pools in the five land cover types.

4.6.3. Changes in carbon stock

Table 14 presents total carbon stock and changes in carbon stock for the year 1985, 2000 and 2016. Carbon stock was 1951 Gg carbon, 1999.8 Gg carbon and 1955.6 Gg carbon in 1985, 2000 and 2016 respectively. In 1985, 2000 and 2016 highest carbon stock was recorded in dense forest followed by open forest. Dense forest recorded highest carbon stock (879.2 Gg) in 2000. Cultivated land recorded highest carbon stock (298.6 Gg) in 2016.

While carbon stock increased by 48.8 Gg between 1985 and 2000, decreased by 44.2 Gg between 2000 and 2016. Between 1985 and 2016 an increase of 4.6 Gg carbon was observed.

Table 14: Total carbon stock and changes in carbon stock for the year 1985, 2000 and 2016.

Land cover types	Carbon stock (Gg)			Carbon stock changes (Gg)		
	1985	2000	2016	1985–2000	2000–2016	1985–2016
Dense forest	812.32	879.15	787.95	66.83	-91.2	-24.37
Open forest	380.42	503.44	454.09	123.02	-49.35	73.67
Grassland	180.20	112.97	214.10	-67.23	101.13	33.9
Cultivated land	245.74	232.29	298.62	-13.45	66.33	52.88
Bare land	332.32	271.96	200.87	-60.36	-71.09	-131.45
Total	1951.00	1999.81	1955.63	48.81	-44.18	4.63

4.7. Relationship between soil organic carbon stock and above ground vegetation properties

4.7.1. Correlations between soil organic carbon stock and Vegetation Parameters

Table 15 presents the Pearson correlation values between vegetation parameters and soil organic carbon stock. Highest statistically significant correlations were found between above ground biomass and DBH (Pearson correlation 0.719, $p<0.01$), followed by above ground biomass and soil organic carbon stock (Pearson correlation 0.699, $P<0.01$). The lowest correlation was found between tree density and DBH (Pearson correlation -0.042, $P>0.05$).

Table 15: Pearson correlation coefficient values of soil organic carbon, DBH, height and biomass.

	SOC	DBH	H	Tree density	agB
SOC	-				
DBH	0.627**	-			
H	0.502*	0.107	-		
Tree density	0.437	-0.042	0.457	-	
AGB	0.699**	0.719**	0.560*	0.382	-

(*: P<0.05; **: P<0.01

4.7.2. Regression models of soil organic carbon stock

Table 16 shows results for simple and multiple regression model of soil organic carbon. The coefficient of determination ranged from 0.13 to 0.59. The combination of tree density, DBH and height were recorded highest determination coefficient with 0.59 adj. R^2. Lowest determination coefficient was recorded for tree density with 0.13 adj. R^2.

Table 16: Regression model of soil organic carbon stock.

Dependent variable	Predictor variables	Adj. R^2	P-value
SOC	Tree density	0.13	0.091
SOC	DBH	0.35	0.009
SOC	Height	0.19	0.047
SOC	agB	0.45	0.003
SOC	Tree density * DBH	0.55	0.002
SOC	Tree density * Height	0.20	0.093
SOC	Tree density * agB	0.45	0.008
SOC	DBH * Height	0.52	0.003
SOC	DBH * agB	0.49	0.008
SOC	Height * agB	0.43	0.010
SOC	Tree density * DBH * Height	0.59	0.003
SOC	Tree density * DBH * agB	0.52	0.007
SOC	Tree density * Height * agB	0.41	0.025
SOC	DBH * Height * agB	0.49	0.011
SOC	Tree density * DBH * Height * agB	0.56	0.010

SOC = Soil organic carbon, DBH = Diameter at breast height, agB = Above ground biomass,

4.8. Effects of land cover change on ecosystem service values

Changes in ecosystem service values in response to land cover change and coefficient of sensitivity were presented in this section.

4.8.1. Changes in ecosystem service values

The changes in the value of ecosystem service from 1985 to 2016 periods for each land cover type and total ecosystem service value were estimated and presented in Table 17. The total ecosystem service values of the whole study landscape were about US$ 16.6, 19.0 and 18.1 million in 1985, 2000 and 2016, respectively. In 1985, of the total ecosystem service values in the study area, dense forest, open forest, grassland and cultivated land recorded US$ 7.9 million (47.7%), 6.4 million (38.8%), 0.77 million (4.6%), and 1.5 million (8.9%), respectively, of the total ecosystem service values. In 2000, dense forest and open forest recorded the highest value, i.e. US$ 8.6 million (45.2%), 8.5 million (44.9%) while cultivated land and grassland contributed US$ 1.4 million (7.4%), 0.48 million (2.5%), respectively. Dense forest, open forest, grassland and cultivated land recorded for about US$ 7.7 million, 7.7 million, 0.9 million, and 1.8 million, respectively in 2016.

While total ecosystem service value increased by 15.7% between 1985 and 2000, it decreased by 5.2% decrease between 2000 and 2016. Between 1985 and 2016, total ecosystem service values increased by 9.6%.

The change in ecosystem service values differed among the land cover types as observed in the contributions of ecosystem service values over the study periods (Table 17). For example, between 1985 and 2000, ecosystem service value of the dense forest and open forest increased by 8.2% and 32.3%, while grassland and cultivated land decreased by 37.3% and 5.5%, respectively. However, between 2000 and 2016, the value of ecosystem service of dense forest and open forest decreased by 10.4% and 9.8%, while grassland and cultivated land increased by 89.5% and 28.9%, respectively.

Table 17: Estimated ecosystem service values and dynamics for each land cover type of the three reference years and periods in US$ million.

Land cover types	Ecosystem service values						Ecosystem service value changes (%)		
	1985		2000		2016		1985-2000	2000-2016	1985-2016
	US$	%	US$	%	US$	%			
Dense forest	7.9	47.7	8.6	45.2	7.7	42.5	8.2	-10.4	-3.0
Open forest	6.4	38.8	8.5	44.9	7.7	42.5	32.3	-9.8	19.3
Grassland	0.77	4.6	0.48	2.5	0.9	5.0	-37.3	89.5	18.8
Cultivated land	1.49	8.9	1.4	7.4	1.8	10.0	-5.5	28.6	21.5
Bare land	0	0	0	0	0	0	0	0	0
Total	16.6	100	19.0	100	18.1	100	15.7	-5.2	9.6

The average ecosystem service values of the land increased from 976.47 US$ ha^{-1} yr^{-1} in 1985 to 1117.64 US$ in 2000 and decreased to 1064.7 in 2016 (Figure 6).

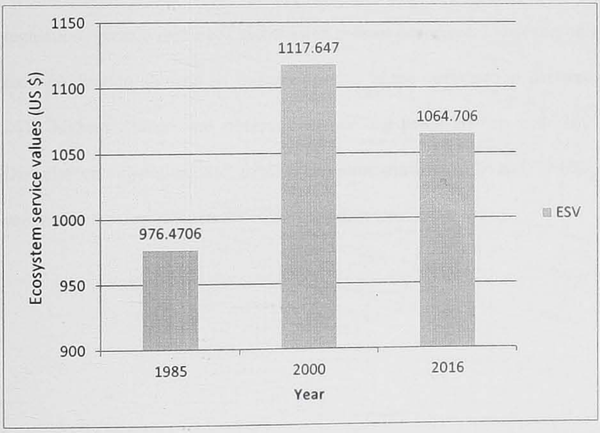

Figure 7: The average ecosystem service value of the study area for 1985, 2000 and 2016.

4.8.2. Estimated services of specific ecosystem functions and their changes

Table 18 indicates estimates of ecosystem service values and their changes for specific ecosystem services. In 1985, specific ecosystem service value ranged from US$ 0.1 to 4.1 million, with the highest recorded in climate regulation service, and lowest on water regulation, gas regulation and cultural services. In 2000, the ecosystem service value varied between US$ 0.1 and 4.8 million, with the highest recorded in climate regulation service, and lowest in gas regulation and cultural service. In 2016, the ecosystem service value varied between US$ 0.1 to 4.4 million, with the highest recorded in climate regulation service, and lowest in water regulation and cultural service.

Between 1985 and 2000 highest change was observed in water regulation with 100% increase. Followed by water regulation, disturbance regulation, recreation and erosion control which recorded 25%, 20.7% and 19.2% increase in value in the same period respectively. Between 2000 and 2016 values of water regulation, recreation, climate regulation, genetic resources and erosion control decreased. However, pollination and food production showed an increased value in the same period. Between 1985 and 2016 highest change was observed in gas regulation service with 100% increase. Disturbance regulation and food production showed 25% and 17.6% increase in ecosystem service value in the same period, respectively.

Table 18: Estimated value of ecosystem service for specific service for 1985, 2000 and 2016 reference years and their changes.

Ecosystem services	Ecosystem service values (ESV) US$ million year^{-1}			Ecosystem service value change (%)		
	ESV_{1985}	ESV_{2000}	ESV_{2016}	1985-2000	2000-2016	1985-2016
Provisioning services						
Water supply	0.5	0.5	0.5	0	0	0.0
Food production	1.7	1.7	2.0	0	17.6	17.6
Raw material	1.1	1.2	1.2	9	0	9.0
Genetic resources	1.0	1.2	1.1	20	-8.3	10.0
Regulating services						
Water regulation	0.1	0.2	0.1	100	-50	0.0
Water treatment	1.7	2.0	1.8	17.6	-10	5.9
Erosion control	2.6	3.1	2.8	19.2	-9.6	7.7
Climate regulation	4.1	4.8	4.4	17.2	-8.3	7.3
Biological control	0.2	0.2	0.2	0	0	0.0
Gas regulation	0.1	0.1	0.2	0	1	100
Disturbance regulation	0.4	0.5	0.5	25	0	25
Cultural services						
Cultural	0.1	0.1	0.1	0	0	0.0
Recreation	2.9	3.5	3.2	20.7	-8.5	10.3
Total	16.5	19.1	18.1	15.7	-5.2	9.6

4.8.3. Ecosystem service sensitivity to land cover change

Table 19 shows ecosystem service sensitivity values obtained from the change in ecosystem service valuation coefficient.

Table 19: Change in total ecosystem service value (ESV) and coefficient of sensitivity (CS) resulting from a 50% change in the ecosystem value coefficients (VC).

Change in valuation coefficient (VC)	The effect of changing CV from the original value			
	1985		2016	
	%	CS	%	CS
Dense forest VC±50%	±23.8	0.48	±21.3	0.43
Open forest VC±50%	±19.3	0.40	±21.2	0.42
Grassland VC±50%	±2.3	0.05	±2.5	0.05
Cultivated land VC±50%	±4.4	0.09	±4.9	0.1

The coefficient of sensitivity was less than one in all cases. The estimated value of the ecosystem service value for the study area increased from a low of 0.05 for grassland to a high of 0.48 for a dense forest when the ecosystem value coefficients for these land cover types were changed by 50%. Changing the ecosystem service value coefficient of the dense forest by 50% affected the estimated 1985 and 2016 ecosystem service value by ±23.8% and ±21.3 % respectively. Similarly, changing the open forest coefficient by 50% affected the estimated 1985 and 2016 ecosystem service value by ±19.3% and ±21.2% respectively. Adjusting the grassland coefficient by 50% affected the estimated 1985 and 2016 ecosystem service value by ±2.3% and ±2.5% respectively. Adjusting the cultivated land coefficient by 50% affected the estimated 1985 ecosystem service value by ±4.4% and the 2016 value by ±4.9%.

CHAPTER FIVE

5. DISCUSSION

The magnitude and the percentage change of land cover types and their drivers, the impact of the change on communities, the changes in vegetation composition and structure of the study area, levels of carbon stocked in different land cover types and their changes, the relationship between soil organic carbon stock and above ground vegetation properties and changes in ecosystem service values were discussed.

5.1. Land cover change, their drivers and impacts

In this study, results of change analysis over 30 years showed the magnitude of change that occurred in five land cover types. The results reflected the substantial land cover change that took place in the Wujig Mahgo Waren forest within the last 30 years. Between 1985 and 2000, the area of dense forests and open forest recovered by 8% and 32% respectively while grassland, cultivated land and the bare land declined by 37%, 6% and 18% respectively. Forest management interventions such as intensive soil and water conservation and exclosure establishment could be responsible for the improvement in dense forest cover between 1985 and 2000. In fact, they allowed the forest some period of time to recover for which some improvements have been observed.

Between 2000 and 2016, the area of dense forests, open forest and bare land degraded by 10%, 10% and 26% respectively while grassland and cultivated land improved by 90% and 29%, respectively. The key causes of the loss in both dense and open forest between 2000 and 2016 were fuelwood collection for household use and income generation, cultivated land expansion, population growth, free grazing and drought. These results compare favourably with previous findings in different parts of the country (Demissie *et al.*, 2017; Fetene *et al.*, 2016; Hailemariam *et al.*, 2016; Kindu *et al.*, 2013). For example, a study by Demissie *et al.* (2017) on land use land cover change and their causes in the Libokemkem district of South Gonder, Ethiopia found that, forest land degraded by 82% between 1973 and 2015 as a result of population pressure, cultivated lands expansions, increasing wood collection for fuel, collection of farm implement and construction wood, charcoal production, regime changes, livestock grazing, the introduction of area closures and government programs such as military camp construction, villagization, land redistribution, and land compensation. Similarly, Fetene *et al.* (2016) revealed that the 38% loss of forest area in Nech Sar national park, Ethiopia between 1985 and 2011 were due to changes in anthropogenic land use and variation in precipitation. A significant loss of forest cover was also reported by Hailemariam *et al.* (2016) in the Bale mountains between 1985 and 2015 which was reportedly caused by cropland and urban expansion. In a similar study by Dessie and Christiansson (2008), a decline in forest area from about 40% at the turn of the 19th century to less than 3% in the year 2000 was observed in the south-central Rift Valley of Ethiopia. In Girmay *et al.* (2010)'s study on change in land use and its associated impacts on soil degradation and surface runoff, an expansion of cultivated land and a decrease in forest and woodland in northern Ethiopia from 1964 to 2006 was observed. Such changes of forest cover are also evident in different parts of the world. For example, Maimaitiaili *et al.* (2018) stated that the 62.5% forest loss observed between 1972 and 2014 in Kashgar region, northwest China was the consequence of population growth, changes in land tenure and socioeconomic development.

The current expansion of farmland and grassland at the expense of forests in the current study area might be a manifestation of the weak or inappropriate institutional arrangements in the study area as seen in other parts of Ethiopia. For example, a study by Hailemariam *et al.* (2016) on institutional arrangements and management of environmental resources found that limited institutional capacity constrained the management of environmental resources in three eco-regions of Ethiopia.

In the current study, respondents pointed out fuelwood collection as a dominant driver of deforestation. In the study area, the main source of energy for the majority of the people is wood due to the lack of alternative energy sources. Similarly, Noriko *et al.* (2012) indicated that fuelwood collection was the main degradation drivers for the African continent. Also, the cultivation of new agricultural lands to feed the ever growing population was identified as significantly contributing to the current deforestation process in the study area. Similar results were recorded by Noriko *et al.* (2012) indicating that agriculture alone caused 73% of all deforestation in developing countries. DeFries *et al.* (2010) and Fisher (2010) also stated that the existing rate of deforestation in Africa is largely driven by small-scale subsistence agriculture. In the present study, the construction of houses construction was also mentioned as one of the prime drivers of forest loss. Kindu *et al.* (2015) drew similar conclusions from their research as they stated housing as the main cause of land use land cover change in the Munessa-Shashemene landscape of the south-central highlands of Ethiopia. As observed, the communities in the present study mainly depend on the forest for the construction of their homes. The species *Cadia purpura* is the preferred species for housing projects in the community.

Population pressure also featured as an indirect driver of forest loss. Previous studies in other parts of the country reported population pressure as a major driver of land cover changes (Dessie & Kleman, 2007; Hurni *et al.*, 2005; Kidane *et al.*, 2012).

Respondents also pointed out that they depend on the selling of fuelwood for subsistence, a reliable source of income during decline or failure of crop production in periods of drought. In most of these cases, farmers living in the study area and their neighbouring areas are forced to depend on the remaining forests either to create additional croplands or to provide a source of income to help sustain them. This is a common survival strategy of rural populations in the events of degradation, drought, and rainfall variability across Africa as perceived by Campbell (1990).

Livestock grazing was also suggested as one of the key drivers of deforestation. Kindu *et al.* (2013) and Demissie *et al.* (2017)'s researches showed that livestock ranching was one of the drivers of land use land cover change in Ethiopia. In the study area, cattle grazed on crop residues in cropland and grasslands. However, due to the inadequate feed in the cropland and grasslands, farmers sent their cattle into the forest. This practice inhibits natural regeneration by suppressing seedlings which can eventually cause forest degradation.

The effect of the current level of deforestation on communities' ecological dynamics has been reaffirmed by respondents. Soil erosion, reduction in pollinating agents like honey bees as a result of reduction in flower production, flooding and drought were the perceived major impacts of the changes. A study on restoration ecology by Hobbs and Harris (2001) indicated that deforestation globally increases the rates of soil erosion since the amount of runoff is increased and the soil is no longer protected from tree litter. A study on the impact of tropical deforestation on hydrological functions by Bruijnzeel (2004) states that deforestation affects water cycle by reducing rainfall infiltration opportunities, with a substantial reduction in dry season flows in southeast Asia. Similarly, David and Roni (2002) concluded that deforestation in the Amazon forest brought about a reduction in precipitation, evapotranspiration, and cloudiness in Amazon and several other regions of the world. In contradiction to this present study's analysis, Bruijnzeel (2004) mentioned that the total rainfall increase with the percentage of forest biomass removed, alongside maximum gains in water yield upon total clearing of forests in southeast Asia. Therefore the impact of land cover change on drought depends on the differences in rainfall and degree of surface disturbance.

Respondents in the present study stated a reduction in pollinating agents like honey bees due to the production of fewer flowers as a result of forest degradation. Ejigu *et al.* (2009)'s findings were compatible with this notion of shortage in bee forage hence posing it as a major constraint of bee-keeping in the Amhara region, Ethiopia.

The ongoing land cover changes in the Wujig Mahgo Waren forest have different implications. If the depletion of dense forest cover continues, it can negatively change the potential use of the study area and may ultimately lead to the loss of productivity. Previous assessments by Lemenih *et al.* (2005) on soil chemical and physical properties following deforestation and subsequent cultivation of land by smallholder farmers in Ethiopia revealed an overall declining trend in almost all soil quality. This can affect the local people by reducing the means of their livelihood, especially for those who depend on the land. Other implications associated with the increasing loss of the forest resource is the decline in biodiversity. Furthermore, the ongoing deforestation supports expectations of Ciais *et al.* (2009) who predicted that the African forest carbon stocks would remain vulnerable due to deforestation. Houghton *et al.* (2012) also revealed that land use and land cover change can cause a change in carbon stocks and release substantial amounts of carbon from the forest to the atmosphere.

5.2. Existing remedies, potential solutions and challenges

To mollify the existing practice and effects of deforestation, several interventions have been implemented by the community and the government. The establishment of exclosures, formulation of bylaws, afforestation and soil and water conservation structures are among the main activities that have been implemented to reduce deforestation in the study area.

In the present study, the forest area was made inaccessible to humans and animals to promote natural regeneration of plants on formerly degraded communal grazing lands. The establishment of exclosures have been one of the strategies for rehabilitating degraded hillsides within catchments delineated for the rehabilitation and soil and water conservation programs. In line with the present study, Kassa *et al.* (2017) and Birhane *et al.* (2017a) have stated that since the 1980s, area exclosure has been introduced to manage degraded lands and increase productivity in northern Ethiopia. In addition, community members work to enrich the forest land by planting seedlings which provide ecological and economic benefits. This result is supported by Birhane *et al.* (2017a), who discovered that community members in Tigray region, northern Ethiopia had participated in enrichment planting programs to improve their forest cover. Inside the forest, different soil and water conservation activities have been carried out to reduce soil erosion and to increase the moisture content of the land so that seed banks easily regenerate, eventually increasing the forest cover. This was common practice in different areas of the country. For instance, Birhane *et al.* (2017a), who studied exclosures as a forest restoration tool indicated that the construction of soil and water conservation structures is one of the measures practised by the community to rehabilitate degraded lands in northern Ethiopia. Seyoum *et al.* (2015) also reported that massive soil and water conservation programs were carried out to restore degraded lands in northern Ethiopia.

In addition to the establishment of exclosure, village bylaws were formulated by the community to manage the forest resources in the study area. The bylaws constituted a set of understandings and principles that were designed by the community and to oversee public assets. Bylaws are used as mechanisms to manage communal grazing lands and exclosures, among others, in northern Ethiopia. For example, Yohannes and Waters-Bayer (2007), who studied integrated watershed management found that communities developed bylaws to manage exclosure in Geregra watershed, northern Ethiopia. Similarly, Yami *et al.* (2013) who studied the effectiveness of village bylaws in the sustainable management of community-managed exclosures found that bylaws developed by communities were used to manage exclosures in Tigray, northern Ethiopia, although they questioned its effectiveness. According to their bylaws, nobody had the right to cut live plants for fuel or construction unless permission was granted. Dead and dried wood could be collected for household use. Animals were also not allowed to graze in the forest. However, beekeeping was allowed, especially for youth association activities.

In this current study, it has been ascertained that farmers are of the view that a portion of the forest land was given to the church in order to strongly preserve it— a solution for the present deforestation problem. The church was considered an important place for forest conservation by many scholars. For instance, a study by Bongers *et al.* (2006) on ecological restoration and church forests in northern Ethiopia stated that churches are secure habitats for plants, animals, and green spaces for people. Moreover, they stated that church compounds served as conservation sites and hotspots of biodiversity, mainly indigenous trees and shrubs of Ethiopia. Also, Wassie *et al.* (2005) who studied church forest in Gondar, northern Ethiopia found that churches comprised plants and animals that have disappeared in most parts of northern Ethiopia indicating that churches could hence be an open door for forest ecosystem preservation and reclamation. Aerts *et al.* (2016) who studied conservation of the Ethiopian church forests stated that the churches have high conservation value with most of the native forests and forest biodiversity confined in the church. In Ethiopia, no individual can touch a tree if it is under the ownership of a church as they consider it to be holy. There is a belief that if you cut a tree inside a church, it is considered a sin. Thus, this provides an important opportunity for conserving the dwindling forest resources in the country.

Even though the aforementioned remedies are in place, it has been clearly shown that there is ongoing forest degradation in the study area. Respondents thereby suggested some extra measures to solve the ongoing deforestation problem in the study area. Possible solutions to the current state of deforestation from respondents' perspective include strengthening of forest protection and monitoring systems, improving soil and water conservation mechanisms, enhancing afforestation, creating awareness about the importance of forest ecosystems and pursuing zero grazing campaigns.

In Wujig Mahgo Waren, the forest area was protected by guards who had been appointed by the community and were being paid by the government. This practice was taking place in different parts of the country. For example, Yohannes and Waters-Bayer (2007) reported that communal forests were being protected by five guards and a success story was recorded in Gergera watershed of northern Ethiopia. In the same report, Yohannes and Waters-Bayer (2007) indicated that pasture land was being protected by paid guards in Sangade pasture, northern Ethiopia. Birhane *et al.* (2017a) also highlighted the fact that restoration areas were under the protection of guards who had been assigned by villagers in Tigray, northern Ethiopia. In the present study, the guards were receiving an equivalent of US$ 25 per month. However, since this salary is not enough, the guards are quitting their jobs to find better income sources. Therefore, the respondents suggested that the guards should be paid more than what they are receiving so they fortify attempts at protecting the forest.

Moreover, education (awareness creation) was also viewed as an essential management tool for acquiring sustainable forest management skills. In line with this study, Yismaw *et al.* (2014) found that 23% of the respondents in Banja district, Amhara region, Ethiopia suggested education as a solution for deforestation. Kebede *et al.* (2014) who studied natural resources use and conflict in Bale mountains mentioned that stockholders suggested awareness creation for local communities in order to reduce conflict on natural resources management. A review by Abebe and Bekele (2018) also suggested awareness creation and development as strategies for solving challenges of natural resources management in Ethiopia.

However, the participants of the focus group discussions expressed their views that there are challenges that hinder the application of the suggested solutions. They accused the government was not committed to supporting the forest management activities already outlined. They further drew attention to the high turnovers of development agents and government staff, and the absence of clarity and transparency in arrangements which are thought to influence forest management. They said that the government was not strongly committed to allocating ample resources to monitor and improve the management of the forests. They also opined that the support from the government was unsatisfactory as more help is needed by way of providing legal assistance for forest users against encroachers and offenders. Illegal timber producers caught and sent to court for prosecutions are often released in a few days without penalty. Upon their return, they harass local administrators and other people who handed them over to the authorities. Due to the weak law enforcement strategies, there is a continuous clash between guards and offenders. A similar case was also reported by Gobeze *et al.* (2009) who stated that weak government support threatened participatory forest management program in Bonga forest, Ethiopia.

The need to sustain livelihoods is paramount to everyone. In this study, farmers asserted that though they are not allowed to cut live trees, they do illegal cutting for different purposes. In the study area, the species *Cadia purpura* is highly exposed to illegal cutting as it is the main sources of construction material in the community. Overall, it was observed that with diminishing legitimate access to the state forests, the illicit activities of cutting down trees is likely to increase because of the lack of alternative forest resources to be used by the community.

The unemployment rate of the local people is another main challenge in the study area. To satisfy their essential needs and prerequisites, the jobless individuals within the zone utilize these forests as a wellspring for earning money through illegitimate means. Similarly, Vajpeyi (2001) reported that economic conditions and unemployment caused deforestation in Kenya. Tariq and Aziz (2015) investigated the assertion that unemployment was the main cause of deforestation in Khyber Pukhtunkhwa, Pakistan. In agreement with the present study, Hussaini (2014) demonstrated that unemployment contributed to deforestation in Bauchi State. Geist and Lambin (2001) also stated that high unemployment rates and a lack of alternative non-agrarian income are among the main drivers of deforestation in Asia and Latin America.

5.3. Woody species composition and vegetation characteristics across land cover types

Woody species composition and vegetation characteristics affect soil characteristics and carbon stocks. In the current study, a total of 45 species belonging to 29 families were recorded in all land cover categories from all plots. In line with this study, Aerts *et al.* (2006) found 40 total species in a small Afromontane forest fragments in northern Ethiopia. Contrary to the present study, however, Mengistu *et al.* (2005) found lower numbers of species with 31 woody species belonging to 19 families at Tiya enclosure site, northern Ethiopia. Findings with a lower number of species than the present study were also made by Abebe *et al.* (2006) who observed 18 woody species in closed areas of Wukro district, northern Ethiopia. Despite these statistics, the number of species at Wujig Mahgo Waren was lower compared to other tropical and subtropical dry forests. For example, Aynekulu (2011) observed 79 species belonging to 51 families in the Hugumburda forest of northern Ethiopia. Mengistu *et al.* (2005) also recorded higher numbers of species with 58 woody species belonging to 30 plant families at Biyo–Kelala enclosure site, central Ethiopia. A study by Giday *et al.* (2018) also found 63 woody plant species in Dessa'a forest, northern Ethiopia. The variation in woody species' richness might be due to the difference in area, level of degradation and variation in precipitation (dos Santos *et al.*, 2007; Hill, 1973; Linder, 2001).

In Wujig Mahgo Waren, 1,618 average tree density per ha was recorded in the dense forest. A similar pattern of results was observed in Mengistu *et al.* (2005)'s research who found 1,746 stems/ha in Biyo–Kelala closed area, central Ethiopia. Conversely, Aynekulu (2011) recorded lower tree stem density in Desa'a forest, northern Ethiopia.

Again, Aynekulu (2011) found a higher stem density in Hugumburda Afromontane forests as compared to the present study. Senbeta and Denich (2006) conducted a similar study in the southwest highlands of Ethiopia. They indicated that Harenna and Berhane-Kontir forest had higher stem densities as compared to those of the present study. The difference in the number of trees per ha across the various sites might be due to the difference in the level of disturbance and management of the forest.

5.4. Effects of land cover change on carbon stock

The study showed how carbon stocks in vegetation, litter and soils vary across land cover types and different periods. With this study, higher biomass carbon stock was recorded in dense forests compared to open forests and that of the grassland. In light of this present study, Rajput *et al.* (2017) and Solomon *et al.* (2017) found higher biomass carbons in forest ecosystems as compared to other land cover types in northwestern Himalaya and northern Ethiopia, respectively. The substantial variation in biomass carbon across the land cover types might be due to the variation in the number of stems and the size of the trees in each land cover type. This is in line with the result of research of Solomon *et al.* (2017) which stated that tree density and diameter have an effect on biomass carbon in northern Ethiopia. Moreover, this present study concludes that the low biomass carbon recorded in grasslands was an effect of overgrazing practices and human intrusion which influenced recovery and growth of herbaceous plant species adversely and smothers tree and shrub growth. This assertion is supported by the study conducted by Mekuria and Yami (2013) who suggested that free grazing affects vegetation composition and growth of herbaceous plant species in the lowlands of northern Ethiopia.

The biomass carbon estimates of the dense forests were within the global estimates range, from 20 to 150 Mg ha^{-1} for semiarid tropics as reported by Tiessen *et al.* (1998). Furthermore, the results were also within the range of tropical dry forests' carbon stock, with regards to the report of Cavanaugh *et al.* (2014), which was between 50 and 350 t ha^{-1}. However, the average biomass carbon stock of Wujig Mahgo Waren forest was lower than that observed for Egdu forest as reported by Feyissa *et al.* (2013) which was 337 t ha^{-1}. The biomass carbon stock of the present study was fairly small compared to the biomass carbon stocks in the moist Bale forest in Ethiopia, an observation made by Watson *et al.* (2013). On the other hand, the biomass carbon stock in the current study was fairly higher compared to Solomon *et al.* (2017) who reported 58.11 Mg ha^{-1} in the managed forest of Tigray, northern Ethiopia. As compared to the present study, Chinasho *et al.* (2015) found lower carbon stock with 45.23 t ha^{-1} in woody plants of Humbo forest, southern Ethiopia. The difference in biomass carbon stock across the different forests might be attributed to the variability in biophysical characteristics such as climate, soil and vegetation type.

Furthermore, the carbon content of litter biomass in this study was significantly higher under dense forests than grasslands. The difference in litter carbon among the land cover types might be due to the variations in vegetation cover. This was confirmed by the study of Descheemaeker *et al.* (2006) who stated that litter accumulation rely upon vegetation cover and is affected by soil fertility in exclosures of the Tigray highlands, Ethiopia. The estimated litter carbon of the present study is in accordance with findings reported by Ordóñez *et al.* (2008), who found between 0.6 and 4.1 Mg ha^{-1} of litter carbon in montane forests of central and southern Mexico. However, the estimated value of litter carbon in the present study was higher than that reported by Aman (2015) who found 1.38 t ha^{-1} litter carbon in dry evergreen montane forests of the Bale mountain national park, Ethiopia. Conversely, compared to the litter carbon stocks of Chilimo forest (9.36 Mg ha^{-1}) per Tesfaye *et al.* (2016a)'s observation, the current result was very low.

Furthermore, in the present study, higher soil organic carbon stock was recorded in grassland and dense forest as compared to open forest, bare land and cultivated land. Also, the differences recorded in soil organic carbon between land cover types were not significant. In agreement with the present study, Haghdoost *et al.* (2013) showed that no significant difference existed in the average total soil carbon stock among land cover types in Noor county, Iran, though higher soil carbon was found in forests as compared to cultivated lands. In line with this current study, Ordóñez *et al.* (2008) also found no significant difference in average total soil carbon in the central highlands of Michoacan, Mexico. The higher mean soil organic carbon stock in grassland compared with the other land uses could be due to higher annual turnover of organic matter from dying grassroots. This notion was supported by the report of Guo and Gifford (2002), who stated that grassroots decompose faster than tree roots and hence contribute higher organic matter to soils. The higher soil organic carbon stock recorded in the dense forest was mainly because of the biomass inputs and low rate of litter decay. In the same vein, Tesfaye *et al.* (2016a) also discovered a higher mean carbon stock in natural forest than in all the other land cover categories in Chilimo, a dry Afromontane forest in Ethiopia. The lower soil organic carbon recorded in the cultivated land under this present study might be due to the low input of organic matter being returned to the soil and high rates of oxidation of soil organic matter by tillage as reported by Dalal and Chan (2001), who studied soil organic matter in rain-fed cropping systems of the Australian cereal belt.

The high carbon content of the soils in the different land cover types was consistent with a previous study by Lemenih and Itanna (2004) who studied soil carbon stock for the upper 60 cm depth of soil in southern Ethiopia. The result of this study were also within the ranges of values for tropical soils of 86 Mg carbon ha^{-1} (Brown & Lugo, 1982), 113 Mg carbon ha^{-1} (Post *et al.*, 1982) and 72.8 to 116.4 Mg carbon ha^{-1} of montane forests of Central Highlands of Michoacan, Mexico (Ordóñez *et al.*, 2008). Contrary to the results of this current study, Feyissa *et al.* (2013) recorded higher soil organic carbon in Egdu Forest, Ethiopia. On the other hand, higher soil organic carbon stock was recorded under the present study as compared to the results reported by Girmay and Singh (2012) for Maileba and Gum Selassa sites of northern Ethiopia.

Land cover change is a major factor in soil carbon stock change. In the present study, the results indicated that alteration of dense forests to cultivated land brought about 25% reductions in soil organic carbon stock. Girmay *et al.* (2008) who reviewed carbon stock in top soils (0–10 cm) of Ethiopia, found that conversion of native forest into croplands and plantations reduced carbon stock by up to 63% and 83%, respectively, an observation that is in line with this current study.

Generally, higher total carbon stock was recorded in dense forest followed by open forest, grassland, cultivated land and bare land under this study. The average total carbon stock of the dense forest in the present study was 181.8 Mg ha^{-1}, which was a little higher than that reported by Mekuria (2013) for exclosures on communal grazing lands in Ethiopia. Similarly, the results were slightly higher than that reported by Andriamananjara *et al.* (2016) for the Malagasy rainforest in eastern Madagascar. On the other hand, the result of the present study was lower than that reported by Rajput *et al.* (2017) for northwestern Himalaya (303.39 Mg ha^{-1}), Feyissa *et al.* (2013) for

Egdu forest (614.72 ton ha^{-1}), Ordóñez *et al.* (2008) for montane forests of central and southern Mexico (220.7 to 266.9 Mg ha^{-1}) and Simegn *et al.* (2014) for low land area of Simien mountains national park (513.4 ton ha^{-1}). The variations in total carbon stock among the different studies might be due to variation in forest composition, soil and other biophysical factors.

In this study, the four carbon pools contributed differently to the five land cover classes. Higher levels of carbon were stored in the soil pool rather than the vegetation biomass and litter carbon of all land cover types. Most of the carbon stocks in grassland, cultivated land and bare land were mainly found in the soil. For example, in grassland, a large percentage (> 90%) of the total carbon was stored in the soil. This was in accordance with the investigation of Chen *et al.* (2003), where the total carbon stock of the savanna was 204 ± 53 Mg ha^{-1}, with around 84% below ground and 16% above ground carbon stock. According to Scurlock and Hall (1998), soil carbon can store over 75% of the global carbon found in terrestrial ecosystems. Birhane *et al.* (2017b) also found more carbon being stored in soils of *Prosopis juliflora* invaded areas. Mekuria (2013) also pointed to higher carbon stock in soil than other carbon pools for exclosures on communal grazing lands in Ethiopia. However, contrary to the findings of this present study, Girardin *et al.* (2010) and Lü *et al.* (2010) found higher carbon stored in biomass followed by soil and litter in tropical forests.

In the present study, the change in carbon stock caused by change in land cover type was also assessed using the area of each land cover type and their corresponding carbon stock values. The study revealed an increase in carbon stock between 1985 and 2000 and a decrease between 2000 and 2016. This can be attributed to the land cover change from forest to cultivated land and grassland hence a reduction in total carbon stock between 2000 and 2016. Forest land is basically a collection of native tree species that has been in existence for quite a long time with many understory vegetation. However, grassland is mainly composed of shrub species with low biomass and total carbon stock as compared to the forest. Consequently, the change from forest to grassland and cultivated land significantly affects total carbon stocks. Our study illustrated that total carbon stock was affected by the land cover change in Wujig Mahgo Waren forest.

In agreement with the present study, previous studies have shown that land cover change is a key factor in carbon stock changes. For example, Shrestha *et al.* (2010) observed a net gain in carbon stock in the larger parts of the mountain watershed in Nepal from 1976 to 1989, while a net loss was recorded in the period between 1989 and 2003. Kashaigili and Majaliwa (2010) also realized a reduction in carbon stock from the year 1980 to 2010 in two forests of Tanzania due to forest cover change. Similarly, Gond *et al.* (2016) also reported a 30% loss in carbon stock from 1984 to 2012 in wood-fuel supply basin of Kinshasa. Furthermore, Gaston *et al.* (1998) showcased a loss in above ground carbon stock by 6.6 Pg due to forest degradation in tropical Africa between 1980 and 1990. In the same period, (Gaston *et al.*, 1998) recorded 30 Tg loss of above ground carbon due to deforestation and degradation in Ethiopia. A study conducted in Tarkwa, a mining town in Ghana by Kumi-Boateng *et al.* (2015) also highlighted an average loss of 412.14 Gg carbon between 1986 and 2007 due to land cover change. A study by Zhang *et al.* (2015b) in China showed that carbon stock reduced by 60 Tg between 1995 and 2010 due to land cover change. In spite of all these, Nakakaawa *et al.* (2011) reported results that are contrary to the findings of this study. They recorded a negligible increase in forest carbon stocks (3,260 t carbon yr^{-1}) in the period 1990–2005 when compared to the loss due to deforestation and forest degradation (2.67 million ton carbon yr^{-1}) in Uganda.

5.5. Linkage between soil organic carbon stock and above ground vegetation properties

Various studies have shown that vegetation variability determines topsoil carbon variability in the Savanna and woodland ecosystems (Bird *et al.*, 2000; Rossi *et al.*, 2009; Wang *et al.*, 2009). In the present study, soil organic carbon was positively linked with above ground vegetation properties, showing that vegetation parameters do appear to be predictors of soil organic carbon stock. Moreover, above ground vegetation parameters such as tree density, DBH and height explained 59% of the variance in soil organic carbon. In a similar study by Li *et al.* (2010b) above ground vegetation parameters such as tree height, above ground biomass and tree density elucidated 74.9% of the variance in soil organic carbon in cold-temperate mountainous forests of Japan. Dar and Sundarapandian (2013) also indicated that above ground vegetation properties are common predictors that can be used to estimate soil organic carbon stock in complex mountainous forests across different spatial scales. Furthermore, Woollen *et al.* (2012) found the strongest correlation between soil carbon and large tree above ground carbon stocks with 24% of soil carbon variability explained by above ground carbon stock. A study by Kurgat *et al.* (2011) showed that vegetation cover explained 89% of the variability in soil organic carbon in the rangelands of northern Kenya. Similarly, a study by Liu *et al.* (2012) in the Qinghai–Tibetan Plateau China showed a significant correlation between above ground biomass and soil organic carbon. Contrary to this present study, Zhang *et al.* (2011) found that plant biomass, woody plant density and tree height did not emerge as significant predictor variables for soil organic carbon in the subalpine coniferous forest in Southwest China. Mathew *et al.* (2016) also found a poor correlation between soil organic carbon stock and above ground carbon in Mount Kilimanjaro, Tanzania. The inconsistency between these studies shows that environmental factors affecting the distributions of vegetation and soil carbon stocks are site-specific.

Findings from this study show that vegetation parameters can be valuable when predicting soil organic carbon stock in the dry Afromontane forests. This discovery is vital for estimating soil carbon stock, particularly in inaccessible landscapes, as above ground vegetation properties are moderately simple to assess and could even be quickly surveyed through remote sensing methods.

5.6. Effects land cover change has on ecosystem service values

The results of the present study show the amount of change that has occurred in the ecosystem service value due to land cover dynamics over the past three decades. The results reveal that total ecosystem service values increased between 1985 and 2000. This is attributed to the improvement of dense forest and open forest as result of the introduction of different interventions such as intensive soil and water conservation, exclosure establishment and community participation. These gave the forest a recovery time for which some improvements have been observed between 1985 and 2000.

Between 2000 and 2016, the total ecosystem service value was reduced due to degradation of dense forest and open forest in the study area. The loss of value of the ecosystem's services as a result of the loss of forest cover was also revealed in various studies (Gashaw *et al.*, 2018; Hu *et al.*, 2008; Kindu *et al.*, 2016; Li *et al.*, 2007; Tolessa *et al.*, 2017a; Tolessa *et al.*, 2017b). For example, Gashaw *et al.* (2018)

showed a US$ 5.83 million reduction in ecosystem service value as a result of forest loss between 1985 and 2015 in Andassa watershed in the Upper Blue Nile basin of Ethiopia. Similarly, a study by Kindu *et al.* (2016) also reported a decline in ecosystem service from US$ 130.5 million in 1973 to US$ 111.1 million in 2012 due to the loss of natural forest in the Munessa-Shashemene landscape of the Ethiopian highlands.

Between 1985 and 2016, the total estimated ecosystem service values showed a slight increase resulting from the increase in open forest, grassland and cultivated land. In a similar study by Wang *et al.* (2014), ecosystem service value increased from US$ 1.82 billion to 2.24 billion between 2000 and 2010 in Ningxia of China, which was ascribed to an increase in of forest and grasslands. Camacho-Valdez *et al.* (2014) reported that total ecosystem service value increased by approximately 9% between 2000 and 2010 in the southern coast of Sinaloa State, Northwest of Mexico. Contrary to the present study, in Chongming Island, China, Zhao *et al.* (2004) estimated a decline in ecosystem service value by 62% between 1990 and 2000. Li *et al.* (2010a) also showed around 5% decrease in ecosystem service value from 1975 to 2005 in Zoige Plateau, China. Tolessa *et al.* (2017a) reportedly suggested a reduction in the overall ecosystem services value by 68% between 1973 and 2014 in the central highlands of Ethiopia, and this was mainly due to deforestation.

Moreover, the contribution of each land cover type to the ecosystem service value differed over the study periods. In both study periods, the main ecosystem provider was the forest ecosystem. Kindu *et al.* (2016) also found higher ecosystem services in natural forest of the Munessa-Shashemene landscape of the Ethiopian highlands. As compared to the forest ecosystem, the contribution of grassland and cultivated land were very insignificant in the present study.

The average ecosystem service value of the study area increased from 976.5 US\$ ha^{-1} yr^{-1} in 1985 to 1,117.6 US\$ ha^{-1} yr^{-1} in 2000 and decreased to 1,064.7 ha^{-1} yr^{-1} in 2016. The average ecosystem service value recorded in the present study was similar to the average ecosystem service value recorded by Kindu *et al.* (2016) in the Munessa-Shashemene landscape, Ethiopia. However, a higher average ecosystem service value was recorded as compared to other findings such as in Toke Kutaye district (Tolessa *et al.*, 2017a), Chillimo forest (Tolessa *et al.*, 2017b), Andassa watershed in the Upper Blue Nile basin of Ethiopia (Gashaw *et al.*, 2018), Lao PDR (Yoshida *et al.*, 2010) and San Antonio area, Texas (Kreuter *et al.*, 2001). On the other hand, a lower average ecosystem service value was recorded in this study in comparison with the values reported by Hu *et al.* (2008) and Wang *et al.* (2015) in Xishuangbanna, Southwest China and Nenjiang River Basin, northeast China, respectively. The discrepancies among these studies most likely resulted from differences in the size of the dominant land cover types and the ecosystem service coefficients used.

Between 2000 and 2016, values of water regulation, recreation, climate regulation, genetic resources and erosion control services decreased as a result of the decline in dense forest and open forest in the study area. This was in agreement with the study conducted by Wang *et al.* (2017), who observed a reduction in the value of gas regulation, climate regulation and other ecosystem services due to a decline in area of grasslands, woodlands and aquatic regions between 2003 and 2013 periods in Manas river basin, China. In a similar study by Kindu *et al.* (2016), water regulation, recreation, climate regulation, genetic resources and erosion control services decreased between 1973 and 2012 because of the loss of natural forests, woodlands, shrubland and grasslands in the Munessa-Shashemene landscape of the Ethiopian highlands. Gashaw *et al.* (2018) also showed a reduction in several individual ecosystem service values between 1985 and 2015 due to forest and shrubland degradation in the Andassa watershed of Ethiopia.

On the other hand, food production service value increased between 2000 and 2016, which was mainly the result of an increase in the cultivated land within the study area. Similarly, Kindu *et al.* (2016) in a study of the Munessa-Shashemene landscape of the Ethiopian highlands, reported that food production services increased between 1973 and 2012 while other ecosystem services such as disturbance regulation, cultural and recreational ecosystem services decreased due to the expansion of cultivated land. Tolessa *et al.* (2017a) also found an improvement in food production services while other ecosystem services such as raw material production, recreation, and cultural services declined in the central highlands of Ethiopia. Gashaw *et al.* (2018) also observed an increase in food production, biological control and pollination in the Andassa watershed, Ethiopia.

CHAPTER SIX

6. CONCLUSION AND RECOMMENDATION

Based on the objective set for the study, the following conclusions are drawn.

6.1. Conclusion

The study area underwent extensive land cover changes over the past 30 years with a rapid transformation from dense and open forest to cultivated land and grassland. Fuelwood collection and expansion of cultivated land were ranked as the key factors responsible for the changes in land cover. Improving forest conservation, soil and water conservation structures, community awareness, enrichment planting, financing for added ecosystem services and the introduction of zero grazing mechanisms are among possible solutions forwarded to reverse the undesired situation in the study area. The carbon stock of the forest is in the range of similar ecosystems elsewhere, where the highest carbon stock was locked in dense forest. The study indicated that the dry Afromontane forest has the potential to sequester large amounts of carbon in its biomass and soil. Carbon was significantly higher in the soil than those in biomass and litter carbon pools in all land cover types. The amount of carbon stock varied in the two study periods following land cover dynamics pattern. Above ground vegetation properties were highly correlated to soil organic carbon indicated that soil organic carbon can be estimated using above ground vegetation parameters in the dry Afromontane forests. Land cover dynamics especially forest cover change dictated gain and loss of ecosystem service values. Generally, this study has shown that land cover change affects carbon sequestration potential of forests and ecosystem services values. Furthermore, to estimate the change in carbon stocks and ecosystem service values remotely sensed data can be an important tool in developing country such as Ethiopia which has not a long-term data.

6.2. Recommendation

Based on the study the following points are recommended:

- An option of forest management that addresses the needs of the community and avoids pressure on the forest has to be studied critically to sustain the forest and its ecosystem services.
- Forest experts must educate local communities on sustainable uses of forest resources to reduce the pressure on the forest.
- Alternate energy sources must be provided by the government to reduce firewood collection from the forest.
- Zero grazing concept to reduce the impact of grazing on the forest has to be introduced by the government.
- The Wujig-Mahgo-waren forest should be highly considered in the carbon trade by REDD+ as it serves a high carbon reservoir.
- Aboveground vegetation parameters can be used by researchers when estimating soil organic carbon stock in dry Afromontane forests.
- Policy makers should consider the importance of ecosystem services during policy formulation and in various intervention strategies.
- Further studies should be conducted on the effect of other biophysical factors on carbon stock.
- It will be worthwhile to study on prediction and modelling of the change in ecosystem service value to determine the extent of change in the future, to maintain the provision of ecosystem goods and services.

REFERENCES

Abebe, F. B., & Bekele, S. E. (2018). Challenges to national park conservation and management in Ethiopia. *Journal of Agricultural Science, 10*(5). doi: 10.5539/jas.v10n5p52

Abebe, M. H., Oba, G., Angassa, A., & Weladji, R. B. (2006). The role of area enclosures and fallow age in the restoration of plant diversity in northern Ethiopia. *African Journal of Ecology, 44*(4), 507-514.

Abelson, P. H. (1990). Medicine from plants. *Science, 247*(4942), 513-514.

Abrha, M. G. (2015). Local climate trends and farmers' perceptions in southern Tigray, northern Ethiopia. *International Journal of Environment and Sustainability, 4*(3).

Achard, F., Eva, H. D., Stibig, H.-J., Mayaux, P., Gallego, J., Richards, T., & Malingreau, J.-P. (2002). Determination of deforestation rates of the world's humid Tropical forests. *Science, 297*(5583), 999-1002. doi: 10.1126/science.1070656

Adefires, W. (2006). *Population status and socio-economic importance of gum and resin bearing species in Borana Lowlands, southern Ethiopia.* (Master of Science), Addis Ababa University, Addis Ababa, Ethiopia.

Aerts, R., Van Overtveld, K., Haile, M., Hermy, M., Deckers, J., & Muys, B. (2006). Species composition and diversity of small Afromontane forest fragments in northern Ethiopia. *Plant Ecology, 187*(1), 127-142. doi: 10.1007/s11258-006-9137-0

Aerts, R., Van Overtveld, K., November, E., Wassie, A., Abiyu, A., Demissew, S., . . . Healey, J. R. (2016). Conservation of the Ethiopian church forests: Threats, opportunities and implications for their management. *Science of the Total Environment, 551-552*, 404-414. doi: https://doi.org/10.1016/j.scitotenv.2016.02.034

Agrawal, A., Cashore, B., Hardin, R., Shepherd, G., Benson, C., & Miller, D. (2013). *Economic contributions of forests*. Istanbul, Turkey

Alexandre, K., & Philippe, A. (2010). State and trends of the carbon market 2010. *Washington DC: World Bank*, 78.

Alloisio, I., Calliari, E., Conci, S., D'Aprile, A., Davide, M., Johnson, K., & Rossi, R. (2015). COP21 and the Clean Development Mechanism: deciding the future of international carbon credits. *International Center for Climate Governance (ICCG)*.

Aman, H. (2015). *Estimation of carbon stock in dry evergreen montane forest and its role in climate change mitigation: The case of Bale mountains national park* (Master of Science), Addis Ababa University Ethiopia

Amanuel, Z., Girmay, G., & Atkilt, G. (2015). *Characterisation of agricultural soils in Cascape intervention woredas in southern Tigray, Ethiopia*. Mekelle, Ethiopia:

Andriamananjara, A., Hewson, J., Razakamanarivo, H., Andrisoa, R. H., Ranaivoson, N., Ramboatiana, N., . . . Razafimbelo, T. (2016). Land cover impacts on aboveground and soil carbon stocks in Malagasy rainforest. *Agriculture, Ecosystems & Environment, 233*(Supplement C), 1-15. doi: https://doi.org/10.1016/j.agee.2016.08.030

Asfaw, A., Lemenih, M., Kassa, H., & Ewnetu, Z. (2013). Importance, determinants and gender dimensions of forest income in eastern highlands of Ethiopia: The case of communities around Jelo Afromontane forest. *Forest Policy and Economics, 28*, 1-7. doi: https://doi.org/10.1016/j.forpol.2013.01.005

Asner, G. P., Powell, G. V., Mascaro, J., Knapp, D. E., Clark, J. K., Jacobson, J., . . . Victoria, E. (2010). High-resolution forest carbon stocks and emissions in the Amazon. *Proceedings of the National Academy of Sciences, 107*(38), 16738-16742.

Ayenew, M. (2014). *The Growth and Transformation Plan: Opportunities, challenges and lessons*. Paper presented at the Reflections on development in Ethiopia. New trends, sustainability and challenges, forum for social studies Friedrich Ebert Stiftung, Addis-Abeba, Ethiopia

Aynekulu, E. (2011). *Forest diversity in fragmented landscapes of northern Ethiopia and implications for conservation*. (Ph.D dissertation), Rheinischen Friedrich-Wilhelms-Universität Bonn, Bonn, Germany.

Aynekulu, E., Denich, M., Tsegaye, D., Aerts, R., Neuwirth, B., & Boehmer, H. J. (2011). Dieback affects forest structure in a dry Afromontane forest in northern Ethiopia. *Journal of Arid Environments, 75*(5), 499-503. doi: https://doi.org/10.1016/j.jaridenv.2010.12.013

Babulo, B. (2007). *Economic valuation and management of common-pool resources: The case of exclosures in the highlands of Tigray, northern Ethiopia.* (Ph. D. dissertation), Katholieke Universiteit, Leuven: Belgium.

Babulo, B., Muys, B., Nega, F., Tollens, E., Nyssen, J., Deckers, J., & Mathijs, E. (2009). The economic contribution of forest resource use to rural livelihoods in Tigray, northern Ethiopia. *Forest Policy and Economics, 11*(2), 109-117. doi: https://doi.org/10.1016/j.forpol.2008.10.007

Baccini, A., Goetz, S. J., Walker, W. S., Laporte, N. T., Sun, M., Sulla-Menashe, D., . . . Houghton, R. A. (2012). Estimated carbon dioxide emissions from tropical deforestation improved by carbon-density maps. *Nature Climate Change, 2*, 182. doi: 10.1038/nclimate1354 https://www.nature.com/articles/nclimate1354#supplementary-information

Baldo-Soriano, E., Joan, C., Raymond, d. C., Christian, E., Francesco, M., & Helen, T. (2012). *What is REDD+? A guide for indigenous communities - 3rd edition.* Chiang Mai, Thailand: Asia Indigenous Peoples' Pact (AIPP) and International Work Group for Indigenous Affairs (IWGIA).

Balick, M. J., Elisabetsky, E., & Laird, S. A. (1996). *Medicinal resources of the tropical forest : biodiversity and its importance to human health.* New York, USA: Columbia University Press.

Balick, M. J., & Mendelsohn, R. (1992). Assessing the economic value of traditional medicines from tropical rain forests. *Conservation Biology, 6*(1), 128-130.

Ball, D. F. (1964). Loss-on-Ignition as an estimate of organic matter and organic carbon in non-calcareous soils. *Journal of Soil Science, 15*(1), 84-92. doi: 10.1111/j.1365-2389.1964.tb00247.x

Balvanera, P., Pfisterer, A. B., Buchmann, N., He, J.-S., Nakashizuka, T., Raffaelli, D., & Schmid, B. (2006). Quantifying the evidence for biodiversity effects on ecosystem functioning and services. *Ecology Letters, 9*(10), 1146-1156. doi: 10.1111/j.1461-0248.2006.00963.x

Bathurst, J. C., Iroumé, A., Cisneros, F., Fallas, J., Iturraspe, R., Novillo, M. G., . . . Ramírez, M. (2011). Forest impact on floods due to extreme rainfall and snowmelt in four Latin American environments 1: Field data analysis. *Journal of Hydrology, 400*(3), 281-291. doi: https://doi.org/10.1016/j.jhydrol.2010.11.044

Beaudrot, L., Ahumada, J. A., O'Brien, T., Alvarez-Loayza, P., Boekee, K., Campos-Arceiz, A., . . . Andelman, S. J. (2016). Standardized assessment of biodiversity trends in tropical forest protected areas: The end Is not in sight. *PLoS Biology, 14*(1), e1002357. doi: 10.1371/journal.pbio.1002357

Beckett, K. P., Freer-Smith, P. H., & Taylor, G. (1998). Urban woodlands: their role in reducing the effects of particulate pollution. *Environmental Pollution, 99*(3), 347-360. doi: https://doi.org/10.1016/S0269-7491(98)00016-5

Beilin, R., Lindborg, R., Stenseke, M., Pereira, H. M., Llausàs, A., Slätmo, E., . . . Queiroz, C. (2014). Analysing how drivers of agricultural land abandonment affect biodiversity and cultural landscapes using case studies from Scandinavia, Iberia and Oceania. *Land Use Policy, 36*(Supplement C), 60-72. doi: https://doi.org/10.1016/j.landusepol.2013.07.003

Bekele, M. (2003). *Forest property rights, the role of the state, and intitutional exigency* (Vol. 409). Uppsala: Sveriges lantbruksuniv., Acta Universitatis agriculturae Sueciae.

Bekele, M., Tesfaye, Y., Mohammed, Z., Zewdie, S., Tebikew, Y., Brockhaus, M., & Kassa, H. (2015). *The context of REDD+ in Ethiopia: Drivers, agents and institutions* (Vol. 127). Bogor, Indonesia: Center for International Forestry Research (CIFOR).

Ben-Dor, E., & Banin, A. (1989). Determination of organic matter content in arid-zone soils using a simple “loss-on-ignition” method. *Communications in Soil Science and Plant Analysis, 20*(15-16), 1675-1695. doi: 10.1080/00103628909368175

Bernard, E., & Fenton, M. B. (2002). Species diversity of bats (Mammalia: Chiroptera) in forest fragments, primary forests, and savannas in central Amazonia, Brazil. *Canadian Journal of Zoology, 80*(6), 1124-1140. doi: 10.1139/z02-094

Bewket, W. (2002). Land cover dynamics since the 1950s in Chemoga watershed, Blue Nile Basin, Ethiopia. *Mountain Research and Development, 22*(3), 263-269. doi: 10.1659/0276-4741(2002)022[0263:LCDSTI]2.0.CO;2

Bhattacharjee, K., & Behera, B. (2017). Forest cover change and flood hazards in India. *Land Use Policy, 67*, 436-448. doi: https://doi.org/10.1016/j.landusepol.2017.06.013

Bhatti, J., Lal, R., Apps, M. J., & Price, M. A. (2005). *Climate change and managed ecosystems*: CRC Press.

Binkley, D., & Fisher, R. F. (2012). *Ecology and management of forest soils*. Hoboken, NJ: Wiley, Blackwell.

Bird, M. I., Veenendaal, E. M., Moyo, C., Lloyd, J., & Frost, P. (2000). Effect of fire and soil texture on soil carbon in a sub-humid savanna (Matopos, Zimbabwe). *Geoderma, 94*(1), 71-90. doi: https://doi.org/10.1016/S0016-7061(99)00084-1

Birhane, E., Mengistu, T., Seyoum, Y., Hagazi, N., Putzel, L., Rannestad, M. M., & Kassa, H. (2017a). Exclosures as forest and landscape restoration tools: lessons from Tigray Region, Ethiopia. *International Forestry Review, 19*(4), 37-50.

Birhane, E., Treydte, A. C., Eshete, A., Solomon, N., & Hailemariam, M. (2017b). Can rangelands gain from bush encroachment? Carbon stocks of communal grazing lands invaded by Prosopis juliflora. *Journal of Arid Environments, 141*, 60-67. doi: http://dx.doi.org/10.1016/j.jaridenv.2017.01.003

Bishaw, B. (2001). Deforestation and land degredation in the Ethiopian highlands: a strategy for physical recovery. *Northeast African Studies, 8*(1), 7-25. doi: 10.1353/nas.2005.0014

Bonan, G. B. (2008). Forests and climate change: Forcings, feedbacks, and the climate benefits of forests. *Science, 320*(5882), 1444-1449. doi: 10.1126/science.1155121

Bongers, F., Wassie, A., Sterck, F. J., Bekele, T., & Teketay, D. (2006). Ecological restoration and church forests in northern Ethiopia. *Journal of the Drylands, 1*(1), 35-44.

Boudreau, J., Nelson, R. F., Margolis, H. A., Beaudoin, A., Guindon, L., & Kimes, D. S. (2008). Regional aboveground forest biomass using airborne and spaceborne LiDAR in Québec. *Remote Sensing of Environment, 112*(10), 3876-3890. doi: https://doi.org/10.1016/j.rse.2008.06.003

Bray, D. B., Merino-Pérez, L., & Barry, D. (2009). *The community forests of Mexico: Managing for sustainable landscapes*. Austin, TX: University of Texas Press.

Brookhuis, B. J., & Hein, L. G. (2016). The value of the flood control service of tropical forests: A case study for Trinidad. *Forest Policy and Economics, 62*, 118-124. doi: https://doi.org/10.1016/j.forpol.2015.10.002

Brown, S. (1997). *Estimating biomass and biomass change of tropical forests : a primer*. Rome: Food and Agriculture Organization of the United Nations.

Brown, S., & Lugo, A. E. (1982). The storage and production of organic matter in tropical forests and their role in the global carbon cycle. *Biotropica, 14*(3), 161-187. doi: 10.2307/2388024

Bruggeman, D., Meyfroidt, P., & Lambin, E. F. (2016). Forest cover changes in Bhutan: Revisiting the forest transition. *Applied Geography, 67*, 49-66. doi: https://doi.org/10.1016/j.apgeog.2015.11.019

Bruijnzeel, L. A. (2004). Hydrological functions of tropical forests: not seeing the soil for the trees? *Agriculture, Ecosystems & Environment, 104*(1), 185-228. doi: https://doi.org/10.1016/j.agee.2004.01.015

Burgess, N. D., Mwasumbi, L. B., Hawthorne, W. J., Dickinson, A., & Doggett, R. A. (1992). Preliminary assessment of the distribution, status and biological importance of coastal forests in Tanzania. *Biological Conservation, 62*(3), 205-218. doi: https://doi.org/10.1016/0006-3207(92)91048-W

Butler, R. A., Koh, L. P., & Ghazoul, J. (2009). REDD in the red: palm oil could undermine carbon payment schemes. *Conservation Letters, 2*(2), 67-73. doi: 10.1111/j.1755-263X.2009.00047.x

Byers, S. C., Mills, E. L., & Stewart, P. L. (1978). A comparison of methods of determining organic carbon in marine sediments, with suggestions for a standard method. *Hydrobiologia, 58*(1), 43-47. doi: 10.1007/bf00018894

Cairns, A. M., Brown, S., Helmer, H. E., & Baumgardner, A. G. (1977). Root biomass allocation in the world's upland forests. *Oecologia, 111*(1), 1-11. doi: 10.1007/s004420050201

Calder, I., Hofer, T., Vermont, S., & Warren, P. (2007). Towards a new understanding of forests and water. *Unasylva, 58*(229), 3-10.

Calvas, B., Knoke, T., Castro, L. M., Hildebrandt, P., Weber, M., Stimm, B., . . . Aguirre, N. (2013). Sustainable Agriculture and Conservation Payments Are Key Factors in Mitigating Tropical Forest Loss. In J. Bendix, E. Beck, A. Bräuning, F. Makeschin, R. Mosandl, S. Scheu, & W. Wilcke (Eds.), *Ecosystem Services, Biodiversity and Environmental Change in a Tropical Mountain Ecosystem of South Ecuador* (pp. 235-244). Berlin, Heidelberg: Springer Berlin Heidelberg.

Camacho-Valdez, V., Ruiz-Luna, A., Ghermandi, A., Berlanga-Robles, C. A., & Nunes, P. A. L. D. (2014). Effects of land use changes on the ecosystem service values of Coastal Wetlands. *Environmental Management, 54*(4), 852-864. doi: 10.1007/s00267-014-0332-9

Campbell, D. J. (1990). Strategies for coping with severe food deficits in rural Africa: A review of the literature. *Food and Foodways, 4*(2), 143-162. doi: 10.1080/07409710.1990.9961976

Campos, J., & Corrales, O. (2005). *The role of model forests for managing ecosystem services at landscape-scales.* Paper presented at the IV International Wallace Conference.

Cavanaugh, K. C., Gosnell, J. S., Davis, S. L., Ahumada, J., Boundja, P., Clark, D. B., . . . Andelman, S. (2014). Carbon storage in tropical forests correlates with taxonomic diversity and functional dominance on a global scale. *Global Ecology and Biogeography, 23*(5), 563-573. doi: 10.1111/geb.12143

CBD. (2009). *Sustainable Forest Management, Biodiversity and Livelihoods: A Good Practice Guide*. Montreal:

Chakravarty, S., Ghosh, S., Suresh, C., Dey, A., & Shukla, G. (2012). Deforestation: causes, effects and control strategies. In *Global perspectives on sustainable forest management*: InTech.

Chao, S. (2012). *Forest peoples: Numbers across the world*. Stratford Road Moreton-in-Marsh, United Kingdom: Forest Peoples Programme Moreton-in-Marsh.

Chaturvedi, R. K., Raghubanshi, A. S., & Singh, J. S. (2011). Carbon density and accumulation in woody species of tropical dry forest in India. *Forest Ecology and Management, 262*(8), 1576-1588. doi: https://doi.org/10.1016/j.foreco.2011.07.006

Chave, J., Andalo, C., Brown, S., Cairns, M. A., Chambers, J. Q., Eamus, D., . . . Yamakura, T. (2005). Tree allometry and improved estimation of carbon stocks and balance in tropical forests. *Oecologia, 145*(1), 87-99. doi: 10.1007/s00442-005-0100-x

Chave, J., Réjou-Méchain, M., Búrquez, A., Chidumayo, E., Colgan, M. S., Delitti, W. B. C., . . . Vieilledent, G. (2014). Improved allometric models to estimate the aboveground biomass of tropical trees. *Global Change Biology, 20*(10), 3177-3190. doi: 10.1111/gcb.12629

Chen, X., Hutley, L. B., & Eamus, D. (2003). Carbon balance of a tropical savanna of northern Australia. *Oecologia, 137*(3), 405-416.

Chinasho, A., Soromessa, T., & Bayable, E. (2015). Carbon stock in woody plants of Humbo forest and its variation along altitudinal gradients: The case of Humbo district, Wolaita zone, southern Ethiopia. *International Journal of Environmental Protection and Policy, 3*(4), 97-103.

Chomitz, K. M., & Buys, P. (2007). At loggerheads? agricultural expansion, poverty reduction, and environment in the tropical forests.

Ciais, P., Piao, S.-L., Cadule, P., Friedlingstein, P., & Chédin, A. (2009). Variability and recent trends in the African terrestrial carbon balance. *Biogeosciences, 6*(9), 1935-1948. doi: https://doi.org/10.5194/bg-6-1935-2009

Cleemput, S., Muys, B., Kleinn, C., & Janssens, M. J. (2004). Biomass estimation techniques for enclosures in a semi-arid area: a case study in northern Ethiopia. *University of Göttingen, Germany [www document]. URL http://www. tropentag. de/2004/abstracts/full/3. pdf.*

Conyers, M. K., Poile, G. J., Oates, A. A., Waters, D., & Chan, K. Y. (2011). Comparison of three carbon determination methods on naturally occurring substrates and the implication for the quantification of 'soil carbon'. *Soil Research, 49*(1), 27-33. doi: https://doi.org/10.1071/SR10103

Costanza, R., d'Arge, R., De Groot, R., Faber, S., Grasso, M., Hannon, B., . . . Paruelo, J. (1997). The value of the world's ecosystem services and natural capital. *Nature, 387*.

Costanza, R., de Groot, R., Sutton, P., van der Ploeg, S., Anderson, S. J., Kubiszewski, I., . . . Turner, R. K. (2014). Changes in the global value of ecosystem services. *Global Environmental Change, 26*(Supplement C), 152-158. doi: https://doi.org/10.1016/j.gloenvcha.2014.04.002

Costenbader, J. (2011). *REDD+ benefit sharing: a comparative assessment of three national policy approaches*.

Craft, C. B., Seneca, E. D., & Broome, S. W. (1991). Loss on ignition and kjeldahl digestion for estimating organic carbon and total nitrogen in estuarine marsh soils: Calibration with dry combustion. *Estuaries, 14*(2), 175-179. doi: 10.2307/1351691

CRGE. (2011). *Ethiopia's climate-resilient green economy strategy*. Addis Ababa, Ethiopia.

CSA. (2013). *Population projection of Ethiopia for all regions at wereda level from 2014 – 2017*. Addis Ababa, Ethiopia: Central Statistical Agency.

D'Almeida, C., Vörösmarty, C. J., Hurtt, G. C., Marengo, J. A., Dingman, S. L., & Keim, B. D. (2007). The effects of deforestation on the hydrological cycle in Amazonia: a review on scale and resolution. *International Journal of Climatology, 27*(5), 633-647. doi: 10.1002/joc.1475

Daily, G. C. (1997). *Nature's services : societal dependence on natural ecosystems*. Washington, DC [u.a.]: Island Press.

Dalal-Clayton, B., & Sadler, B. (2014). *Sustainability appraisal: a sourcebook and reference guide to international experience*: Routledge.

Dalal, R. C., & Chan, K. Y. (2001). Soil organic matter in rainfed cropping systems of the Australian cereal belt. *Soil Research, 39*(3), 435-464. doi: https://doi.org/10.1071/SR99042

Dar, J., & Sundarapandian, S. (2013). Soil organic carbon stock assessment in two temperate forest types of western Himalaya of Jammu and Kashmir, India. *Forest Research, 3*(114), 2. doi: http://dx.doi.org/10.4172/2168-9776.1000114

Darbyshire, I., Lamb, H., & Umer, M. (2003). Forest clearance and regrowth in northern Ethiopia during the last 3000 years. *The Holocene, 13*(4), 537-546. doi: 10.1191/0959683603hl644rp

David, W., & Roni, A. (2002). The local and global effects of Amazon deforestation. *Journal of Geophysical Research: Atmospheres, 107*(D20), LBA 55-51-LBA 55-58. doi: doi:10.1029/2001JD000717

Davidson, J. (1989). *The Eucalyptus Dilemma, Arguments for and Against Eucalyptus Plantation in Ethiopia*, Addis Ababa, Ethiopia.

de Groot, R., Brander, L., van der Ploeg, S., Costanza, R., Bernard, F., Braat, L., . . . van Beukering, P. (2012). Global estimates of the value of ecosystems and their services in monetary units. *Ecosystem Services, 1*(1), 50-61. doi: https://doi.org/10.1016/j.ecoser.2012.07.005

de Groot, R. S., Wilson, M. A., & Boumans, R. M. J. (2002). A typology for the classification, description and valuation of ecosystem functions, goods and services. *Ecological Economics, 41*(3), 393-408. doi: http://dx.doi.org/10.1016/S0921-8009(02)00089-7

De Vos, B., Lettens, S., Muys, B., & Deckers, J. A. (2007). Walkley–Black analysis of forest soil organic carbon: recovery, limitations and uncertainty. *Soil Use and Management, 23*(3), 221-229. doi: 10.1111/j.1475-2743.2007.00084.x

DeFries, R. S., Rudel, T., Uriarte, M., & Hansen, M. (2010). Deforestation driven by urban population growth and agricultural trade in the twenty-first century. *Nature Geoscience, 3*, 178. doi: 10.1038/ngeo756https://www.nature.com/articles/ngeo756#supplementary-information

Demissie, F., Yeshitila, K., Kindu, M., & Schneider, T. (2017). Land use/Land cover changes and their causes in Libokemkem District of South Gonder, Ethiopia. *Remote Sensing Applications: Society and Environment, 8*(Supplement C), 224-230. doi: https://doi.org/10.1016/j.rsase.2017.10.001

Deng, J., Arano, K. G., Pierskalla, C., & McNeel, J. (2010). Linking urban forests and urban tourism: A case of Savannah, Georgia. *Tourism Analysis, 15*(2), 167-181. doi: 10.3727/108354210X12724863327641

Descheemaeker, K., Muys, B., Nyssen, J., Poesen, J., Raes, D., Haile, M., & Deckers, J. (2006). Litter production and organic matter accumulation in exclosures of the Tigray highlands, Ethiopia. *Forest Ecology and Management, 233*(1), 21-35. doi: https://doi.org/10.1016/j.foreco.2006.05.061

Dessie, G., & Christiansson, C. (2008). Forest decline and its causes in the south-central Rift Valley of Ethiopia: Human impact over a one hundred year perspective. *AMBIO: A Journal of the Human Environment, 37*(4), 263-271. doi: 10.1579/0044-7447(2008)37[263:FDAICI]2.0.CO;2

Dessie, G., & Kleman, J. (2007). Pattern and magnitude of deforestation in the south central Rift Valley region of Ethiopia. *Mountain Research and Development, 27*(2), 162-168. doi: 10.1659/mrd.0730

Dewan, A. M., Yamaguchi, Y., & Ziaur Rahman, M. (2012). Dynamics of land use/cover changes and the analysis of landscape fragmentation in Dhaka Metropolitan, Bangladesh. *GeoJournal, 77*(3), 315-330. doi: 10.1007/s10708-010-9399-x

Domke, G. M., Perry, C. H., Walters, B. F., Woodall, C. W., Russell, M. B., & Smith, J. E. (2016). Estimating litter carbon stocks on forest land in the United States. *Science of the Total Environment, 557-558*, 469-478. doi: https://doi.org/10.1016/j.scitotenv.2016.03.090

Donato, D. C., Kauffman, J. B., Murdiyarso, D., Kurnianto, S., Stidham, M., & Kanninen, M. (2011). Mangroves among the most carbon-rich forests in the tropics. *Nature Geoscience, 4*, 293. doi: 10.1038/ngeo1123https://www.nature.com/articles/ngeo112#supplementary-information

dos Santos, K., Kinoshita, L. S., & Santos, F. A. M. d. (2007). Tree species composition and similarity in semideciduous forest fragments of southeastern Brazil. *Biological Conservation, 135*(2), 268-277. doi: https://doi.org/10.1016/j.biocon.2006.10.027

Drake, J. B., Knox, R. G., Dubayah, R. O., Clark, D. B., Condit, R., Blair, J. B., & Hofton, M. (2003). Above-ground biomass estimation in closed canopy Neotropical forests using lidar remote sensing: factors affecting the generality of relationships. *Global Ecology and Biogeography, 12*(2), 147-159. doi: 10.1046/j.1466-822X.2003.00010.x

Du, H., Zhou, G., Fan, W., Ge, H., Xu, X., Shi, Y., & Fan, W. (2010). Spatial heterogeneity and carbon contribution of aboveground biomass of moso bamboo by using geostatistical theory. *Plant Ecology, 207*(1), 131-139. doi: 10.1007/s11258-009-9659-3

Duivenvoorden, J. E. (1995). Tree species composition and rain forest-environment relationships in the middle Caquetá area, Colombia, NW Amazonia. *Vegetatio, 120*(2), 91-113. doi: 10.1007/bf00034341

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

Ejigu, K., Gebey, T., & Preston, T. (2009). Constraints and prospects for apiculture research and development in Amhara region, Ethiopia. *Livestock Research for Rural Development, 21*(10), 172.

Ellison, D., Futter, M. N., & Bishop, K. (2012). On the forest cover–water yield debate: from demand- to supply-side thinking. *Global Change Biology, 18*(3), 806-820. doi: doi:10.1111/j.1365-2486.2011.02589.x

Ellison, D., Morris, C. E., Locatelli, B., Sheil, D., Cohen, J., Murdiyarso, D., . . . Sullivan, C. A. (2017). Trees, forests and water: Cool insights for a hot world. *Global Environmental Change, 43*, 51-61. doi: https://doi.org/10.1016/j.gloenvcha.2017.01.002

Enbakom, H. W., Feyssa, D. H., & Takele, S. (2017). Impacts of deforestation on the livelihood of smallholder farmers in Arba Minch Zuria Woreda, southern Ethiopia. *African Journal of Agricultural Research, 12*(15), 1293-1305. doi: 10.5897/AJAR2015.10123

Engida, T. G., & Teshoma, A. J. (2012). The socio economic effects of community forest management: Evidence from Dendi District, Ethiopia. *International Affairs and Global Strategy, 4*(2224-8951), 15-22.

Ensermu, K., & Abenet, G. (2011). *Multiple roles of forests in Ethiopia vs associated challenges : maximizing benefits while curbing limitations.* Paper presented at the Forum for Environment, Addis Ababa, Ethiopia.

Escobedo, F. J., & Nowak, D. J. (2009). Spatial heterogeneity and air pollution removal by an urban forest. *Landscape and Urban Planning, 90*(3), 102-110. doi: https://doi.org/10.1016/j.landurbplan.2008.10.021

Eshete, A., Teketay, D., & Hulten, H. (2005). The socio-economic importance and status of populations of Boswellia papyrifera (del.) Hochst. In northern Ethiopia: the case of north Gonder zone. *Forests, Trees and Livelihoods, 15*(1), 55-74. doi: 10.1080/14728028.2005.9752507

Eshetu, Z., & Högberg, P. (2000). Reconstruction of forest site history in Ethiopian highlands based on 13C natural abundance of soils. *AMBIO: A Journal of the Human Environment, 29*(2), 83-89. doi: 10.1579/0044-7447-29.2.83

Estoque, R. C., & Murayama, Y. (2013). Landscape pattern and ecosystem service value changes: Implications for environmental sustainability planning for the rapidly urbanizing summer capital of the Philippines. *Landscape and Urban Planning, 116*(Supplement C), 60-72. doi: https://doi.org/10.1016/j.landurbplan.2013.04.008

FAO. (2005a). *Global Forest Resources Assessment 2005: Progress towards sustainable forest management*. Rome, Italy: www.fao.org/docrep/008/a0400e/a0400e00.HTM

FAO. (2005b). *The State of the World's Forests*. Rome, Italy: www.fao.org/3/a-y5574e.pdf

FAO. (2010a). *Global forest resources assessment 2010*. Rome, Italy: www.fao.org/3/a-i1757e.pdf

FAO. (2010b). *Global Forest Resources Assessment 2010 - Country Report Ethiopia*. Rome, Italy: www.fao.org/forest-resources-assessment/past-assessments/fra-2010/en/

FAO. (2011). *The state of the world's land and water resources for food and agriculture (SOLAW) – Managing systems at risk*. Rome, Italy and London, UK Food and Agriculture Organization of the United Nations and Earthscan.

FAO. (2015a). *Global forest resources assessment 2015: How are the world's forests changing?* Rome: www.fao.org/forestry/fra

FAO. (2015b). *Global forest resources assessments*. Rome, Italy: www.fao.org/forest-resources-assessment/past-assessments/fra-2015/en/

FAO. (2017). *Soil organic carbon: the hidden potential*. Rome, Italy: www.fao.org/3/a-i6937e.pdf

FDRE. (2011). *Ethiopia's climate resilient green economy strategy: The path to sustainable development*. Addis Abeba, Ethiopia Retrieved from www.undp.org/content/dam/ethiopia/docs/Ethiopia%20CRGE.pdf.

FDRE. (2016). *Ethiopia's forest reference level submission to the UNFCCC*. Addis Ababa, Ethiopia: https://redd.unfccc.int/files/2016_submission_frel_ethiopia.pdf

Fernandes, R. B. A., Carvalho Junior, I. A. d., Ribeiro Junior, E. S., & Mendonça, E. d. S. (2015). Comparison of different methods for the determination of total organic carbon and humic substances in Brazilian soils. *Revista Ceres, 62*, 496-501. doi: http://dx.doi.org/10.1590/0034-737X201562050011

Fetene, A., Hilker, T., Yeshitela, K., Prasse, R., Cohen, W., & Yang, Z. (2016). Detecting trends in land use and land cover change of Nech Sar national park, Ethiopia. *Environmental Management, 57*(1), 137-147. doi: 10.1007/s00267-015-0603-0

Feyissa, A., Soromessa, T., & Argaw, M. (2013). Forest carbon stocks and variations along altitudinal gradients in Egdu forest: Implications of managing forests for climate change mitigation. *Science, Technology and Arts Research Journal, 2*(4), 40-46. doi: http://dx.doi.org/10.4314/star.v2i4.8

Fierer, N., Craine, J. M., McLauchlan, K., & Schimel, J. P. (2005). Litter quality and the temperature sensitivity of decomposition *Ecology, 86*(2), 320-326. doi: doi:10.1890/04-1254

Fikir, D., Tadesse, W., & Gure, A. (2016). Economic contribution to local livelihoods and households dependency on dry land forest products in Hammer District, southeastern Ethiopia. *International Journal of Forestry Research, 2016*, 11. doi: 10.1155/2016/5474680

Fisher, B. (2010). African exception to drivers of deforestation. *Nature Geosci, 3*(6), 375-376.

Foley, J. A., DeFries, R., Asner, G. P., Barford, C., Bonan, G., Carpenter, S. R., . . . Snyder, P. K. (2005). Global consequences of land use. *Science, 309*(5734), 570-574. doi: 10.1126/science.1111772

Friis, I., Demissew, S., & Breugel, P. v. (2010). *Atlas of the potential vegetation of Ethiopia.* Copenhagen, Denmark Det Kongelige Danske Videnskabernes Selskab.

García-Ruiz, J. M. (2010). The effects of land uses on soil erosion in Spain: A review. *CATENA, 81*(1), 1-11. doi: https://doi.org/10.1016/j.catena.2010.01.001

García, M., Riaño, D., Chuvieco, E., & Danson, F. M. (2010). Estimating biomass carbon stocks for a Mediterranean forest in central Spain using LiDAR height and intensity data. *Remote Sensing of Environment, 114*(4), 816-830. doi: https://doi.org/10.1016/j.rse.2009.11.021

Gashaw, T., Tulu, T., Argaw, M., Worqlul, A. W., Tolessa, T., & Kindu, M. (2018). Estimating the impacts of land use/land cover changes on ecosystem service values: The case of the Andassa watershed in the Upper Blue Nile basin of Ethiopia. *Ecosystem Services, 31, Part A*, 219-228. doi: https://doi.org/10.1016/j.ecoser.2018.05.001

Gaston, G., Brown, S., Lorenzini, M., & Singh, K. D. (1998). State and change in carbon pools in the forests of tropical Africa. *Global Change Biology, 4*(1), 97-114. doi: 10.1046/j.1365-2486.1998.00114.x

Gebre Egziabher, T. (1986). Ethiopian vegetation: past, present and future trends. *SINET Ethiop J Sci, 9*(Suppl), 1-11.

Gebrehiwot, K., Muys, B., Haile, M., & Mitloehner, R. (2003). Introducing Boswellia papyrifera (Del.) Hochst and its non-timber forest product, frankincense. *International forestry review, 5*(4), 348-353. doi: https://doi.org/10.1505/IFOR.5.4.348.22661

Geist, H. J., & Lambin, E. F. (2001). *What drives tropical deforestation*. Louvain-la-Neuve, Belgium: www.pik-potsdam.de/~luedeke/lucc4.pdf

Gelaw, A. M., Singh, B. R., & Lal, R. (2014). Soil organic carbon and total nitrogen stocks under different land uses in a semi-arid watershed in Tigray, Northern Ethiopia. *Agriculture, Ecosystems & Environment, 188*(Supplement C), 256-263. doi: https://doi.org/10.1016/j.agee.2014.02.035

Gessesse, T. A., & Khamzina, A. (2018). How reliable is the Walkley-Black method for analyzing carbon-poor, semi-arid soils in Ethiopia? *Journal of Arid Environments, 153*, 98-101. doi: https://doi.org/10.1016/j.jaridenv.2018.01.008

Gibbs, H., & Brown, S. (2007). Geographical distribution of woody biomass carbon stocks in tropical Africa: an updated database for 2000. *Carbon Dioxide Information Center, Oak Ridge National Laboratory, Oak Ridge, TN.*

Gibbs, H. K., Brown, S., Niles, J. O., & Foley, J. A. (2007). Monitoring and estimating tropical forest carbon stocks: making REDD a reality. *Environmental Research Letters, 2*(4), 045023.

Giday, K., Eshete, G., Barklund, P., Aertsen, W., & Muys, B. (2013). Wood biomass functions for Acacia abyssinica trees and shrubs and implications for provision of ecosystem services in a community managed exclosure in Tigray, Ethiopia. *Journal of Arid Environments, 94*. doi: 10.1016/j.jaridenv.2013.03.001

Giday, K., Humnessa, B., Muys, B., Taheri, F., & Azadi, H. (2018). Effects of livestock grazing on key vegetation attributes of a remnant forest reserve: The case of Desa'a Forest in northern Ethiopia. *Global Ecology and Conservation, 14*, e00395. doi: https://doi.org/10.1016/j.gecco.2018.e00395

Gillman, G. P., Sinclair, D. F., & Beech, T. A. (1986). Recovery of organic carbon by the walkley and black procedure in highly weathered soils. *Communications in Soil Science and Plant Analysis, 17*(8), 885-892. doi: 10.1080/00103628609367759

Girardin, C. A. J., Malhi, Y., AragÃO, L. E. O. C., Mamani, M., Huaraca Huasco, W., Durand, L., . . . 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*(12), 3176-3192. doi: 10.1111/j.1365-2486.2010.02235.x

Girmay, G., & Singh, B. R. (2012). Changes in soil organic carbon stocks and soil quality: land-use system effects in northern Ethiopia. *Acta Agriculturae Scandinavica, Section B — Soil & Plant Science, 62*(6), 519-530. doi: 10.1080/09064710.2012.663786

Girmay, G., Singh, B. R., & Dick, Ø. (2010). Land-use changes and their impacts on soil degradation and surface runoff of two catchments of Northern Ethiopia. *Acta Agriculturae Scandinavica, Section B — Soil & Plant Science, 60*(3), 211-226. doi: 10.1080/09064710902821741

Girmay, G., Singh, B. R., Mitiku, H., Borresen, T., & Lal, R. (2008). Carbon stocks in Ethiopian soils in relation to land use and soil management. *Land Degradation & Development, 19*(4), 351-367. doi: 10.1002/ldr.844

Gleason, C. J., & Im, J. (2012). Forest biomass estimation from airborne LiDAR data using machine learning approaches. *Remote Sensing of Environment, 125*(Supplement C), 80-91. doi: https://doi.org/10.1016/j.rse.2012.07.006

Gobeze, T., Bekele, M., Lemenih, M., & Kassa, H. (2009). Participatory forest management and its impacts on livelihoods and forest status: the case of Bonga forest in Ethiopia. *International Forestry Review, 11*(3), 346-358. doi: 10.1505/ifor.11.3.346

Goldin, A. (1987). Reassessing the use of loss-on-ignition for estimating organic matter content in noncalcareous soils. *Communications in Soil Science and Plant Analysis, 18*(10), 1111-1116. doi: 10.1080/00103628709367886

Gómez, D. R., Watterson, J. D., Americano, B. B., Ha, C., Marland, G., Matsika, E., . . . Treanton, K. (2006). IPCC guidelines for national greenhouse gas inventories. *Institute for Global Environmental Strategies, Kamiyamaguchi Hayama, Japan.*

Gond, V., Dubiez, E., Boulogne, M., Gigaud, M., Peroches, A., Pennec, A., . . . Peltier, R. (2016). Forest cover and carbon stock change dynamics in the Democratic Republic of Congo: Case of the wood-fuel supply basin of Kinshasa. *Bois et Forêts des Tropiques*(327), 19-28.

Gorte, R. W. (2009). *Carbon sequestration in forests*. Washington, D.C.: U.S. of Library of Congress, Congressional Research Service.

Graham, K. (2011). *REDD+ and adaptation: will REDD+ contribute to adaptive capacity at the local level?* https://theredddesk.org/sites/default/files/.../redd_adaption_long_-_master_final_0.pdf

Gregorich, E. G., Rochette, P., VandenBygaart, A. J., & Angers, D. A. (2005). Greenhouse gas contributions of agricultural soils and potential mitigation practices in eastern Canada. *Soil and Tillage Research, 83*(1), 53-72. doi: http://dx.doi.org/10.1016/j.still.2005.02.009

Gunderson, L. H. (2000). Ecological resilience--In theory and application. *Annual Review of Ecology and Systematics, 31*, 425-439.

Guo, L. B., & Gifford, R. M. (2002). Soil carbon stocks and land use change: a meta analysis. *Global Change Biology, 8*(4), 345-360. doi: 10.1046/j.1354-1013.2002.00486.x

Guo, Z., Xiao, X., Gan, Y., & Zheng, Y. (2001). Ecosystem functions, services and their values – a case study in Xingshan County of China. *Ecological Economics, 38*(1), 141-154. doi: https://doi.org/10.1016/S0921-8009(01)00154-9

Haghdoost, N., Akbarinia, M., & Hosseini, S. M. (2013). Land-use change and carbon stocks: A case study, Noor County, Iran. *Journal of Forestry Research, 24*(3), 461-469. doi: 10.1007/s11676-013-0340-2

Hailemariam, S., Soromessa, T., & Teketay, D. (2016). Land use and land cover change in the Bale mountain Eco-Region of Ethiopia during 1985 to 2015. *Land, 5*(4), 41. doi: 10.3390/land5040041

Hance, J. (2008). Tropical deforestation is one of the worst crises since we came out of our caves. *Mongabay. com/A Place Out of Time: Tropical Rainforests and the Perils They Face.*

Hansen, C. (1997). Making available information on the conservation and utilization of forest genetic resources. In: The FAO Worldwide Information System on Forest genetic resources.

Hansen, M. C., Potapov, P. V., Moore, R., Hancher, M., Turubanova, S. A., Tyukavina, A., . . . Townshend, J. R. G. (2013). High-resolution global maps of 21st-century forest cover change. *Science, 342*(6160), 850-853. doi: 10.1126/science.1244693

Haregeweyn, N., Berhe, A., Tsunekawa, A., Tsubo, M., & Meshesha, D. T. (2012). Integrated watershed management as an effective approach to curb land degradation: A case study of the Enabered watershed in northern Ethiopia. *Environmental Management, 50*(6), 1219-1233. doi: 10.1007/s00267-012-9952-0

Harmon, M. E., & Sexton, J. (1996). *Guidelines for measurements of woody detritus in forest ecosystems* (Vol. 20): US LTER Network Office Seattle (WA).

Hartanto, H., Prabhu, R., Widayat, A. S. E., & Asdak, C. (2003). Factors affecting runoff and soil erosion: plot-level soil loss monitoring for assessing sustainability of forest management. *Forest Ecology and Management, 180*(1), 361-374. doi: https://doi.org/10.1016/S0378-1127(02)00656-4

Hasen-Yusuf, M., Treydte, A., Abule, E., & Sauerborn, J. (2013). Predicting aboveground biomass of woody encroacher species in semi-arid rangelands, Ethiopia. *Journal of Arid Environments, 96*. doi: 10.1016/j.jaridenv.2013.04.007

Hayes, D. J., & Sader, S. A. (2001). Comparison of change-detection techniques for monitoring tropical forest clearing and vegetation regrowth in a time series. *Photogrammetric Engineering and Remote Sensing, 67*(9), 1067-1075.

Heiri, O., Lotter, A. F., & Lemcke, G. (2001). Loss on ignition as a method for estimating organic and carbonate content in sediments: reproducibility and comparability of results. *Journal of Paleolimnology, 25*(1), 101-110. doi: 10.1023/a:1008119611481

Hemp, A. (2009). Climate change and its impact on the forests of Kilimanjaro. *African Journal of Ecology, 47*(s1), 3-10. doi: doi:10.1111/j.1365-2028.2008.01043.x

Henry, M., Picard, N., Trotta, C., Manlay, R., 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*. doi: 10.14214/sf.38

Herzog, H., & Golomb, D. (2004). Carbon capture and storage from fossil fuel use. *Encyclopedia of Energy, 1*.

Hesslerová, P., Pokorný, J., Brom, J., & Rejšková – Procházková, A. (2013). Daily dynamics of radiation surface temperature of different land cover types in a temperate cultural landscape: Consequences for the local climate. *Ecological Engineering, 54*, 145-154. doi: https://doi.org/10.1016/j.ecoleng.2013.01.036

Hill, M. O. (1973). Diversity and Evenness: A Unifying Notation and Its Consequences. *Ecology, 54*(2), 427-432. doi: 10.2307/1934352

Hobbs, R. J., & Harris, J. A. (2001). Restoration ecology: Repairing the Earth's ecosystems in the New Millennium. *Restoration Ecology, 9*(2), 239-246. doi: 10.1046/j.1526-100x.2001.009002239.x

Holly, K. G., Sandra, B., John, O. N., & Jonathan, A. F. (2007). Monitoring and estimating tropical forest carbon stocks: making REDD a reality. *Environmental Research Letters, 2*(4), 045023.

Hosonuma, N., Herold, M., De Sy, V., De Fries, R. S., Brockhaus, M., Verchot, L., . . . Romijn, E. (2012). An assessment of deforestation and forest degradation drivers in developing countries. *Environmental Research Letters, 7*(4), 044009.

Houghton, J. T. (1996). *Climate change 1995: The science of climate change: contribution of working group I to the second assessment report of the Intergovernmental Panel on Climate Change* (Vol. 2): Cambridge University Press.

Houghton, R. A. (1999). The annual net flux of carbon to the atmosphere from changes in land use 1850–1990*. *Tellus B, 51*(2), 298-313. doi: 10.1034/j.1600-0889.1999.00013.x

Houghton, R. A. (2007). Balancing the global carbon budget. *Annual Review of Earth and Planetary Sciences, 35*(1), 313-347. doi: doi:10.1146/annurev.earth.35.031306.140057

Houghton, R. A., House, J. I., Pongratz, J., van der Werf, G. R., DeFries, R. S., Hansen, M. C., . . . Ramankutty, N. (2012). Carbon emissions from land use and land-cover change. *Biogeosciences, 9*(12), 5125-5142. doi: 10.5194/bg-9-5125-2012

Houghton, R. A., Lawrence, K. T., Hackler, J. L., & Brown, S. (2001). The spatial distribution of forest biomass in the Brazilian Amazon: a comparison of estimates. *Global Change Biology, 7*(7), 731-746. doi: doi:10.1111/j.1365-2486.2001.00426.x

Howard, P. J. A., & Howard, D. M. (1990). Use of organic carbon and loss-on-ignition to estimate soil organic matter in different soil types and horizons. *Biology and Fertility of Soils, 9*(4), 306-310. doi: 10.1007/bf00634106

Hu, H., Liu, W., & Cao, M. (2008). Impact of land use and land cover changes on ecosystem services in Menglun, Xishuangbanna, Southwest China. *Environmental Monitoring and Assessment, 146*(1), 147-156. doi: 10.1007/s10661-007-0067-7

Hughes, R. F., Kauffman, J. B., & Jaramillo, V. J. (1999). Biomass, carbon, and nutrient dynamics of secondary forests in a humid tropical region of Mexico. *Ecology, 80*(6), 1892-1907. doi: 10.1890/0012-9658(1999)080[1892:BCANDO]2.0.CO;2

Humphreys, D. (2014). *Forest politics: the evolution of international cooperation*: Routledge.

Hurni, H. (1993). Land degradation, famine, and land resource scenarios in Ethiopia. In D. Pimentel (Ed.), *World Soil Erosion and Conservation* (pp. 27-62). Cambridge: Cambridge University Press.

Hurni, H., Tato, K., & Zeleke, G. (2005). The implications of changes in population, land use, and land management for surface runoff in the Upper Nile Basin area of Ethiopia. *Mountain Research and Development, 25*(2), 147-154. doi: 10.1659/0276-4741(2005)025[0147:TIOCIP]2.0.CO;2

Hussaini, M. U. (2014). The impact of economic activities on deforestation in Bauchi State. (A theoretical perspective.). *IOSR Journal Of Humanities And Social Science, 19*(3), 14-18.

IPCC. (2007a). *Climate change 2007-Mitigation of climate change: Working group III contribution to the fourth assessment report of the IPCC*. Cambridge, United Kingdom and New York, NY, USA: Cambridge University Press.

IPCC. (2007b). *Climate change 2007 (Fourth assessment report): Synthesis report summary for policymakers*. Geneva:

IPCC. (2013). *Climate change 2013: The physical science basis. Working Group I contribution to the IPCC Fifth Assessment Report.* Cambridge, United Kingdom: www.ipcc.ch/report/ar5/wg1

IPCC. (2014). *Working group II: Impacts, adaptation and vulnerability*. New York, USA:

Jackson, T. (2011). *Prosperity without growth: Economics for a finite planet*: Routledge.

Jacob, M., Frankl, A., Beeckman, H., Mesfin, G., Hendrickx, M., Guyassa, E., & Nyssen, J. (2015). North Ethiopian Afro-Alpine tree line dynamics and forest-cover change since the early 20th century. *Land Degradation & Development, 26*(7), 654-664. doi: doi:10.1002/ldr.2320

Jacob, M., Frankl, A., Hurni, H., Lanckriet, S., De Ridder, M., Guyassa, E., . . . Nyssen, J. (2017). Land cover dynamics in the Simien Mountains (Ethiopia), half a century after establishment of the National Park. *Regional Environmental Change, 17*(3), 777-787. doi: 10.1007/s10113-016-1070-8

Jarno, H. (2016). *Study of causes of deforestation and forest degradation in Ethiopia and the identification and prioritization of strategic options to address those*. Addis Ababa, Ethiopia: https://ethiopiared.org/.../National-study-on-drivers-of-deforestation-and-forest-degra...

Jeanjean, A. P. R., Buccolieri, R., Eddy, J., Monks, P. S., & Leigh, R. J. (2017). Air quality affected by trees in real street canyons: The case of Marylebone neighbourhood in central London. *Urban Forestry & Urban Greening, 22*, 41-53. doi: https://doi.org/10.1016/j.ufug.2017.01.009

Kalaba, F. K., Quinn, C. H., & Dougill, A. J. (2013). Contribution of forest provisioning ecosystem services to rural livelihoods in the Miombo woodlands of Zambia. *Population and Environment, 35*(2), 159-182. doi: 10.1007/s11111-013-0189-5

Kangas, A., & Maltamo, M. (2006). *Forest inventory: methodology and applications* (Vol. 10). Dordrecht, The Netherlands: Springer Science & Business Media.

Kashaigili, J., Mdemu, M., Nduganda, A., & Mbilinyi, B. (2013). Integrated assessment of forest cover change and above-ground carbon stock in Pugu and Kazimzumbwi forest reserves, Tanzania. *Advances in Remote Sensing, 2*(1), 9. doi: 10.4236/ars.2013.21001

Kashaigili, J. J., & Majaliwa, A. M. (2010). Integrated assessment of land use and cover changes in the Malagarasi river catchment in Tanzania. *Physics and Chemistry of the Earth, Parts A/B/C, 35*(13), 730-741. doi: https://doi.org/10.1016/j.pce.2010.07.030

Kassa, H., Birhane, E., Bekele, M., Lemenih, M., Tadesse, W., Cronkleton, P., . . . Baral, H. (2017). Shared strengths and limitations of participatory forest management and area exclosure: two major state led landscape rehabilitation mechanisms in Ethiopia. *International Forestry Review, 19*(4), 51-61.

Kebede, A. G., Bekele, M., & Woldeamanuel, T. (2014). Natural resource use conflict in Bale Mountains National Park, Southeast Ethiopia. *International Journal of Biodiversity and Conservation, 6*(12), 814-822. doi: 10.5897/IJBC2014.0766

Kedir, H., Negash, M., Yimer, F., & Limenih, M. (2018). Contribution of participatory forest management towards conservation and rehabilitation of dry Afromontane forests and its implications for carbon management in the tropical Southeastern Highlands of Ethiopia. *Journal of Sustainable Forestry, 37*(4), 357-374. doi: 10.1080/10549811.2017.1414614

Keeling, R. F. (2009). Atmospheric CO2 records from sites in the SIO air sampling network. In *Trends: A Compendium of Data on Global Change; Carbon Dioxide Information Analysis Center*. Oak Ridge, TN, USA: Oak Ridge National Laboratory, U.S. Department of Energy.

Keenan, R. J., Reams, G. A., Achard, F., de Freitas, J. V., Grainger, A., & Lindquist, E. (2015). Dynamics of global forest area: Results from the FAO Global Forest Resources Assessment 2015. *Forest Ecology and Management, 352*, 9-20. doi: https://doi.org/10.1016/j.foreco.2015.06.014

Keith, H., Mackey, B. G., & Lindenmayer, D. B. (2009). Re-evaluation of forest biomass carbon stocks and lessons from the world's most carbon-dense forests. *Proceedings of the National Academy of Sciences, 106*(28), 11635-11640. doi: 10.1073/pnas.0901970106

Kellomäki, S., Peltola, H., Nuutinen, T., Korhonen, K. T., & Strandman, H. (2008). Sensitivity of managed boreal forests in Finland to climate change, with implications for adaptive management. *Philosophical Transactions of the Royal Society B: Biological Sciences, 363*(1501), 2339-2349. doi: 10.1098/rstb.2007.2204

Kidane, Y., Stahlmann, R., & Beierkuhnlein, C. (2012). Vegetation dynamics, and land use and land cover change in the Bale Mountains, Ethiopia. *Environmental Monitoring and Assessment, 184*(12), 7473-7489. doi: 10.1007/s10661-011-2514-8

Kindu, M., Schneider, T., Teketay, D., & Knoke, T. (2013). Land use/land cover change analysis using object-based classification Approach in Munessa-Shashemene landscape of the Ethiopian highlands. *Remote Sensing, 5*(5), 2411. doi: 10.3390/rs5052411

Kindu, M., Schneider, T., Teketay, D., & Knoke, T. (2015). Drivers of land use/land cover changes in Munessa-Shashemene landscape of the south-central highlands of Ethiopia. *Environmental Monitoring and Assessment, 187*(7), 452. doi: 10.1007/s10661-015-4671-7

Kindu, M., Schneider, T., Teketay, D., & Knoke, T. (2016). Changes of ecosystem service values in response to land use/land cover dynamics in Munessa–Shashemene landscape of the Ethiopian highlands. *Science of the Total Environment, 547*, 137-147. doi: http://dx.doi.org/10.1016/j.scitotenv.2015.12.127

Kirschbaum, M. (2004). Direct and indirect climate change effects on photosynthesis and transpiration. *Plant Biology, 6*(03), 242-253.

Kitula, R. A. (2007). Use of medicinal plants for human health in Udzungwa Mountains Forests: a case study of New Dabaga Ulongambi Forest Reserve, Tanzania. *Journal of Ethnobiology and Ethnomedicine, 3*(1), 7. doi: 10.1186/1746-4269-3-7

Köhl, M., Lasco, R., Cifuentes, M., Jonsson, Ö., Korhonen, K. T., Mundhenk, P., . . . Stinson, G. (2015). Changes in forest production, biomass and carbon: Results from the 2015 UN FAO Global Forest Resource Assessment. *Forest Ecology and Management, 352*, 21-34. doi: http://dx.doi.org/10.1016/j.foreco.2015.05.036

Konen, M. E., Jacobs, P. M., Burras, C. L., Talaga, B. J., & Mason, J. A. (2002). Equations for predicting soil organic carbon using loss-on-ignition for north central U.S. Soils. *Soil Science Society of America Journal, 66*, 1878-1881. doi: 10.2136/sssaj2002.1878

Kremen, C., Niles, J. O., Dalton, M. G., Daily, G. C., Ehrlich, P. R., Fay, J. P., . . . Guillery, R. P. (2000). Economic incentives for rain forest conservation across scales. *Science, 288*(5472), 1828-1832. doi: 10.1126/science.288.5472.1828

Kreuter, U. P., Harris, H. G., Matlock, M. D., & Lacey, R. E. (2001). Change in ecosystem service values in the San Antonio area, Texas. *Ecological Economics, 39*(3), 333-346. doi: http://dx.doi.org/10.1016/S0921-8009(01)00250-6

Kroeger, T., & Casey, F. (2007). An assessment of market-based approaches to providing ecosystem services on agricultural lands. *Ecological Economics, 64*(2), 321-332. doi: https://doi.org/10.1016/j.ecolecon.2007.07.021

Kumar, R., Nandy, S., Agarwal, R., & Kushwaha, S. P. S. (2014). Forest cover dynamics analysis and prediction modeling using logistic regression model. *Ecological Indicators, 45*, 444-455. doi: http://dx.doi.org/10.1016/j.ecolind.2014.05.003

Kumi-Boateng, B., Mireku-Gyimah, D., & Duker, A. (2015). A spatio-temporal based estimation of sequestered carbon in the Tarkwa mining area of Ghana. *Ghana Mining Journal, 15*(1), 10-21.

Kurgat, B. K., Golicha, D., Giese, M., Kuria, S., & Asch, F. (2011). Relationship between vegetation cover types and soil organic carbon in the rangelands of Northern Kenya. *Livestock Research for Rural Development, 26*(9).

Lanly, J.-P. (2003). *Deforestation and forest degradation factors.* Paper presented at the Congress Proceedings B, XII World Forestry Congress, Quebec, Canada.

Larsson, R. Å., & Strömquist, L. (1991). *A practical approach to satellite image analysis for environmental monitoring*: Ministry of Agriculture and Marketing.

Law, B. E., Falge, E., Gu, L., Baldocchi, D. D., Bakwin, P., Berbigier, P., . . . Wofsy, S. (2002). Environmental controls over carbon dioxide and water vapor exchange of terrestrial vegetation. *Agricultural and Forest Meteorology, 113*(1–4), 97-120. doi: http://dx.doi.org/10.1016/S0168-1923(02)00104-1

Le Quere, C., Raupach, M. R., Canadell, J. G., & Marland, G. (2009). Trends in the sources and sinks of carbon dioxide. *Nature Geosci, 2*(12), 831-836. doi: http://www.nature.com/ngeo/journal/v2/n12/suppinfo/ngeo689_S1.html

Le Toan, T., Quegan, S., Davidson, M. W. J., Balzter, H., Paillou, P., Papathanassiou, K., . . . Ulander, L. (2011). The BIOMASS mission: Mapping global forest biomass to better understand the terrestrial carbon cycle. *Remote Sensing of Environment, 115*(11), 2850-2860. doi: https://doi.org/10.1016/j.rse.2011.03.020

Lefsky, M. A., Cohen, W. B., Harding, D. J., Parker, G. G., Acker, S. A., & Gower, S. T. (2002). Lidar remote sensing of above-ground biomass in three biomes. *Global Ecology and Biogeography, 11*(5), 393-399. doi: 10.1046/j.1466-822x.2002.00303.x

Lemenih, M., & Itanna, F. (2004). Soil carbon stocks and turnovers in various vegetation types and arable lands along an elevation gradient in southern Ethiopia. *Geoderma, 123*(1–2), 177-188. doi: http://dx.doi.org/10.1016/j.geoderma.2004.02.004

Lemenih, M., Karltun, E., & Olsson, M. (2005). Assessing soil chemical and physical property responses to deforestation and subsequent cultivation in smallholders farming system in Ethiopia. *Agriculture, Ecosystems & Environment, 105*(1), 373-386. doi: https://doi.org/10.1016/j.agee.2004.01.046

Lemenih, M., & Woldemariam, T. (2010). *Review of forest, woodland and bushland resources in Ethiopia up to 2008.* Paper presented at the Ethiopian environment review.

Lettens, S., De Vos, B., Quataert, P., Van Wesemael, B., Muys, B., & Van Orshoven, J. (2007). Variable carbon recovery of Walkley-Black analysis and implications for national soil organic carbon accounting. *European Journal of Soil Science, 58*(6), 1244-1253. doi: 10.1111/j.1365-2389.2007.00916.x

Li, J., Wang, W., Hu, G., & Wei, Z. (2010a). Changes in ecosystem service values in Zoige Plateau, China. *Agriculture, Ecosystems & Environment, 139*(4), 766-770. doi: http://dx.doi.org/10.1016/j.agee.2010.10.019

Li, P., Wang, Q., Endo, T., Zhao, X., & Kakubari, Y. (2010b). Soil organic carbon stock is closely related to aboveground vegetation properties in cold-temperate mountainous forests. *Geoderma, 154*(3–4), 407-415. doi: http://dx.doi.org/10.1016/j.geoderma.2009.11.023

Li, R.-Q., Dong, M., Cui, J.-Y., Zhang, L.-L., Cui, Q.-G., & He, W.-M. (2007). Quantification of the Impact of Land-Use Changes on Ecosystem Services: A Case Study in

Pingbian County, China. *Environmental Monitoring and Assessment, 128*(1), 503-510. doi: 10.1007/s10661-006-9344-0

Li, Y., Zhao, M., Motesharrei, S., Mu, Q., Kalnay, E., & Li, S. (2015). Local cooling and warming effects of forests based on satellite observations. *Nature Communications, 6*, 6603. doi: 10.1038/ncomms7603https://www.nature.com/articles/ncomms760#supplementary-information

Liang, S., Li, X., & Wang, J. (2012). *Advanced remote sensing : terrestrial information extraction and applications*. San Diego, USA: Academic Press.

Linder, H. P. (2001). Plant diversity and endemism in sub-Saharan tropical Africa. *Journal of Biogeography, 28*(2), 169-182. doi: 10.1046/j.1365-2699.2001.00527.x

Liniger, H., & Weingartner, R. (1998). Mountains and freshwater supply. *Unasylva (FAO)*.

Liu, W., Chen, S., Qin, X., Baumann, F., Scholten, T., Zhou, Z., . . . Qin, D. (2012). Storage, patterns, and control of soil organic carbon and nitrogen in the northeastern margin of the Qinghai–Tibetan Plateau. *Environmental Research Letters, 7*(3), 035401. doi: doi:10.1088/1748-9326/7/3/035401

Loomis, J. (2005). Valuing environmental and natural resources: The econometrics of non-market valuation. *American Journal of Agricultural Economics, 87*(2), 529-530. doi: 10.1111/j.0002-9092.2005.740_2.x

Loomis, J., Kent, P., Strange, L., Fausch, K., & Covich, A. (2000). Measuring the total economic value of restoring ecosystem services in an impaired river basin: results from a contingent valuation survey. *Ecological Economics, 33*(1), 103-117. doi: https://doi.org/10.1016/S0921-8009(99)00131-7

Lu, D. (2005). Aboveground biomass estimation using Landsat TM data in the Brazilian Amazon. *International Journal of Remote Sensing, 26*(12), 2509-2525. doi: 10.1080/01431160500142145

Lu, D. (2006). The potential and challenge of remote sensing-based biomass estimation. *International Journal of Remote Sensing, 27*(7), 1297-1328. doi: 10.1080/01431160500486732

Lu, D., Mausel, P., Brondízio, E., & Moran, E. (2004). Change detection techniques. *International Journal of Remote Sensing, 25*(12), 2365-2401. doi: 10.1080/0143116031000139863

Lü, X.-T., Yin, J.-X., Jepsen, M. R., & Tang, J.-W. (2010). Ecosystem carbon storage and partitioning in a tropical seasonal forest in Southwestern China. *Forest Ecology and Management, 260*(10), 1798-1803. doi: https://doi.org/10.1016/j.foreco.2010.08.024

Lunetta, R. (1998). Applications, project formulation, and analytical approach. *Remote sensing change detection: environmental monitoring methods and applications*, 1-19.

Luong, N. V., Tateishi, R., Hoan, N. T., & Tu, T. T. (2015). Forest Change and Its Effect on Biomass in Yok Don National Park in Central Highlands of Vietnam Using Ground Data and Geospatial Techniques. *Advances in Remote Sensing, 4*(02), 108.

Luterbacher, U., & Sprinz, D. F. (2001). *International relations and global climate change*. Cambridge, Mass.: MIT Press.

MacDicken, K., Jonsson, Ö., Piña, L., Maulo, S., Contessa, V., Adikari, Y., . . . D'Annunzio, R. (2016). *Global forest resources assessment 2015: how are the world's forests changing?* (9251092834). Rome, Italy: https://www.uncclearn.org/sites/default/files/inventory/a-i4793e.pdf

MacLean, M. G., & Congalton, R. G. (2012). *Map accuracy assessment issues when using an object-oriented approach.* Paper presented at the Proceedings of the American Society for Photogrammetry and Remote Sensing 2012 Annual Conference.

Maimaitiaili, A., Aji, X., Matniyaz, A., & Kondoh, A. (2018). Monitoring and analysing land use/cover changes in an arid region based on multi-satellite data: The Kashgar region, northwest China. *Land, 7*(1), 6.

Majumdar, S., Deng, J., Zhang, Y., & Pierskalla, C. (2011). Using contingent valuation to estimate the willingness of tourists to pay for urban forests: A study in Savannah, Georgia. *Urban Forestry & Urban Greening, 10*(4), 275-280. doi: https://doi.org/10.1016/j.ufug.2011.07.006

Malhi, Y., Roberts, J. T., Betts, R. A., Killeen, T. J., Li, W., & Nobre, C. A. (2008). Climate change, deforestation, and the fate of the Amazon. *Science, 319*(5860), 169-172. doi: 10.1126/science.1146961

Mamo, G., Sjaastad, E., & Vedeld, P. (2007). Economic dependence on forest resources: A case from Dendi District, Ethiopia. *Forest Policy and Economics, 9*(8), 916-927. doi: https://doi.org/10.1016/j.forpol.2006.08.001

Mathew, M. M., Majule, A. E., Sinclair, F., & Marchant, R. (2016). Relationships between on-farm tree stocks and soil organic carbon along an altitudinal gradient, Mount Kilimanjaro, Tanzania. *Forests, Trees and Livelihoods, 25*(4), 255-266. doi: 10.1080/14728028.2016.1202790

Mayes, M. T., Mustard, J. F., & Melillo, J. M. (2015). Forest cover change in Miombo Woodlands: modeling land cover of African dry tropical forests with linear spectral mixture analysis. *Remote Sensing of Environment, 165*, 203-215. doi: https://doi.org/10.1016/j.rse.2015.05.006

McCann, J. C. (1997). The plow and the forest: Narratives of deforestation in Ethiopia, 1840-1992. *Environmental History, 2*(2), 138-159. doi: 10.2307/3985505

McIntyre, S., & Lavorel, S. (2007). A conceptual model of land use effects on the structure and function of herbaceous vegetation. *Agriculture, Ecosystems & Environment, 119*(1), 11-21. doi: https://doi.org/10.1016/j.agee.2006.06.013

McMahon, S. M., Parker, G. G., & Miller, D. R. (2010). Evidence for a recent increase in forest growth. *Proceedings of the National Academy of Sciences, 107*(8), 3611-3615. doi: 10.1073/pnas.0912376107

MEA. (2005a). *Ecosystem and human well-being* (Vol. 5). Washington, DC: Island Press.

MEA. (2005b). *Ecosystems and human well-being: Biodiversity Synthesis* (Vol. 100). Washington, DC: Island Press

MEA. (2005c). Ecosystems and human well-being: current state and trends. In. Washington, DC: Island Press.

MEFCC. (2016). *Ethiopia's national definition of forest*. Addis Ababa, Ethiopia.

Mekuria, W. (2013). Changes in regulating ecosystem services following establishing exclosures on communal grazing lands in Ethiopia: a synthesis. *Journal of Ecosystems, 2013*, 12. doi: http://dx.doi.org/10.1155/2013/860736

Mekuria, W., Veldkamp, E., & Haile, M. (2009a). *Carbon stock changes with relation to land use conversion in the lowlands of Tigray, Ethiopia.* Paper presented at the Proceedings of Conference on International Research on Food Security, Natural Resource Management and Rural Development; Proceedings of the 36th Meeting of the Italian Society of Agronomy.

Mekuria, W., Veldkamp, E., Haile, M., Gebrehiwot, K., Muys, B., & Nyssen, J. (2009b). Effectiveness of exclosures to control soil erosion and local community perception on soil erosion in Tigray, Ethiopia. *African Journal of Agricultural Research, 4*(4), 365-377.

Mekuria, W., & Yami, M. (2013). Changes in woody species composition following establishing exclosures on grazing lands in the lowlands of northern Ethiopia. *African Journal of Environmental Science and Technology, 7*(1), 30-40. doi: DOI: 10.5897/AJEST11.378

Mengistu, T., Teketay, D., Hulten, H., & Yemshaw, Y. (2005). The role of enclosures in the recovery of woody vegetation in degraded dryland hillsides of central and northern Ethiopia. *Journal of Arid Environments, 60*(2), 259-281. doi: http://dx.doi.org/10.1016/j.jaridenv.2004.03.014

Metz, B., Davidson, O., Bosch, P., Dave, R., & Meyer, L. (2007). *Climate change 2007: Mitigation: Contribution of working group III to the fourth assessment report of the Intergovernmental Panel on Climate Change: Summary for policymakers and technical summary*: Cambridge University Press.

Miura, S., Amacher, M., Hofer, T., San-Miguel-Ayanz, J., Ernawati, & Thackway, R. (2015). Protective functions and ecosystem services of global forests in the past quarter-century. *Forest Ecology and Management, 352*, 35-46. doi: https://doi.org/10.1016/j.foreco.2015.03.039

MoA. (2000). *Agro ecological Zones of Ethiopia*. Addis Ababa, Ethiopia.

Moges, Y., Eshetu, Z., & Nune, S. (2010). *Ethiopian forest resources: Current status and future management options in view of access to carbon finances*. Addis Ababa, Ethiopia

Moges, Y., & Tenkir, E. (2014). Overview of REDD+ process in Ethiopia. *Overview of REDD+ process in Ethiopia. REDD+ secretariat. Ministry of Environment and Forest, Addis Ababa.*

Montès, N., Gauquelin, T., Badri, W., Bertaudière, V., & Zaoui, E. H. (2000). A non-destructive method for estimating above-ground forest biomass in threatened woodlands. *Forest Ecology and Management, 130*(1), 37-46. doi: https://doi.org/10.1016/S0378-1127(99)00188-7

Morales-Hidalgo, D., Oswalt, S. N., & Somanathan, E. (2015). Status and trends in global primary forest, protected areas, and areas designated for conservation of biodiversity from the Global Forest Resources Assessment 2015. *Forest Ecology and Management, 352*, 68-77. doi: https://doi.org/10.1016/j.foreco.2015.06.011

Muluneh, A., van Loon, E., Bewket, W., Keesstra, S., Stroosnijder, L., & Burka, A. (2017). Effects of long-term deforestation and remnant forests on rainfall and temperature in the Central Rift Valley of Ethiopia. *Forest Ecosystems, 4*(1), 23. doi: 10.1186/s40663-017-0109-8

Munishi, P. K. T., & Shear, T. H. (2004). Carbon storage in Afromontane rain forests of the eastern arc mountains of Tanzania: Their net contribution to atmospheric carbon *Journal of Tropical Forest Science, 16*(1), 78-93.

Murty, D., Kirschbaum, M. U. F., McMurtrie, R. E., & McGilvray, H. (2002). Does conversion of forest to agricultural land change soil carbon and nitrogen? a review of the literature. *Global Change Biology, 8*(2), 105-123. doi: 10.1046/j.1354-1013.2001.00459.x

Musa, L., Peters, K., & Ahmed, M. (2006). On farm characterization of Butana and Kenana cattle breed production systems in Sudan. *Livestock Research for Rural Development, 18*(12), 2006.

Nakakaawa, C. A., Vedeld, P. O., & Aune, J. B. (2011). Spatial and temporal land use and carbon stock changes in Uganda: implications for a future REDD strategy. *Mitigation and Adaptation Strategies for Global Change, 16*(1), 25-62. doi: 10.1007/s11027-010-9251-0

Návar, J. (2009). Allometric equations for tree species and carbon stocks for forests of northwestern Mexico. *Forest Ecology and Management, 257*(2), 427-434. doi: https://doi.org/10.1016/j.foreco.2008.09.028

Neary, D. G., Ice, G. G., & Jackson, C. R. (2009). Linkages between forest soils and water quality and quantity. *Forest Ecology and Management, 258*(10), 2269-2281. doi: https://doi.org/10.1016/j.foreco.2009.05.027

Negash, M. (2013). The indigenous agroforestry systems of the south-eastern Rift Valley escarpment, Ethiopia: Their biodiversity, carbon stocks, and litterfall. *Tropical Forestry Reports*(44).

Nelson, D., & Sommers, L. E. (1982). Total carbon, organic carbon, and organic matter. *Methods of soil analysis. Part 2. Chemical and microbiological properties*(methodsofsoilan2), 539-579.

Nelson, D. W., & Sommers, L. E. (1996). Total carbon, organic carbon, and organic matter. *Methods of soil analysis part 3—chemical methods*(methodsofsoilan3), 961-1010.

Nkonya, E., Mirzabaev, A., & Von Braun, J. (2016). *Economics of land degradation and improvement: a global assessment for sustainable development*: Springer.

Noble, I., Bolin, B., Ravindranath, N., Verardo, D., & Dokken, D. (2000). *Land use, land use change, and forestry*: Cambridge University Press.

Noriko, H., Martin, H., Veronique De, S., Ruth, S. D. F., Maria, B., Louis, V., . . . Erika, R. (2012). An assessment of deforestation and forest degradation drivers in developing countries. *Environmental Research Letters, 7*(4), 044009.

Nowak, D. J., Crane, D. E., & Stevens, J. C. (2006). Air pollution removal by urban trees and shrubs in the United States. *Urban Forestry & Urban Greening, 4*(3), 115-123. doi: https://doi.org/10.1016/j.ufug.2006.01.007

Nune, S., Kassie, M., & Mungatana, E. (2010). *Forestry resource accounting: the experience of Ethiopia.*

Olesen, J. E., Carter, T. R., Díaz-Ambrona, C. H., Fronzek, S., Heidmann, T., Hickler, T., . . . Sykes, M. T. (2007). Uncertainties in projected impacts of climate change on European agriculture and terrestrial ecosystems based on scenarios from regional climate models. *Climatic Change, 81*(1), 123-143. doi: 10.1007/s10584-006-9216-1

Olofsson, P., Foody, G. M., Herold, M., Stehman, S. V., Woodcock, C. E., & Wulder, M. A. (2014). Good practices for estimating area and assessing accuracy of land change. *Remote Sensing of Environment, 148*, 42-57. doi: http://dx.doi.org/10.1016/j.rse.2014.02.015

Ordóñez, J. A. B., de Jong, B. H. J., García-Oliva, F., Aviña, F. L., Pérez, J. V., Guerrero, G., . . . Masera, O. (2008). Carbon content in vegetation, litter, and soil under 10 different land-use and land-cover classes in the central highlands of Michoacan, Mexico. *Forest Ecology and Management, 255*(7), 2074-2084. doi: https://doi.org/10.1016/j.foreco.2007.12.024

Ostle, N. J., Levy, P. E., Evans, C. D., & Smith, P. (2009). UK land use and soil carbon sequestration. *Land Use Policy, 26, Supplement 1*, S274-S283. doi: http://dx.doi.org/10.1016/j.landusepol.2009.08.006

Pagiola, S., Arcenas, A., & Platais, G. (2005). Can payments for environmental services help reduce poverty? An exploration of the issues and the evidence to date from Latin America. *World Development, 33*(2), 237-253. doi: http://dx.doi.org/10.1016/j.worlddev.2004.07.011

Pan, Y., Birdsey, R. A., Fang, J., Houghton, R., Kauppi, P. E., Kurz, W. A., . . . Hayes, D. (2011). A large and persistent carbon sink in the world's forests. *Science*. doi: 10.1126/science.1201609

Paustian, K., Ravindranath, N. H., & Amstel, A. R. v. (2006). *2006 IPCC guidelines for national greenhouse gas inventories*. [S.l.]: http://edepot.wur.nl/52458

Payn, T., Carnus, J.-M., Freer-Smith, P., Kimberley, M., Kollert, W., Liu, S., . . . Wingfield, M. J. (2015). Changes in planted forests and future global implications. *Forest Ecology and Management, 352*, 57-67. doi: https://doi.org/10.1016/j.foreco.2015.06.021

Pearson, T., Walker, S., & Brown, S. (2005). *Sourcebook for land use, land-use change and forestry projects*. Arlington, USA: Winrock International and the Bio-carbon fund of the World Bank.

Peltier, R., Njiti Forkong, C., Ntoupka, M., Manlay, R. J., J., Henry, M., & Morillon, V. (2007). Evaluation of carbon stocks and fuelwood productivity in a Shea tree zone in northern Cameroon *Bois et Forêts des Tropiques*(294), 39-50.

Penman, J., Gytarsky, M., Hiraishi, T., Krug, T., Kruger, D., Pipatti, R., . . . Tanabe, K. (2003). *Good practice guidance for land use, land-use change and forestry* (4887880030). Tokyo, Japan

Perez-Verdin, G., Sanjurjo-Rivera, E., Galicia, L., Hernandez-Diaz, J. C., Hernandez-Trejo, V., & Marquez-Linares, M. A. (2016). Economic valuation of ecosystem services in Mexico: Current status and trends. *Ecosystem Services, 21*, 6-19. doi: https://doi.org/10.1016/j.ecoser.2016.07.003

Peskett, L. (2011). Benefit sharing in REDD+: exploring the implications for poor and vulnerable people. *World Bank and REDD-net*, 244362-1232059926563.

Place, F., Pender, J., & Ehui, S. (2006). Key issues for the sustainable development of smallholder agriculture in the East African highlands. *Strategies for Sustainable Land Management in the East African Highlands*, 1-30.

Pokorny, J., Brom, J., Cermak, J., Hesslerova, P., Huryna, H., Nadezhdina, N., & Rejskova, A. (2010). Solar energy dissipation and temperature control by water and plants. *International Journal of Water, 5*(4), 311-336. doi: 10.1504/ijw.2010.038726

Polasky, S., Nelson, E., Pennington, D., & Johnson, K. A. (2011). The impact of land-use change on ecosystem services, biodiversity and returns to landowners: A case study in the State of Minnesota. *Environmental and Resource Economics, 48*(2), 219-242. doi: 10.1007/s10640-010-9407-0

Portela, R., & Rademacher, I. (2001). A dynamic model of patterns of deforestation and their effect on the ability of the Brazilian Amazonia to provide ecosystem services. *Ecological Modelling, 143*(1–2), 115-146. doi: http://dx.doi.org/10.1016/S0304-3800(01)00359-3

Post, W., Emanuel, W., Zinke, P., & Stangenberger, A. (1982). Soil carbon pools and world life zones. *Nature, 298*(5870), 156-159.

Post, W. M., & Kwon, K. C. (2000). Soil carbon sequestration and land-use change: processes and potential. *Global Change Biology, 6*(3), 317-327. doi: 10.1046/j.1365-2486.2000.00308.x

Potapov, P. V., Turubanova, S. A., Tyukavina, A., Krylov, A. M., McCarty, J. L., Radeloff, V. C., & Hansen, M. C. (2015). Eastern Europe's forest cover dynamics from 1985 to 2012 quantified from the full Landsat archive. *Remote Sensing of Environment, 159*, 28-43. doi: https://doi.org/10.1016/j.rse.2014.11.027

Prescott, C. E. (2010). Litter decomposition: what controls it and how can we alter it to sequester more carbon in forest soils? *Biogeochemistry, 101*(1), 133-149. doi: 10.1007/s10533-010-9439-0

Price, M., Gratzer, G., Alemayehu Duguma, L., Kohler, T., & Maselli, D. (2011). *Mountain forests in a changing world: Realizing values, adressing challenges*. Rome, Italy: Food and Agriculture Organization of the United Nations (FAO) and Swiss Agency for Development and Cooperation (SDC).

Priyadarshan, P. M. (2011). *Biology of Hevea rubber*. Wallingford, UK: Springer.

Rajput, B. S., Bhardwaj, D. R., & Pala, N. A. (2017). Factors influencing biomass and carbon storage potential of different land use systems along an elevational gradient in temperate northwestern Himalaya. *Agroforestry Systems, 91*(3), 479-486. doi: 10.1007/s10457-016-9948-5

Ravindranath, N., & Ostwald, M. (2008). Methods for estimating above-ground biomass. In *Carbon inventory methods handbook for greenhouse gas inventory, carbon mitigation and roundwood production projects* (pp. 113-147). Dordrecht, Netherlands: Springer

Reddy, T. (2011). Carbon trading in Africa : a critical review. *Institute for Security Studies Monographs, 2011*(184), 194.

Reichle, D., Houghton, J., Kane, B., Ekmann, J., & et al. (1999). *Carbon sequestration research and development* (DOE/SC/FE-1; TRN: US200310%%188 United States10.2172/810722TRN: US200310%%188Fri Oct 10 16:34:43 EDT 2014OSTI as DE00810722DOESCEnglish). http://www.osti.gov/scitech//servlets/purl/810722/

Reusing, M. (1998). Monitoring of forest resources in Ethiopia. *Ministry of Agriculture, Addis Ababa.*

Rossi, J., Govaerts, A., De Vos, B., Verbist, B., Vervoort, A., Poesen, J., . . . Deckers, J. (2009). Spatial structures of soil organic carbon in tropical forests—A case study of southeastern Tanzania. *CATENA, 77*(1), 19-27. doi: https://doi.org/10.1016/j.catena.2008.12.003

Rovero, F., Martin, E., Rosa, M., Ahumada, J. A., & Spitale, D. (2014). Estimating species richness and modelling habitat preferences of tropical forest mammals from camera trap data. *PLOS ONE, 9*(7), e103300. doi: 10.1371/journal.pone.0103300

Saatchi, S. S., Harris, N. L., Brown, S., Lefsky, M., Mitchard, E. T., Salas, W., . . . Hagen, S. (2011). Benchmark map of forest carbon stocks in tropical regions across three continents. *Proceedings of the National Academy of Sciences, 108*(24), 9899-9904.

Salinger, M. J. (2007). Agriculture's influence on climate during the Holocene. *Agricultural and Forest Meteorology, 142*(2–4), 96-102. doi: http://dx.doi.org/10.1016/j.agrformet.2006.03.024

Sands, R. (2013). *Forestry in a global context, 2nd edition*. University of Canterbury, New Zealand CABI Books

Santilli, M., Moutinho, P., Schwartzman, S., Nepstad, D., Curran, L., & Nobre, C. (2005). Tropical deforestation and the kyoto protocol. *Climatic Change, 71*(3), 267-276. doi: 10.1007/s10584-005-8074-6

Sato, J. H., Figueiredo, C. C. d., Marchão, R. L., Madari, B. E., Benedito, L. E. C., Busato, J. G., & Souza, D. M. d. (2014). Methods of soil organic carbon determination in Brazilian savannah soils. *Scientia Agricola, 71*, 302-308.

Schröter, D., Cramer, W., Leemans, R., Prentice, I. C., Araújo, M. B., Arnell, N. W., . . . Zierl, B. (2005). Ecosystem Service Supply and Vulnerability to Global Change in Europe. *Science, 310*(5752), 1333-1337. doi: 10.1126/science.1115233

Scurlock, J. M. O., & Hall, D. O. (1998). The global carbon sink: a grassland perspective. *Global Change Biology, 4*(2), 229-233. doi: 10.1046/j.1365-2486.1998.00151.x

Segura, M., & Kanninen, M. (2005). Allometric models for tree volume and total aboveground biomass in a tropical humid forest in Costa Rica. *Biotropica, 37*(1), 2-8. doi: 10.1111/j.1744-7429.2005.02027.x

Selmi, W., Weber, C., Rivière, E., Blond, N., Mehdi, L., & Nowak, D. (2016). Air pollution removal by trees in public green spaces in Strasbourg city, France. *Urban Forestry & Urban Greening, 17*, 192-201. doi: https://doi.org/10.1016/j.ufug.2016.04.010

Sen, D. (2018). Information about a forest ecosystem. Retrieved from https://sciencing.com/about-6325114-information-forest-ecosystem.html

Senbeta, F., & Denich, M. (2006). Effects of wild coffee management on species diversity in the Afromontane rainforests of Ethiopia. *Forest Ecology and Management, 232*(1), 68-74. doi: https://doi.org/10.1016/j.foreco.2006.05.064

Seyoum, Y., Birhane, E., Mengistu, T., Esmael, N., Hagazi, N., & Kassa, H. (2015). *Enhancing the role of the forestry sector in Ethiopia forest management practices in Tigray region with emphasis on area exclosure*.

Shackleton, C. M., Shackleton, S. E., Buiten, E., & Bird, N. (2007). The importance of dry woodlands and forests in rural livelihoods and poverty alleviation in South Africa. *Forest Policy and Economics, 9*(5), 558-577. doi: https://doi.org/10.1016/j.forpol.2006.03.004

Sharma, V., Hussain, S., Sharma, K. R., & Arya, V. M. (2014). Labile carbon pools and soil organic carbon stocks in the foothill Himalayas under different land use systems. *Geoderma, 232-234*(Supplement C), 81-87. doi: https://doi.org/10.1016/j.geoderma.2014.04.039

Shoo, R. A., & Songorwa, A. N. (2013). Contribution of eco-tourism to nature conservation and improvement of livelihoods around Amani nature reserve, Tanzania. *Journal of Ecotourism, 12*(2), 75-89. doi: 10.1080/14724049.2013.818679

Shrestha, B. M., Dick, Ø. B., & Singh, B. (2010). Effects of land-use change on carbon dynamics assessed by multi-temporal satellite imagery in a mountain watershed of Nepal. *Acta Agriculturae Scandinavica, Section B — Soil & Plant Science, 60*(1), 10-23. doi: 10.1080/09064710802537678

SILVA, J. G. D., STEINER, A., & SOLHEIM, E. (2018). Forests: A Natural Solution to Climate Change, Crucial for a Sustainable Future.

Simegn, T. Y., Soromessa, T., & Bayable, E. (2014). Forest carbon stocks in lowland area of Simien Mountains National Park: Implication for climate change mitigation. *Science, Technology and Arts Research Journal, 3*(3), 29-36. doi: http://dx.doi.org/10.4314/star.v3i3.5

Singer, S. F. (Ed.) (2008). *Nature, not human activity, rules the climate: Summary for policymakers of the report of the nongovernmental International Panel on Climate Change*. The Heartland Institute: Chicago, USA.

Slik, J. W. F., Arroyo-Rodríguez, V., Aiba, S.-I., Alvarez-Loayza, P., Alves, L. F., Ashton, P., . . . Venticinque, E. M. (2015). An estimate of the number of tropical tree species. *Proceedings of the National Academy of Sciences, 112*(24), 7472-7477. doi: 10.1073/pnas.1423147112

Smith, J., & Scherr, S. J. (2002). *Forest carbon and local livelihoods: Assessment of opportunities and policy recommendations*. CIFOR Occasional Paper, no. 37. Center for International Forestry Research Jakarta, Indonesia Retrieved from http://hdl.handle.net/10535/3725

Soares-Filho, B., Moutinho, P., Nepstad, D., Anderson, A., Rodrigues, H., Garcia, R., . . . Maretti, C. (2010). Role of Brazilian Amazon protected areas in climate change mitigation. *Proceedings of the National Academy of Sciences, 107*(24), 10821-10826. doi: 10.1073/pnas.0913048107

Solomon, N., Birhane, E., Tadesse, T., Treydte, A. C., & Meles, K. (2017). Carbon stocks and sequestration potential of dry forests under community management in Tigray, Ethiopia. *Ecological Processes, 6*(1), 20. doi: 10.1186/s13717-017-0088-2

Spies, T. A., Franklin, J. F., & Thomas, T. B. (1988). Coarse woody debris in Douglas-Fir forests of western Oregon and Washington. *Ecology, 69*(6), 1689-1702. doi: doi:10.2307/1941147

Stern, N. H., Peters, S., Bakhshi, V., Bowen, A., Cameron, C., Catovsky, S., . . . Edmonson, N. (2006). *Stern review: The economics of climate change* (Vol. 30): Cambridge University Press Cambridge.

Stibig, H.-J., Achard, F., Carboni, S., Rasi, R., & Miettinen, J. (2014). Change in tropical forest cover of Southeast Asia from 1990 to 2010. *Biogeosciences, 11*(2), 247. doi: 10.5194/bg-11-247-2014

Stinson, G., Kurz, W. A., Smyth, C. E., Neilson, E. T., Dymond, C. C., Metsaranta, J. M., . . . Blain, D. (2011). An inventory-based analysis of Canada's managed forest carbon dynamics, 1990 to 2008. *Global Change Biology, 17*(6), 2227-2244. doi: doi:10.1111/j.1365-2486.2010.02369.x

Streck, C. (2012). Financing REDD+: matching needs and ends. *Current Opinion in Environmental Sustainability, 4*(6), 628-637.

Styers, D. M., Chappelka, A. H., Marzen, L. J., & Somers, G. L. (2010). Developing a land-cover classification to select indicators of forest ecosystem health in a rapidly urbanizing landscape. *Landscape and Urban Planning, 94*(3–4), 158-165. doi: http://dx.doi.org/10.1016/j.landurbplan.2009.09.006

Sundquist, E., Burruss, R., Faulkner, S., Gleason, R., Harden, J., Kharaka, Y., . . . Waldrop, M. (2008). Carbon sequestration to mitigate climate change: US Geological Survey, Fact Sheet 2008–3097. In.

Syktus, J. I., & McAlpine, C. A. (2016). More than carbon sequestration: Biophysical climate benefits of restored savanna woodlands. *Scientific Reports, 6*, 29194. doi: 10.1038/srep29194https://www.nature.com/articles/srep2919#supplementary-information

Tariq, M., & Aziz, R. (2015). An overview of deforestation causes and its environmental hazards in Khyber Pukhtunkhwa. *Journal of Natural Sciences Research, 5*(1).

Teketay, D. (1996). *Seed ecology and regeneration in dry Afromontane forests of Ethiopia.* Umeå, Sweden: Swedish University of Agricultural Sciences.

Teketay, D. (2000). *Facts and experience on eucalyptus in Ethiopia and elsewhere: ground for making wise and informed decision.* Paper presented at the Workshop on Eucalyptus Dilemma.

Teketay, D. (2001). Deforestation, wood famine, and environmental degradation in Ethiopia's highland ecosystems: urgent need for action. *Northeast African Studies, 8*(1), 53-76.

Teketay, D. (2004). *Forestry research in Ethiopia: Past, present and future.* Paper presented at the A National Conference on Forest Resources of Ethiopia,, Addis Abeba (Ethiopia), 27-29 Nov 2002.

Teketay, D., Lemenih, M., Bekele, T., Yemshaw, Y., Feleke, S., Tadesse, W., . . . Nigussie, D. (2010). *Forest resources and challenges of sustainable forest management and conservation in Ethiopia*. London and New York: Earthscan

Temam, A. (2010). *Impact of disturbance on carbon stocks in Harana Bulluk natural forests, Balöe Zone, southest Ethiopia.* (MSc), Hawassa University, Wondo Genet.

Tesema, A. B., Ann, B., & Bo, T. (1993). Useful Trees and Shrubs for Ethiopia: Identification, Propagation and Management for Agricultural and Pastoral Communities.

Tesfaye, M., Bravo, F., Ruiz-Peinado, R., Pando, V., & Bravo-Oviedo, A. (2016a). Impact of changes in land use, species and elevation on soil organic carbon and total nitrogen in Ethiopian Central Highlands. *Geoderma, 261*(Supplement C), 70-79. doi: https://doi.org/10.1016/j.geoderma.2015.06.022

Tesfaye, M. A., Bravo-Oviedo, A., Bravo, F., & Ruiz-Peinado, R. (2016b). Aboveground biomass equations for sustainable production of fuelwood in a native dry tropical afro-montane forest of Ethiopia. *Annals of Forest Science, 73*(2), 411-423. doi: 10.1007/s13595-015-0533-2

Tiessen, H., Feller, C., Sampaio, E. V. S. B., & Garin, P. (1998). Carbon sequestration and turnover in semiarid Savannas and dry forest. *Climatic Change, 40*(1), 105-117. doi: 10.1023/a:1005342932178

Tilahun, B. E., & Fayera, S. (2012). *Impacts of deforestation on rural livelihoods.* (Masters of Arts), Addis Ababa University, Addis Ababa

Tolessa, T., Senbeta, F., & Abebe, T. (2017a). Land use/land cover analysis and ecosystem services valuation in the central highlands of Ethiopia. *Forests, Trees and Livelihoods, 26*(2), 111-123. doi: 10.1080/14728028.2016.1221780

Tolessa, T., Senbeta, F., & Kidane, M. (2017b). The impact of land use/land cover change on ecosystem services in the central highlands of Ethiopia. *Ecosystem Services, 23*, 47-54. doi: https://doi.org/10.1016/j.ecoser.2016.11.010

Tschakert, P. (2004). The costs of soil carbon sequestration: an economic analysis for small-scale farming systems in Senegal. *Agricultural Systems, 81*(3), 227-253. doi: http://dx.doi.org/10.1016/j.agsy.2003.11.004

Tschakert, P., Khouma, M., & Sène, M. (2004). Biophysical potential for soil carbon sequestration in agricultural systems of the Old Peanut Basin of Senegal. *Journal of Arid Environments, 59*(3), 511-533. doi: http://dx.doi.org/10.1016/j.jaridenv.2004.03.026

Ubuy, M. H., Gebrehiwot, K., & Raj, A. J. (2014). Biomass estimation of exclosure in the Debrekidan watershed, Tigray region, northern Ethiopia. *International Journal of Agriculture and Forestry, 4*(2), 88-93. doi: 10.5923/j.ijaf.20140402.07

UN. (1992). United Nations Framework Convention on Climate Change. 1992. *Available on the Internet: http://www. unfccc. de.*

UNEP DTU. (2015). *CDM pipeline overview*. Copenhagen, Danish Technical University, Copenhagen:

US Global Change Research Program. (2009). *Global climate change impacts in the United States : a state of knowledge report from the U.S. Global Change Research Program*. New York: Cambridge University Press.

Vajpeyi, D. K. (2001). *Deforestation, environment, and sustainable development : a comparative analysis*. Westport, Conn. [u.a.]: Praeger.

van der Ent, R. J., Savenije, H. H. G., Schaefli, B., & Steele-Dunne, S. C. (2010). Origin and fate of atmospheric moisture over continents. *Water Resources Research, 46*(9). doi: doi:10.1029/2010WR009127

Van der Ploeg, S., & De Groot, R. (2010). The TEEB Valuation Database–a searchable database of 1310 estimates of monetary values of ecosystem services. *Foundation for Sustainable Development, Wageningen, The Netherlands*.

van der Werf, G. R., Morton, D. C., DeFries, R. S., Olivier, J. G. J., Kasibhatla, P. S., Jackson, R. B., . . . Randerson, J. T. (2009). CO2 emissions from forest loss. *Nature Geosci, 2*(11), 737-738. doi: http://www.nature.com/ngeo/journal/v2/n11/suppinfo/ngeo671_S1.html

Vashum, K. T., & Jayakumar, S. (2012). Methods to estimate above-ground biomass and carbon stock in natural forests-a review. *J. Ecosyst. Ecogr, 2*(4), 1-7. doi: http://dx.doi.org/10.4172/2157-7625.1000116

Vedeld, P., Angelsen, A., Bojö, J., Sjaastad, E., & Kobugabe Berg, G. (2007). Forest environmental incomes and the rural poor. *Forest Policy and Economics, 9*(7), 869-879. doi: https://doi.org/10.1016/j.forpol.2006.05.008

Vira, B., Wildburger, C., & Mansourian, S. (2015). *Forests and food : addressing hunger and nutrition across sustainable landscapes*. Cambridge, UK: Cambridge.

Virgós, E. (2001). Relative value of riparian woodlands in landscapes with different forest cover for medium-sized Iberian carnivores. *Biodiversity & Conservation, 10*(7), 1039-1049. doi: 10.1023/a:1016684428664

Walkley, A., & 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*(1), 29-38.

Wang, L., Okin, G. S., Caylor, K. K., & Macko, S. A. (2009). Spatial heterogeneity and sources of soil carbon in southern African savannas. *Geoderma, 149*(3), 402-408. doi: https://doi.org/10.1016/j.geoderma.2008.12.014

Wang, Q., Li, Y., & Wang, Y. (2011). Optimizing the weight loss-on-ignition methodology to quantify organic and carbonate carbon of sediments from diverse sources. *Environmental Monitoring and Assessment, 174*(1), 241-257. doi: 10.1007/s10661-010-1454-z

Wang, X., Dong, X., Liu, H., Wei, H., Fan, W., Lu, N., . . . Xing, K. (2017). Linking land use change, ecosystem services and human well-being: A case study of the Manas River Basin of Xinjiang, China. *Ecosystem Services, 27*, 113-123. doi: https://doi.org/10.1016/j.ecoser.2017.08.013

Wang, Y., Gao, J., Wang, J., & Qiu, J. (2014). Value assessment of ecosystem services in nature reserves in Ningxia, China: A response to ecological restoration. *PloS one, 9*(2), e89174. doi: https://doi.org/10.1371/journal.pone.0089174

Wang, Z., Wang, Z., Zhang, B., Lu, C., & Ren, C. (2015). Impact of land use/land cover changes on ecosystem services in the Nenjiang River Basin, Northeast China. *Ecological Processes, 4*(1), 11. doi: 10.1186/s13717-015-0036-y

Wang, Z., Zhang, B., Zhang, S., Li, X., Liu, D., Song, K., . . . Duan, H. (2006). Changes of land use and of ecosystem service values in Sanjiang Plain, Northeast China. *Environmental Monitoring and Assessment, 112*(1), 69-91. doi: 10.1007/s10661-006-0312-5

Wassie, A., Teketay, D., & Powell, N. (2005). Church forests in north Gonder administrative zone, northern Ethiopia. *Forests, Trees and Livelihoods, 15*(4), 349-373. doi: 10.1080/14728028.2005.9752536

Watson, C., Mourato, S., & Milner-Gulland, E. J. (2013). Uncertain emission reductions from forest conservation: REDD in the Bale Mountains, Ethiopia. *Ecology and Society, 18*(3). doi: 10.5751/ES-05670-180306

Watson, R. T. (2000). *Ipcc special report on land use, land-use change, and forestry*: IPCC.

Watson, R. T., Noble, I. R., Bolin, B., Ravindranath, N., Verardo, D. J., & Dokken, D. J. (2000). Land use, land-use change and forestry. A special report of the Intergovernmental Panel on Climate Change (IPCC). *Cambridge: Cambridge University.*

WBISPP. (2000). *Manual for woody biomass inventory*. Addis Ababa, Ethiopia: Woody Biomass Inventory and Strategic Planning Project, Ministry of Agriculture.

WBISPP. (2005). *Woody Biomass Inventory and Strategic Planning Project, A strategic plan for the sustainable development, conservation and management of the woody biomass resources* The Federal Democratic Republic of Ethiopia Ministry of Agriculture Addis Ababa, Ethiopia.

Wei, Y., Li, M., Chen, H., Lewis, B. J., Yu, D., Zhou, L., . . . Dai, L. (2013). Variation in carbon storage and its distribution by stand age and forest type in Boreal and Temperate forests in northeastern China. *PLOS ONE, 8*(8), e72201. doi: 10.1371/journal.pone.0072201

Werth, D., & Avissar, R. (2002). The local and global effects of Amazon deforestation. *Journal of Geophysical Research: Atmospheres, 107*(D20), LBA 55-51-LBA 55-58. doi: 10.1029/2001JD000717

Whiteman, A., Wickramasinghe, A., & Piña, L. (2015). Global trends in forest ownership, public income and expenditure on forestry and forestry employment. *Forest Ecology and Management, 352*, 99-108. doi: https://doi.org/10.1016/j.foreco.2015.04.011

Wilson, M. A., & Carpenter, S. R. (1999). Economic valuation of freshwater ecosystem services in the United States: 1971–1997. *Ecological Applications, 9*(3), 772-783. doi: 10.1890/1051-0761(1999)009[0772:EVOFES]2.0.CO;2

Wittmann, F., Schöngart, J., Montero, J. C., Motzer, T., Junk, W. J., Piedade, M. T. F., . . . Worbes, M. (2006). Tree species composition and diversity gradients in white-water forests across the Amazon Basin. *Journal of Biogeography, 33*(8), 1334-1347. doi: doi:10.1111/j.1365-2699.2006.01495.x

Woollen, E., Ryan, C. M., & Williams, M. (2012). Carbon stocks in an African woodland landscape: Spatial distributions and scales of variation. *Ecosystems, 15*(5), 804-818. doi: 10.1007/s10021-012-9547-x

Woomer, P. L., Touré, A., & Sall, M. (2004). Carbon stocks in Senegal's Sahel Transition Zone. *Journal of Arid Environments, 59*(3), 499-510. doi: http://dx.doi.org/10.1016/j.jaridenv.2004.03.027

World Bank. (2010). State and trends of the carbon market 2010. In: World Bank.

Wright, A. F., & Bailey, J. S. (2001). Organic carbon, total carbon, and total nitrogen determinations in soils of variable calcium carbonate contents using a Leco CN-2000 dry combustion analyzer. *Communications in Soil Science and Plant Analysis, 32*(19-20), 3243-3258. doi: 10.1081/CSS-120001118

Wu, J., & Hobbs, R. (2002). Key issues and research priorities in landscape ecology: An idiosyncratic synthesis. *Landscape Ecology, 17*(4), 355-365. doi: 10.1023/a:1020561630963

Wubet, T., Kottke, I., Teketay, D., & Oberwinkler, F. (2003). Mycorrhizal status of indigenous trees in dry Afromontane forests of Ethiopia. *Forest Ecology and Management, 179*(1–3), 387-399. doi: http://dx.doi.org/10.1016/S0378-1127(02)00546-7

Yami, M., Mekuria, W., & Hauser, M. (2013). The effectiveness of village bylaws in sustainable management of community-managed exclosures in Northern Ethiopia. *Sustainability Science, 8*(1), 73-86. doi: 10.1007/s11625-012-0176-2

Yang, J., McBride, J., Zhou, J., & Sun, Z. (2005). The urban forest in Beijing and its role in air pollution reduction. *Urban Forestry & Urban Greening, 3*(2), 65-78. doi: https://doi.org/10.1016/j.ufug.2004.09.001

Yemiru, T., Roos, A., Campbell, B., & Bohlin, F. (2010). Forest incomes and poverty alleviation under participatory forest management in the Bale Highlands, Southern Ethiopia. *International Forestry Review, 12*(1), 66-77.

Yemishaw, Y., Teketay, D., Worku, A., & Yohannes, Y. (2008). Gathering storms: the fate of forest research in Ethiopia. *the Proceeding of the Ethiopian Forestry at Crossroads: the Need for strengthened institution*, 11-37.

Yesuf, M., Mekonnen, A., Kassie, M., & Pender, J. (2005). *Cost of land degradation in Ethiopia: A critical review of past studies*. Addis Ababa, Ethiopia: EDRI/EEPFE.

Yismaw, A., Gedif, B., Addisu, S., & Zewudu, F. (2014). Forest cover change detection using remote sensing and GIS in Banja district, Amhara region, Ethiopia. *International Journal of Environmental Monitoring and Analysis, 2*(6), 354-360. doi: doi: 10.11648/j.ijema.20140206.19

Yohannes, G., & Waters-Bayer, A. (2007). *Trees are our backbone : integrating environment and local development in Tigray Region of Ethiopia* (Vol. 145). London: International Institute for Environment and Development.

Yoshida, A., Chanhda, H., Ye, Y.-M., & Liang, Y.-R. (2010). Ecosystem service values and land use change in the opium poppy cultivation region in Northern Part of Lao PDR.

Acta Ecologica Sinica, 30(2), 56-61. doi: https://doi.org/10.1016/j.chnaes.2010.03.002

Yuan, F., Sawaya, K. E., Loeffelholz, B. C., & Bauer, M. E. (2005). Land cover classification and change analysis of the Twin Cities (Minnesota) Metropolitan Area by multitemporal Landsat remote sensing. *Remote Sensing of Environment, 98*(2), 317-328. doi: https://doi.org/10.1016/j.rse.2005.08.006

Zegeye, H. (2017). Major drivers and consequences of deforestation in Ethiopia: implications for forest conservation. *Asian Journal of Science and Technology, 8*(8), 5166-5175.

Zeleke, G., & Hurni, H. (2001). Implications of land use and land cover dynamics for mountain resource degradation in the northwestern Ethiopian highlands. *Mountain Research and Development, 21*(2), 184-191. doi: 10.1659/0276-4741(2001)021[0184:IOLUAL]2.0.CO;2

Zhang, B., Xie, G.-d., Li, N., & Wang, S. (2015a). Effect of urban green space changes on the role of rainwater runoff reduction in Beijing, China. *Landscape and Urban Planning, 140*, 8-16. doi: https://doi.org/10.1016/j.landurbplan.2015.03.014

Zhang, M., Huang, X., Chuai, X., Yang, H., Lai, L., & Tan, J. (2015b). Impact of land use type conversion on carbon storage in terrestrial ecosystems of China: A spatial-temporal perspective. *Scientific Reports, 5*, 10233. doi: 10.1038/srep10233

Zhang, Y., Duan, B., Xian, J., Korpelainen, H., & Li, C. (2011). Links between plant diversity, carbon stocks and environmental factors along a successional gradient in a subalpine coniferous forest in Southwest China. *Forest Ecology and Management, 262*(3), 361-369. doi: https://doi.org/10.1016/j.foreco.2011.03.042

Zhao, B., Kreuter, U., Li, B., Ma, Z., Chen, J., & Nakagoshi, N. (2004). An ecosystem service value assessment of land-use change on Chongming Island, China. *Land Use Policy, 21*(2), 139-148. doi: https://doi.org/10.1016/j.landusepol.2003.10.003

Zhao, K., Popescu, S., & Nelson, R. (2009). Lidar remote sensing of forest biomass: A scale-invariant estimation approach using airborne lasers. *Remote Sensing of Environment, 113*(1), 182-196. doi: https://doi.org/10.1016/j.rse.2008.09.009

Zheng, D., Rademacher, J., Chen, J., Crow, T., Bresee, M., Le Moine, J., & Ryu, S.-R. (2004). Estimating aboveground biomass using Landsat 7 ETM+ data across a managed landscape in northern Wisconsin, USA. *Remote Sensing of Environment, 93*(3), 402-411. doi: https://doi.org/10.1016/j.rse.2004.08.008

APPENDICES

Appendix A: Questionnaire on drivers of forest cover change and perception of the community.

Questionnaire Number/code: ______________

Name of the interviewer...
Signature........................

Survey Area: District:__________ Kebele: ________ Village: _______ Distance from the forest______________

Location (coordinates) X _________________, Y_________ Elevation

Date of interview: Day: ______ Month _______ Year: _________

1. **Personal information**

1.1. Name of household head: ____________________

1.2. Respondent's name (if different from the head):

____

1.3. Gender of head 1) M= .... 2) F= ....

1.4. Age of respondent ....

1.5. Educational status (year of schooling) _______________

If illiterate record zero; if literate record one

1.6. House hold family size: ....

1.7. Land holding size: ______________________

1.8. Mean household income:__________________

2. What are the major uses of forests in your area?

3. .Do you think that deforestation is the major problem in your locality?

4. .How is today's coverage of the forest when compared to the conditions before 1985?

A. Declined B. Increased C. No change

5. .According to your knowledge is severe and rapid forest cover change observed today?

A. yes B. No

6. .If the answer to question number '4' is yes, what were/ are the major causes of deforestation? (Put in order)

Drivers	Rank
Cultivated land expansion	
Fuelwood	
Charcoal production	
Grazing land	
Housing	
Drought	
Cutting trees to get rid of wild animals	
Wildfire	
Income generation	
Population growth	
Settlement	
Road access	
Civil war and conflict	
Market access	
Land tenure	
Rainfall variability	

7. What is your major source of income?

A. Sale of cash crops B, Sale of wood and charcoal C, Other__________

8. What types of fuel do you use for household needs (List them in order).

Description	Before 1985	Between 1985 to 2000	From 2000 to present
1. Forest trees			
2. Crop residue			
3. Cow Dung			
4. Charcoal			
5. Kerosene			

9. On the basis of your knowledge, what are the impacts of deforestation/forest cover change in the area? (Put in order).

10. Are there species of "trees" and wild animals, in danger of extinction due to forest cover change from the local region? Please mention if any?

11. What do you think about the possible solution to alleviate the current problem of deforestation and to use forest resources in a sustainable manner?

12. What are the existing efforts to reduce deforestation and forest degradation in the study region?

13. What are the challenges in implementing the efforts to reduce deforestation and forest degradation in the study region/area?

Appendix B: Land cover types and biome equivalents with the corresponding value coefficients.

LULC categories	Equivalent biome	Ecosystem service coefficient (US$ ha^{-1} yr^{-1})
Dense forest	Tropical forest	1775
Open forest	Tropical forest	1775
Grassland	Grass/rangelands	447
Cultivated land	Cropland	463
Bare lands	Desert	0

Source: Computed from the Economic of Ecosystem and Biodiversity valuation database (TEEB) (Van der Ploeg and De Groot, 2010) and (Kindu et al., 2016).

Appendix C: Details of annual value coefficients for ecosystem service functions of each land cover type.

Ecosystem services	Ecosystem service values (US$ ha^{-1} yr^{-1}) of land cover types			
	Dense forest	Open forest	Grassland	Cultivated land
Provisioning services				
Water supply	41	41	72	
Food production	57	57	159	310
Raw material	110	110	21	43
Genetic resources	103	103	18	51
Regulating services				
Water regulation	15	15	7	
Waste treatment	201	201	50	
Erosion control	320	320	23	5
Climate regulation	482	482	51	20
Biological control	5	5	30	30
Gas regulation	15	15	10	
Disturbance regulation	53	53		
Cultural services				
Cultural	11	11		
Recreation	362	362	6	4
Total	1775	1775	447	463

Source: Computed from the Economic of Ecosystem and Biodiversity valuation database (TEEB) (Van der Ploeg and De Groot, 2010) and (Kindu et al., 2016).

Appendix D: Allometric equations used for biomass calculation.

Woody species	Dependent variable	Allometric equation	*Unit*	r^2	References
Juniperus procera	AGB	$AGB = 1.12 \times DBH^{1.54}$	Kg	0.95	(Solomon *et al.*, 2017)
Acacia abyssinica	AGB	$AGB = 0.55 \times DBH^{1.89}$	Kg	0.97	(Solomon *et al.*, 2017)
Acacia etbaica	TDW	$Ln\ totWt = 2.11 + 2.19 \times LnDSH$	Kg	0.96	(Ubuy *et al.*, 2014)
Euclea shimperi	TDW	$Y = 63.07 \times DSH^{1.78}$	g	0.95	(Cleemput *et al.*, 2004)
Otostegia integrifolia	TDW	$Y = 45.80 \times DSH^{2.26}$	g	0.99	(Cleemput *et al.*, 2004)
Other shrub sps.	TDW	$Y = (0.3197 \times DSH) + (0.0383 \times DSH^{2.6})$	Kg	0.93	(WBISPP, 2000)

Appendix E: Species identified in 71 plots and average (±standard error) and their respective measured vegetation parameters in Wujig Mahgo Waren forest, northern Ethiopia

Scientific name	DBH (cm)	H (m)	Biomass (kg)
Abutilon longicuspe Hochst.exARich	5.39±0.73	3.23±0.28	4.91±1.6
Acacia abyssinica Hochst. ex Benth	11.3±0.61	4.18±0.17	33.47±4.3
Acacia etbaica Sihweinf.	7.81±0.59	3.12±0.22	10.30±1.7
Acacia seyal Del.	21.0±3.0	7.20±1.20	99.51±32.7
Acacia tortilis (Forssk.) Hayne	12.65±2.76	4.98±0.25	35.86±17.9
Acokanthera schimperi (A.DC.) Schweinf	9.8±2.84	3.71±0.40	29.76±18.3
Allophylus macrobotrys Gilg	11.7±5.14	5.42±1.57	40.04±33.8
Berberis holstii Engl.	5.77±0.84	3.66±0.45	5.98±2.2
Bersama abyssinica Fresen.	4.01±0.81	2.95±0.38	2.91±1.4
Buddleja polystachya Fresen	3.3±0.85	2.14±0.24	1.48±0.9
Cadia purpurea (Picc.)Ait	3.72±0.05	2.43±0.02	2.02±0.1
Calpurnia aurea (Ait.) Benth.	4.40±0.14	3.39±0.08	3.16±0.3
Carissa spinarum L	4.55±0.19	3.39±0.13	3.45±0.4
Celtis africana Burm.f	8.93±3.35	4.18±0.60	28.16±18.5
Clerodendron myricoides (Hochst.) Vatke	3.46±0.59	2.70±0.52	1.61±0.5
Clutia abyssinica Jaub. & Spach.	2.65±0.35	2.74±0.36	0.84±0.3
Cupressus lusitanica Miller	18.5±0.97	12.63±0.5	81.58±10.2
Discopodium penninervium Hochst.	9.95±1.63	3.65±0.34	20.02±7.1
Dodonaea angustifolia L.f.	4.33±0.09	2.70±0.04	3.27±0.3
Dovyalis abyssinica (A.Rich.)Warb.	4.62±0.70	3.31±0.30	3.96±1.5
Dovyalis verrucosa (Hochst.) Warb.	3.9±0.25	3.38±0.18	2.13±0.3
Ekebergia capensis Sparrm.	12.38±2.38	5.73±0.65	123.6±95.9
Erica arborea L.	5.59±0.36	3.39±0.15	5.27±0.8
Eucalyptus camaldulensis Dehnh	8.14±0.68	6.29±0.47	20.66±4.1
Eucalyptus globlus	11.4±0.85	11.49±0.5	34.50±6.6
Euclea racemosa Murr. subsp. Schimperi	5.38±0.92	2.94±0.40	4.57±1.6
Juniperus procera Hochst.ex.Endl	11.1±0.44	5.14±0.17	40.56±5.7
Maytenus arbutifolia (A.Rich.)Wilczek	5.07±0.13	2.96±0.06	4.74±0.4
Maytenus undata (Thunb.) Blakelock	9.7±0.67	4.67±0.23	19.55±3.6
Myrsine africana L.	3.05±0.14	2.78±0.14	1.09±0.1
Nuxia congesta R.Br.ex.Fresen	8.38±1.43	4.10±0.26	19.64±9.7
Olea europaea L. Subsp. *Cuspidate*	9.88±1.01	3.93±0.22	69.24±18.5
Osyris quadripartita Decn.	4.52±0.24	2.66±0.13	3.07±0.5
Otostegia integrifolia Benth.	2.00±0	0.90±0.30	0.37±0.3
Pittosporum viridiflorum Sims	11.69±0.94	5.27±0.31	34.96±6.2
Podocarpus (Afrocarpus) falcatus (Thun) Mirb.	17.32±4.67	8.09±0.99	419.52±249.4
Psydrax schimperiana (A.Rich.)Bridson	10.00±2.71	4.18±0.52	39.91±28.6

Pterolobium stellatum (Forssk.)	4.64±0.27	4.37±0.32	3.31±0.7
Rhus glutinosa A.Rich.	5.24±0.33	3.61±0.14	5.05±0.8
Rhus natalensis Bernh. ex Krauss	4.42±0.28	3.22±0.17	2.86±0.4
Rosa abyssinica Lindely	4.35±0.78	3.74±0.47	3.10±1.4
Rumex nervosus Steud.ex A.Rich.	2.93±0.58	1.43±0.43	1.04±0.5
Solanum schimperianum Hochst	3.24±0.12	2.30±0.08	1.28±0.1
Teclea simplicifolia (Engl.) Verdoorn	14.41±4.28	6.85±1.83	208.5±116.4
Toddalia asiatica (L.) Lam.	3.33±0.98	9.33±0.33	1.60±1.1