# THE DYNAMICS OF LAND USE MANAGEMENT ON CARBON SEQUESTRATION, SOIL PROPERTIES AND VEGETATION ATTRIBUTES IN BORANA, SOUTHERN ETHIOPIA

**PhD Thesis**

**SUBMITTED TO THE SCHOOL OF PLANT AND HORTICULTURAL SCIENCES, COLLEGE OF AGRICULTURE, HAWASSA UNIVERSITY**

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**KENEA FEYISA JIRATA**

MAJOR ADVISOR: SHELEME BEYENE (PhD. PROF.)

CO-ADVISOR: AYANA ANGASSA (PhD. ASSOC. PROF.)

**HAWASSA, ETHIOPIA**

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# STATEMENT OF THE AUTHOR

I declare that this dissertation is my *bonafide* work and that all sources of materials used for this dissertation have been appropriately acknowledged.

This dissertation has been submitted in partial fulfillment of the requirements for the the degree of Doctor of Philosophy in Plant Sciences with specialization in **Soil Science**at the Hawassa University and is deposited at the University Library to be made available to borrowers under rules of the Library.

I hereby declare that this dissertation has never been submitted to any other institution anywhere for the award of any academic degree, diploma, or certificate. I certify that the published articles presented in chapter two and three have been written by me as first author.

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Name: Kenea Feyisa Jirata

Place: Hawassa University, Hawassa, Ethiopia

Date of Submission: October 2018

Signature

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| | |
|---|---|
| AGB | Aboveground biomass |
| Adj.$R^2$ | Adjusted coefficient of determination |
| AIC | Akaki Information Criterion |
| ANOVA | Analysis of Variance |
| Av.P | Available Phosphorus |
| BMZ | Bundesministerium für wirtschaftliche Zusammenarbeit und Entwicklung |
| CF | Correction factor |
| C:N | Carbon to Nitrogen ratio |
| CEC | Cation Exchange Capacity |
| CDM | Clean Development Mechanism |
| CGIAR | Consultative Group on International Agricultural Research |
| $CO_2$ | Carbon dioxide |
| CA | Crown Area |
| CV | Crown Volume |
| DBH | Diameter at breast height |
| DSH | Diameter at stump height |
| ETM+ | Enhanced Thematic Mapper plus |
| FAO | Food and Agricultural Organization |
| GPS | Geographical Positioning System |
| ICRAF | World Agroforestry Centre |
| ILRI | International Livestock Research Institute |
| IPCC | Inter governmental panel on climate change |

| | |
|---|---|
| Ln | Natural logarithm |
| LULC | Land use /land cover |
| Mg | Mega gram |
| Masl | Mean above sea level |
| MSE | Mean square error |
| MSS | Multi Spectral Scanner |
| NGO | Non-Government Organization |
| PES | Payement for Environmental Services |
| SOC | Soil Organic Carbon |
| SE | Standard error |
| SAS | Statistical Analysis System |
| TM | Thematic Mapper |
| TN | Total Nitrogen |
| TH | Total tree height |
| TLU | Tropical Livestock Units |
| VIF | Variance Inflation Factor |

# PAPERS/ MANUSCRIPTS

This dissertation consists of the following manuscripts in chapter formats:

## Chapter II

Effects of enclosure management on carbon sequestration, soil properties and vegetation attributes in Borana rangelands of southern Ethiopia.

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## Chapter III

Assessment of land use change and its effect on soil organic carbon and total nitrogen stocks in savanna rangelands, southern Ethiopia: Kenea Feyisa, Sheleme Beyene, Ayana Angassa, Ashenafi Burka and Ermias Aynekulu (In review). Geoderma (ELSIVER).

## Chapter IV

Allometric equations for predicting above-ground biomass of selected woody species to estimate carbon in Borana rangelands, southern Ethiopia.

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## Chapter V

Effects of long-term ban of fire on carbon dynamics and soil Properties in Borana rangelands, southern Ethiopia.Kenea Feyisa, Sheleme Beyene and Ayana Angassa (under final review).Catena (ELSIEVIER).

# THE DYNAMICS OF LAND USE MANAGEMENT ON CARBON SEQUESTRATION, SOIL PROPERTIES AND VEGETATION ATTRIBUTES IN BORANA, SOUTHEN ETHIOPIA

## ABSTRACT

*The Borana rangelands of southern Ethiopia have been extensively used as grazing lands by pastoralists for millennia.However, there has been a dramatic shift from prime grazingland to cultivation, left unused because of bush encroachment, and highly degraded leading to different land use/land cover types in the region. Therefore, this thesis work was designed to study the dynamics of land use management on carbon sequestration, soil properties and vegetation attributes by considering the roles played by enclosure management, different land use systems, the aboveground biomass of woody species, and long-term ban of prescribed range fire. This study was conducted in Yabello district of Borana, southern Ethiopia during the period from June-August 2013. A paired-site design approach was used in this study, where sampling plots (30 m x 30m each) with nested' sub-plots for field data collection on soil and vegetation layers were establishedin a systematic random sampling techniquealomga 500 m long transect line in each adjacent experimental site.Using enclosures versus the adjacent open-grazed as control, our results showed that the SOC and TN contents and stocks increased in enclosures as compared to the adjacent open-grazed rangelands, although the differences were not significant ($P > 0.05$) and varied along the age sequence and soil depths.Overall, totalmean SOC stocks of 39.6 ± 3.5 Mg ha$^{-1}$ in the younger (< 20 years old), 40.8 ± 3.4 Mg ha$^{-1}$ in the medium (20–30 years old ) and and 51.0 ± 4.4 Mg ha$^{-1}$ in older (> 30 years old) enclosures age categories, whereas in the adjacent open-grazed areas the values ranged from 34.4 ± 2.5 to 47.9 ± 5.1 Mg ha$^{-1}$in 0-30 cm. The herbaceous biomass was significantly ($P<0.05$) higher inside enclosures (115.4 gm m$^{-2)}$ than that of the adjacent open-grazed rangeland areas (43.6 gm m$^{-2}$).The study in this thesis showed that the Borana rangelands had undergone substantial changes in land use/land cover during the last 37 years. Our results also showed that mean SOC stocks (0-30 cm) in woodland was 55.94±3.41 Mg ha$^{-1}$,while for enclosure, grazing* *and cultivated lands the values were 50.03±3.03, 45.79±4.00 and 38.10±2.39 Mg ha$^{-1}$, respectively. Additionally, woodland had the highest (7.52±0.43 Mg ha$^{-1}$), while cultivated land had the lowest (5.58±0.35 Mg ha$^{-1}$) total nitrogen stock. The potential changes of SOC and TN stocks also showed both gain and loss based on the present measurements and historical land use change. The developed species specific and mixed species allometric equation models for majorities of the investigated woody species ithat related the total above-ground, stem and branches biomass components well fit to the measured dendrometric variablesas as indicated by their adjusted coefficient of determinations and highly significant (adj.$R^2 > 0.80$; $P < 0.001$). The study on long-term of ban of fire on carbon stocks in soil and woody biomass, and TN stock across the two landscape site showe that relatively higher SOC and TN contents and stocks as well as herbaceous biomass carbon in burned than unburned areas, whereas more accumulation of woody biomass carbon was recorded in the unburned sites (40 years of fire exclusion).Overall, this study will contribute to the existing knowledge gaps in terms of the potential of SOC and TN stocks related to different rangeland management practices as well as anestimate of the above-ground woody biomass in arid and semi-arid ecosystems of southern Ethiopia. However, it is suggested further study including other variables such as climatic factors, seasonality and inherent soil properties across wider landscapes, which may have confounding effects on the dynamics of carbon sequestration other than land management practices for the sustainable use of the savanna rangelands of southern Ethiopia.*

**Key words**: Above-ground biomass, Borana rangeland, Carbon dynamics, Enclosure, Fire, Land use change, Soil

# CHAPTER ONE

# GENERAL

This chapter presents general introduction, problem statement and justification, objectives, hypotheses of the study, literature review and general materials and methods.

## 1.1. GENERAL INTRODUCTION

This studyexamined the dynamics of land use management on carbon sequestration, soil properties and vegetation attributes in Borana, southern Ethiopia, by considering the effects of enclosure[1] management, different land use systems, above-ground vegetation biomass and long-term prescribed fire ban.

Though the expressions 'savanna ecosystem' and 'rangelands' are used interchangeably in this thesis, the former refers mostly to the flora and fauna and the latter to the systems of resource exploitation for extensive livestock production in pastoral areas (Angassa, 2007).

Rangelands constitute the largest and most diverse land resources, make up about 50 to 70% of the world's landmass of which over 50% are located in arid and semi-arid regions (Holechek et al., 2005).

On the continent of Africa, savanna rangelands comprise more than 60% of the ecosystems (Neely et al., 2009), and mainly used for pastoral and agro-pastoral activities that support up to 50 million pastoralists and 200 million agro-pastoralists (Jode, 2009).

In Ethiopia, rangeland areas are located within the arid and semi-arid agro-ecology below 1500 meter above sea level (m.a.s.l.), covering about 65% of the total land area and support 12–15% of the country's human population and a large number of livestock (PADS, 2004).

Rangelands, because of their vast extent, hold great potential for carbon sequestration (Reeder and Schuman, 2002; Derner and Schuman, 2007).Infact, rangelands store

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[1] Enclosure is defined as area of rangelands, which is enclosed by a fence with branches of thorny Acacia trees as well as traditional rules to protect vegetation from grazing and/or browsing with the exception of calves and sick animals (Coppock, 1994; Oba, 1998). In some cultures, such enclosed areas may be called "exclosures" (Aerts et al., 2009).

enormous amount of terrestrial carbon stock both globally (36%) and in Africa (59%) (Campbell et al. 2008).

The potential of rangelands to store carbon is reviewed in details in many studies (e.g., Tennigkeit and Wilkes, 2008; Perez-Quenada et al., 2011; Mengistu and Mekuriaw, 2014) and lies within the natural state of rangelands or rangelands moderately disturbed by grazing (Perez-Quezada et al., 2011).The potential of rangelands for carbon sequestration and their managementcan help in mitigating climate change (Neely et al., 2009).

Others (e.g., Reid et al. 2004; Tennigkeit and Wilkes, 2008)suggested reducing poverty among pastoralists through payment for environmental services based on carbon trading.Sequestration of carbon in soil system is also essential for the improvement of soil quality, nutrient rente3ntion and water holding capacity to increase the net primary productivity for more carbon assimilation (Lal, 2015).

The carbon stock in arid and semi-arid rangelands consists of biomass and soil carbon pools (Derner and Schuman, 2007). In these ecosystems, however, the largest pools of carbon are typically found in soils than in the above-ground biomass (e.g. Rau et al., 2010).

Globally, soil stress carbon (C) about three ties as much as in the atmosphereic pool and more than four times as in the biotic (vegetation), sequestered manily in decomposed plant litter and residues(Lal, 2008), and  plays a major role in the global carbon cycle (Lal, 2004).

In this thesis, carbon sequestration is a term describing the process that removes carbon dioxide ($CO_2$) from the atmosphere through photosynthesis and stores as carbon in biomass (tree trunks, branches, foliage and roots) and soils (Izaurralde et al., 2001). In Kyoto Protocol, the term “soil carbon stock” is equivalent to “soil carbon storage”, however, it does not make distinction in the forms of carbon: stable and liable portion (FAO, 2001).

Hence, the carbon fluxes between terrestrial or soil organic carbon and the atmosphere can be positive (sequestration) or negative (emission) depending on anthropogenic alterations of atmospheric carbon dioxide ($CO_2$) (Lal, 2008).

Despite the potentials of rangelands for soil carbon sequestration, heavy grazing pressure and landuse change have contributed to the rapid losses of carbon (C) and nitrogen(N), two important elements linked to climate change (Reid et al., 2008; Liu et al., 2013).

In particular, degradation of pastoral rangelands has been associated with inappropriate land use systems such as overgrazing and frequent use of fire (Neely and de Leeuw, 2011).Grazing and fire modify the structure and function of ecosystems with major effects on soil carbon and nitrogen storages (Piñeiro et al., 2010).

Moreover, a significant land use change happened in the rangelands of Africa in the past few decades due to increased food scarcity and demand for more lands through expansion of cultivation to cope with economic hardships (Hoffman and Vogel 2008; Reid et al., 2014).

Simultaneously, expansion of woody species into savanna rangelands changes the way the rangeland ecosystem cycles carbon (C) and nitrogen (N), mainly through the amount of woody biomass (Liu et al., 2013).An increase in bush encroachment is recognized as a major factor that aggravates degradation of rangeland in dry land areas of the world (Belay et al., 2013).

In general, inappropriate rangeland management and the breakdown of traditional resource management probably contributing to the loss of rangeland ecosystem and vegetation structure as well as loss of soil quality and reduction in carbon stocks (Reid et al., 2004), with a considerable impact on the environment and livelihoods of the pastoral communities (Oba et al., 2000; Angassa, 2014).

The interest in carbon sequestration as mechanisms for both environmental protection and livelihood diversification options in developing countries has increased considerably in the last decades (Perez, et al., 2007).Hence, there is a growing intrest to improve understanding of the current and potential effects of grazing and land management on crbon dynamics (Kibebew and Tessema, 2016).

Several studies (e.g., Reid et al., 2004; Mekuria et al., 2007;Girmay et al., 2008; Abebe et al., 2006) have suggested that soil restoration, vegetation regeneration by use of enclosures and reduced grazing pressureto increase the carbon pools, and can simultaneously achieve carbon sequestration (Trumper et al., 2008).

Enclosure asmanagement and the age sequence is an important strategy for the regeneration and accumulation of above-ground biomass (Abebe et al., 2006; Angassa and Oba, 2010), and bcontribute to enhance the terrestrial carbon storage (WenHong et al., 2010). Lal (2004) also indicated that restoration of degraded land is a key strategy to reduce soil carbon loss.

Prescribedfire in comination with other rangeland restoration technques(i.e. cutting trees followed by fire, cutting trees followed by resting the land etc) also plays an important role to improvetheproductivity of the grass layer(Angassa and Oba, 2008),and this maycontribute increase soil carbon storage.

In contrast, clearing of natural vegetation (bushes) and burning often increase the release of carbon into the atmosphere and thus significantly contributing to climate change (McAlpine et al., 2009).

Nevertheless, the strategies currently being implemented range from the use of enclosuremanagement and fire on soil properties, more specifically on carbon and nitrogen dynamics varied among different ecosystems and regioins depending on climate (precipitation and temperature), soil properties, and vegetation types and landscapes (Nosetto et al., 2006; Pineiro et al., 2010;McSherry and Ritchie; 2013; Booker et al., 2013).

Furhtermore, the complexity of vegetation struacture and changes in land use systems in savanna ecosystems areadditional inadequacies to fully understand whether rangeland ecosystems function as carbon (C) sinks or sources (Perez-Quezada et al., 2011; Angassa et al., 2012b).

Rangelands are generally considered mosaics of diverse ecological conditions created by spatial variation in soils, topography and micro-climate resulting in skewed distribution of carbon storage (Booker et al., 2013; Dabasso et al., 2014).

The Borana rangelands in southern Ethiopia have been extensively used as savanna grasslands traditionally managed with the use of periodic fire until the early 1970s (Coppock, 1994).However, these rangelands have been in a dramatic shift form grass cover to towards a bush dominated savannasince then (Angassa and Oba, 2008).

Moreover, the establishment of permanent water points and accelerated semi-sedentary settlements followed by transformation of the communal grazing lands that involve theintroduction of range enclosures and an increase in crop farming created different land use systems in the region (Angassa et al., 2012b; Elias et al., 2015).

As copying mechanism, the communities have had engaged in livelihood diversification activities.In particular, the mixed grazing approach such as calves with young cattle and goats apparently benefited the local people through subsistence income source generated from hay, firewood and fattened young bulls for market sale (Tache, 2010).

Such efforts may be more sustainable if complemented with the Payments for Environmental Services (PES) through carbon trading. However, terms such as carbon sequestration is are new to pastoral communities in Borana, let alone relating rangeland management practices to payement for carbonsequestration and reduction of carbon emmssion.

Previous studies conducted in the Borana rangelands were mainly focused on vegetation ecology and range condition assessment (e.g., Dalle et al., 2006; Solomon et al., 2007; Angassa and Oba, 2010).

Besides, few recent studies (Hasen-Yusuf et al., 2015; Bikila et al., 2016), who quantified carbon stocks in relatin to different landmanagement use types in Borana of southern Ethiopia are limited to specific sites anddid not focus on the effects of enclosure management along the age chronosequence, while none of them assessed the impacts of land use change and long-term banning of fire on carbondynamics.

Furthermore, presence of limited dataon carbon sequestration potential of arid and semi-arid envirnonments resulted in lack of recognition of the rangelands contributionin an international carbon trading discourseofKyoto compliance market, like the Clean Development Mechanism (CDM) in developing financial incentives to reward carbon emission-reduction land management (Tennigkeit and Wilkes, 2008).

Therefore, the present study was designed to fill this knowledge gap through generating empirical evidencesin terms of potentials of carbon stocks and associated soil properties and vegetation characteristics related to different land use systems in the Borana ranglenads of southern Ethiopia.

It is also important to develop sustainable land management practices to enhance carbon sequestration potential that is specific for arid and semi-arid ecosystemof southern Ethiopia. Therefore, this study aimed at providing answers to the following research problems and hypotheses.

## 1.2. STATEMENT OF THE PROBLEMS

The Borana rangelands of southern Ethiopia used to be among the most productive pastoral areas in East Africa, comprise extensive grazing lands with a unique feature of landscapes and indigenous knowledge on natural resource management (Homann et al., 2008).

The Borana pastoralists managetheir grazing lands in terms of wet and dry seasons grazing systems and separate grazing culture through categorizing the herds according age, sex, lactation and health conditions of the animalsto respond to grazing impacts in a sequential and predictable manner even under the influence of the non-equilibrium environment(Oba, 1998),

Such management system keeps animal numbers below the threshold set by pasture availabilityand signifies “controls of livestock density-dependent of Borana pastoralism”,which partly explains why the Borana rangelands remained in relatively better contion, regardless of their use for many centures (Angassa and Oba, 2007).

For the last four decades, however, the Borana rangelandshave undergone substantial reduction in grassland cover due to bush encroachment , expansion of cultivation, and increased sttlements (Dalle et al., 2006; Solomon et al., 2007; Angassa andOba, 2008),

Particularly, long-term prohibition of range fire, cultivation of bottomlands and continuous grazing on the remaining portion of the communal rangelands have induced the invasion of bush encroachment and degradation in terms of soil and reduced pasture quality (Oba and Kotile, 2001), which is apparent in Yabello district (Elias et al.,2015; Abate and Angassa, 2016).

The subsequentexpansion of invasive woody species resulted in various forms of land degradation in terms of depletion of soil quality and nutrient , and  palatable grass species in these rangelands (Angassa, 2014), with negative consequences on the livelihood of the local communities (Desta and Coppock, 2004).

Thesis research work, therefore, was undertaken as a part of a bigger project “Livelihood diversifying potential of livestock-based carbon sequestration options in pastoral and agro-pastoral systems in Africa”as additional livelihood option and improved management isexpected to significantly increase the carbon sink in dryland ecosystems.

However, there are little or no empirical evidenceson how much carbon is currently and can potentially be stored in the Borana rangelands to provide accurate data for the establishment of PES systems a to diverfify their liveloood option for  their masustainable rangeland managemen Hence the following research questions were addressed under the following researchable themes;

(i) Existence of enclosuresas management and the age sequence (effective years since establishment), particularly in Dida-Hara area isa unique opportunity to explore the important aspect of rangeland management, and information of the impact on soil carbon dynamics in savanna rangelands of southern Ethiopia, which will also provide livelihooddiversifying option to local pastoralists through carbon trading (C**hapter II)**.

(ii) Different land use systems and land use/land cover change in the study areas,in particular,the introduction of crop cultivation and heavy grazing pressure is an intresting dimentions to estimate the potential loss and gain of SOC and TN and stocks at ecosystem level (**Chapter III**).

(iii)The various types of woody species under varied management, landscapes and soil characterstics in the Borana rangelands of southern Ethiopiademanding current and accurate information on woody biomass. However, either species specific or generalized allometric equations for predicting the aboveground biomass are rarely available for the region. Moreover, use of established allometric equations may be source of error, and developing site and species specific allometric equation models is imperative(**Chapter IV**).

(iv) The use of prescribed fire was a common vegetation management tool in the Borana ragelnds of nsouthern Ethiopia, but this practice was discontinued in the 1970's following a ban on bush fire imposed by the government. As a result, the physiognomy of the vegetation of the Borana plateau has changed from extensive grassland to state with greater cover by bushes and woody species, a transformation that has been attributed to the ban of firesince then (Angassa and Oba, 2008).

One of the important impacts of change in vegetation is a significant increase in woody biomass and an associated increase in the amount of carbon sequestered in the woody vegetation. Thus far, there has been no attempt to quantify the impact of long-term fire suppression from the system on the potential ofcarbon sequestration in the below- (soil) and above-ground vegetation biomass across different landscapes of theBorana rangelands (**Chapter V**).

## 1. 3. OBJECTIVES OF THE STUDY AND HYPOTHESIS

The general objective of this thesis work was to investigate the effects of land management practices and disturbances viz., enclosure management, different land use systems, above-ground vegetation biomass and long-term fire ban on carbon sequestration, selected soil properties and vegetation attributes.

**The specific objectives were:**

1) To investigate the effects of grazing management (enclosure versus adjacent open-grazing systems) on soil properties, more specifically SOC and TN contents and stocks as well as vegetation characteristic, while accounting for effects of the age of enclosures and soil depths;

2) To assess land use/land cover change between 1976 and 2013, and its effect on soil properties, more specifically SOC and TN contents and stocks, and estimate changes in SOC and TN stocks due to land use/land cover change;

3) To develop site and species-specific allometric models to estimate the aboveground biomass and its components for most dominant woody species in Borana rangelands of southern Ethiopia and contribute to the accurate estimation of the above-ground biomass and carbon stocks of woody vegetation in the savanna rangelands of East Africa;

4) To investigate the impact of long-term ban of fire (burned versus adjacent unburned areas) on carbon stocks in soil and above-ground woody and herbaceous biomass, while accounting for effects of landscape position (upland and bottomland) and soil depths.

**Research hypotheses**

The current study was based on the hypotheses that that were formulated to test the separate objectives of the individual studies;

i. Enclosure management would favor more accumulation of SOC and TN contents and stocks, and the recovery of herbaceous vegetation than the adjacent open-grazed areas, and the older age enclosures would accumulate more SOC and TN contents and stocks as well as herbaceous biomass than the younger enclosures (**specific objective 1)**;

ii. The magnitude and rates of land use/land cover changes in the study sites is very high with further increase in recent years; soil properties, more spefically SOC and TN contents and stocks significantly varied in different land use types as well as the dynamics in land use brings considerable changes in SOC and TN stocks over the last decades (**specific objective 2)**;

iii. The biomass prediction models for (total above-ground and the components biomass) would vary in different woody species due to variation in their growth form and canopy architecture; Use of two or more of the predicator variables (DBH, DSH, TH, CA and CV) would provide the best goodness of model fit, while it could also make the model complex and sources of bias in some woody species (**specific objective 3)**;

iv. Prescribed fire would increase soil nutrients, more specifically soil organic cabon total nitrogen due to the increase of understory layer and the reduction of the direct release of nutrients in the fire; while unburned areas accumulate more carbon in woody biomass as compared to the burned rangeland units (**specific objective 4).**

## 1. 4. STRUCTURE OF THE THESIS

The thesis has been organized into six chapters;

**Chapter one** provides a general introduction, researcheable problems, objectives, hypothesis, literature review and over all description of the research methodology (description of the study areas, experimental design approaches , data collection and laboratory analysis of soil and vegetation samples , and statistical analysis) used in the study.

**Chapters II through V** wasself-contained and structured in the formats of manuscripts. The findings of the studies in this thesis in the form of papers which have published and or submitted to peer-reviewed international journals for publications**.**Specifically,

**Paper-I-**C**hapterII**presented the results on carbon sequestration, soil properties and vegetation attributes in the enclosures along theage chronosequence and the adjacent open-grazed lands**,**

**Paper-II-ChapterIII** assessed the land use/land cover use change and its effect on soil organic carbon and total nitrogen contents and stocks and changes attributable to land use/land cover change.

**Paper-III-**C**hapterIV** focused on development of allometric equations for predicting the total above-ground biomass and its components for selected woody species dominantly found in Borana rangelands of southern Ethiopia through destructive harvesting of samples of woody plants.

**Paper- IV-**C**hapter V,** we investigated the effects of long-term banning of fire on selected soil properties and carbon sequestration in soil and above-ground vegetation biomass along the two different landscapes.

Finally, **chapter VI** provides the generalsummary, conclusions and recommendations suggested for future research.

## 1. 5. LITERATURE REVIEW

### 1. 5.1. Overview of rangelands and their importance

There are various definitions of rangelands that have been used in reference to diverse vegetation cover, use, management and environments (Lund, 2007). More broadly, rangelands can bedefined ras the land managed as a natural ecosystem with natural vegetation including grasslands, shrub-lands, savannas, tundra, and woodlands that often contain a human dimension implied by management associated with grazing and fire (SRM, 2006).

In Africa, rangelands (savanna ecosystems) constitute about 43% of the total land surface area and are mainly used for pastoral and agro-pastoral activities to support more than250million people , and account for more than 60% of the total landmass of the East African region (Neely and Bunning, 2008).

The natural vegetation of rangelands also provide about 70% of the feed needs of domestic ruminants worldwide and more than 85% of the total feed needs of ruminants in African countries (Lund, 2007). Furthermore, rangelands offer important ecological services ranging from protecting fragile soils and providing habitat for wild fauna and flora (Booker et al., 2013)

In Ethiopia, rangelands areas are located within the arid and semi-arid agro-ecology and most of these areas are found below 1,500 meter above sea level (m.a.s.l.) and estimated to be 61-67% of the country's total area. These areas are located in the east, northeast, southeast, and southern parts of the country which support pastoral and agro-pastoral communities of about 12-15% of the country's total populations (PADS, 2004).

In these areas, livestock plays a central role in the livelihood and social value systems of the pastoral and agro-pastoral communities and supports the livelihoods of about 80% of the rural population, and it contributes 15 to 17% of overall growth domestic product and over 90% the live animal and meat export is sourced from rangeland based livestock production in the pastoral areas of the country (PADS, 2004).

The Borana rangelands in southern Ethiopia, which cover about 95000 $km^2$ is more suitable for extensive livestock production whose gross incomes from livestock keeping are at least 50% (Desta and Coppock, 2002).

In recent decades, these rangelands, however, are in state of transition to different vegetation states (e.g., from a grassland (the first 'state') to a bush land (the second 'state') due to several factors including climate change, recurrent drought,overgrazing and increased human and livesock population and alteration of traditional communal resource (Dalle et al.,2006; Angassa and Oba,2008).

### 1.5.2. Rangeland management systems in Borana, southern Ethiopia

The pastoral communities in Borana traditionally divided the grazing spaces (communal rangelands) into categories of land units each with designated time of access 'temporal use areas'.In this context, rangeland is synonymous with 'grazing land' in the utilization of the natural resources (Coppock, 1994).

In this management system, permanent settlements(villages) are prohibited within 10-15 km radius of the *tula-well* (i.e., non-settlement landscapes) that is usually used by the homestead (*warra*) herds during the dry season, and the area beyond that is grazed by the *dry* (*foora*)herd (Oba, 1998).

The *foora* isused for dry cows, bulls, mature males, oxen and heifers, while the *warra* consists of lactating cows, sick and weak animals and calves, which return to the encampment every day.The separation of dry herd (non-lactating cattle) from the rest of the herd is a key rangeland management strategy towards controlling grazing pressure (Angassa, 2007).

The mobility patterns corresponded with local rainfall and rangeland productivity, shifting towards dry areas in the wet season and humid areas in dry seasons (Oba et al., 2000). The earlier management system also involved periodic burning of the rangelands (Coppock, 1994).

Overall, about 70% of the communal rangelands is used as for open grazing systems, while about less than one third of the total areas were managed for controlled (dry seaon grazed areas(Kamara et al.,2004).

The protected rangelands are locally known as "*kalo*" ,managed at *reera* (*groups of villages)* set aside and keep livestock off from grazing except the calves and sick animls Ad few lactating animals , and such management system protecting paths of degradation and design of sustainable rangeland use(Napier and Desta, 2012)

Nevertheless, the weakening of the traditional community leadership structu, the influx of people, increasing human and livestock populations forced pastoralists to intensifythegrazing land resulted in deterioration of the rangelandsled to overgraqzing and degradation resulting to an ecosystem that can hardly maintain its stability, function and structure (Angassa and Oba, 2008; Bikila et al., 2014).

In general, improved rangeland management systems through enclosure approach for rehabilitation of degraded grazing land that transilate to economic benefits for the local people (Wairore et al., 2015).

### 1.5.3. Major challenges in the Borana rangelands of southern Ethiopia

#### *1. 5.3.1. Rangeland (soil) degradation*

Rangelands are sensitive ecosystems which could be adversely affected by inappropriate land uses. Globally, about 10–20% of drylands, and 31% of African rangelands soils suffer from recent degradation as well as losses of soil carbon due to intensive grazing and agricultural practices (MEA, 2005; Trumper et al., 2008).

The term “soil degradation” and “land degradation” is generally used interchangeably in many literatures and refers to the reduction of both soil and vegetation of interest to humans and useful to nature’s functions (FAO, 2011).

Recently, rangeland areas, however, are under state of degradation due to many factors, among others, deforestation, overgrazing, poor land management and increasing pressure on the rangeland resources (Kiage, 2013).

In pastoral rangelands, overgrazing accounts for more than 30% of all forms of land degradation (Vågen et al., 2005; Wehrden et al., 2012) and depletion of soil organic carbon (SOC) associated with intensification of grazinglands, the dominant features in the Borana rangeland (Elias et al., 20015).

Besides, rangeland areas are experiencing increasing population growth rate, livestock changes, under varying climatic conditions and divergent environmental policies (Abate and Angassa, 2016).

The continued degradation of rangelands in terms of soil and vegetation is becoming a serious problem with considerable impact on the environment and livelihoods of the pastoral communities (Oba et al., 2000; Angassa, 2014).

Therefore, restoration of degraded rangeland soils by means of protecting the natural vegetation resources (woodland), and improved grazing management practices such as the use of enclosure are important strategies to minimizing losses of soil carbon and nutrient (e.g. nitrogen) and improve the environment (FAO,2011).

#### *1.5.3.2. Expansionush bush encroachment*

The history of Borana in southern Ethiopia tells that the region is dominated by open grassland savannas containing mixtures of perennial herbaceous and woody vegetation until 1970s and being transformed into thick bushes since then (Oba, 1998).

The major woody plant species which are considered to be encroachers are *Acacia brevispica (*Hammaresa*), A.bussie (*Hallo*), A. drepanolobium (*Fullensa*), A. melifera (*Saphansa*), A. reficiens (*Sigirsa*), A. seyal (*Wachu*)* and *Commiphora africana (H*ammessa*)* are the dominant species that make up major plant communities of the area (Dalle et al., 2005).

By the mid-1980s, about 40% Borana rangelands were affected by bush encroachment (Coppock, 1994) and further increased to 52% around the year 2004 (Dalle et al., 2006). Moreover, Hasen-Yusuf (2013) reported considerable increase of bush cover from 22 to 61% between 1976 and 2012 in the same region.

This consistent increase of bushes and shrubs cover and density in the region can be attributed to numerous factors such as suppression of fire and increased grazing pressure, exclusion of browsers (Dalle et al., 2006; Angassa and Oba, 2008) and episodic climatic events (Angassa and Oba, 2007)

In line with, previous studies that have been carried in Borana rangelands (Angassa et al., 2012a; Bikila et al., 2014)showedthat controlling of invasive bush species by cutting increased herbaceous biomass on thinned grazing lands as compared to non-thinned rangeland units.

Furhtermore, use of prescribed fire in combination with other management technqunes such as bush cutting also suggested asbest approach in terms of improving herbaceous biomass and suppression of eencoracher invasive bush tree species (Angassa et al., 2012a; Bikila et al.,2014).

Hence, bush control through treecutting has become ubiquitous activitsy in Borana that government and many non-governmental orgnaqizations NGOs) provide cash for work and food aid for bush clearing (Bikila et al., 2014).

In contrast, clearing of natural vegetation (bushes) often increases the release of carbon into the atmosphere that will contribute to climate change (McAlphine et al., 2009). As a result, there is a growing intrest to understand dynamics of carbon changes in relation woody encroacgment in the dry savannas (Aragao et al., 2014; Grace et al., 2014).

Furthermore, to balance the rangeland economic services loss (grass quality, quantity, and livestock production), with environmental services gain (carbon storage, soil fertility, and animal diversity) from bush expansion, development of sustainable and sound rangeland management policy is imperative(González-Roglich et al., 2014).

Thus, the increase of woody cover and denswity in Borana rangelands has a potenal to increase the carbon accured in above-ground woody vegetation biomass, which will foster furture carbon trade disccsuins with respect to strategies in mitigating climate change.

However, little no experinces exist about the integration of management of busg encroachment into relvant national and community based natural resources management strategies and policies in order into integrate the payement for envrinmentl services through carbon trading in mitigating the climate change.

#### *1.5.3.3. Change in land use patterns*

The savanna rangelands in southern Ethiopia comprise extensive rangelands with a unique feature of landscapes and supports pastoralism, small scale farming and other ecosystem functions (Homann et al., 2008), and being recognized long times for their resilient ecosystems in East Africa (Oba, 1998).

However, since the last four decades these rangelands have been experiencing considerable changes in patterns of land uses (Oba et al., 2000). Particualry, from 1980s on wards Increasing human and livestock populations, changes in fire regimes, expansion of crop production were the major transitions resulted in an ecological and economic loss of productivity of the communal rangelands in southern Ethiopia (Angassa and Oba, 2008; Haile et al., 2010

In the region, reliance on the cmmual resource pools was gradyually restricted; whereas private laqnd sue (private enclosure and farmlands) were increased. For example, only 5% of the total area of the Borana lowlands was covered by crop cultivation during the 1980s, whereas the total lands allocated to crop production was increased to about 16% after 20 years (Kamara et al., 2004).

A few recent study (e.g., Elias et al., 2015) aslo showed that there was a significant increase in the proportion of cultivated land from 8.43% in 1985 to 20.92% in 2011. However, there are few pocket areas in Borana rangelands with relatively wetter areas receiving relatively good precipitation that can support rain-fed crop production (teff, sorghum, barley, and haricot beans).

The main reasons contributing to the rapid conversion of savanna rangelands to marginal croplands were losses of pasture and declined in livestock prodtivity and economic hardships (Abate and Angasa, 2016). Moreover, the Ethiopiangovernment policyalso encourages dryfarming to cope up with the economic hardships of the local people (Tsegaye et al., 2010; Abate and Angassa, 2016).

This type of shifts in natural state of rangelands, however, accelerates soil erosion and deterioration of randgelsnds and loss of soil carbon.However, recent information on the spatial and temporal changes in land use and its effect on the status of soil and nutrient dyanmics is rarely available to design sustaianable use of rangeland resources and to enhance carbon sink potential in the systems

### 1.5.4. Strategies to improve rangeland management

#### *1.54.1. Enclosure as management strategy to enhance carbon sequestration*

Establishment of seasonal grazing enclosure has become an important rangelands rehabilitation strategy,which is enclosed by a fence with branches of thorny bushes of Acacia species and traditional rules to protect vegetation from grazing /or browsing with the exception of calves and sick animals (Angassa and Oba, 2010; Mureith et al, 2014).

The semi-private enclosures are widely practiced by pastoralists in East Africa for dry season grazing (Verdodt et al., 2009; Angassa and Oba 2010). In Borana, use of enclosures locally known as *kaloos*was the gradual shifts from the traditional calf reserves (*lafa seera yaabi*) to the communal enclosures ranging from tens to hundreds ha to cope up the periodic shortages of forage, especially for calves ((Napier and Desta, 2012).

In fact, the size of enclosures is determined by the number of vulnerable animals (calves, lactating cows etc.) and their use is restricted to members of the community that built the fences, usually one or more *olla's* (encampment).

Numerous studies in different locations of Ethiopia (e.g., Abebe et al., 2006; Yayneshet et al, 2009) showed the importance of enclosurein tems of vegetation regeneration and aboveground biomass.Other studies(e.g., Mekuria et al., 2007; Haftay et al., 2013) alsoreported thar enclosure a management enhanced carbon pools as compared to that of the adjacent open grazing lands.

Enclosure management through controled grazing allows vegetation regeneration , which has a positive effect on biodiversity (Abebe et al.,2006), soil fertility restoration (Mekuria et al.,2007), but varied depending on local biophysical and management conditions (Haftay et al., 2013; Wairore et al., 2015).

Soil nutrients are essential for the normal growth in rangelands and development of plants, and the growth, maintenance and productivity of herbivores (Teka et al., 2012).The communities in the southern Ethiopis are restoring indigenous vegetation inside enclosures in an effort to combat severe land degradation and address their livelihood problems through income source generated from hay, firewood and fattened young bulls for market sale (Tache, 2010).

Thus, enclosures offere opportunity of being integrated in carbon sequestration projects funded by International organizationsor governments (Olsson et al., 2001; Deng et al., 2014), which could allow large areas of extensivegrazinglands ofotherwise degraded and/or cultivated to revert to semi-natural vegetatiion

#### *1.5.4.2. The role of prescribedfire in rangeland management and carbon dynamics*

Fire is a key environmental driver that controls the function of savannah ecosystems. In southern Ethiopia, the historical role of fire in the management has been widely documented(e.g., Coppock, 1994; Dalle et al., 2006).

It was a tool used by Borana pastoralists to suppress bush growth by killing encroaching woody species, stimulate fresh grass growth, and to help control tick populations (Angassa and Oba, 2008). However, a ban on range burning was implemented as of mid-1970s with the goal of safeguarding the natural forests against wildfires (Angassa and Oba, 2008).

The reason for the fire ban was the official conservation policy linked to the loss of forest cover in the Ethiopian highlands (Reid et al., 2004), and was perceived as the major factor that caused encroachment of woody plants in the Borana lowlands (Coppock, 1994). As a result, the physiognomy of the vegetation of the Borana plateau changed from extensive grassland to state with greater cover by bushes and woody species.

Over the last 30 years, the traditional use of fire was no longer possible as mentioned by pastoralists and development staff alike, are that the fuel load (ground cover) is now so patchy that it can be difficult to get a fire started and maintained. Moreover, the proliferation of villages in the area could constitute a hazard should fires get out of control (Angassa and Oba, 2008).

Since fire had been used traditionally and also preferred the idea of this rangeland management tool as it echoed one of their own practices with associated good results to revive range burning received positive attention by the policy makers (LaMalfa et al., 2008).

In a practical study conducted to compare the merits of different rangeland management systems, Angassa found that combining fire with grazing achieved a restoration of herbaceous plant diversity and suggested as an effective strategy in restoring a healthy grass understory in Borana rangelands(Angassa, 2007) , and this could help to enhance soil carbon storage.

Precautions would have to be taken, however, to ensure that burning is kept within prescribed areas as not all areas are appropriate (Gebru et al., 2007). Bearing these factors in mind, NGOs are re-introducing burning as a rangeland management technique through ‘prescribed fire’-the controlled and managed application of fire to defined units of rangelands (Gebru et al., 2007).

It is, therefore, imperative to investigate the impact of long-term fire ban versus the re-introduction of prscribed fire on carbon in below-(soil) ansd above-ground vegetation biomass that will improve our knowledge on the changing ecologyof Borana rangelands and to design effect management of the savanna ecosystems.

#### *1.5.4.3. Role of allometric equations for estimating above-ground biomass*

Estimation of above-ground biomass has received significant attention in recent years because of the fact that the change in above-ground biomass is associated with the potential of woody vegetation to sequester and store carbon dioxide as components of climate change mitigation strategy (Lu et al., 2002).

Above-ground biomass is commonly divided into three major components: bole (main stem), stem bark, and crown (branch and foliage) and each component require different models. Moreover, different tree components are used for different purposes requiring separate estimates of component biomasses (Houghton, 2005).

Woody plants species encroachment into grasslands and savannas play dual role as the carbon sink by removing carbon dioxide through photosynthesis, converting that photosynthetic to vegetation biomass, and as a carbon source by releasing carbon dioxide through respiration, wildfires, and decomposition (Stephen et al., 2009) .

This opportunity is vital in Borana rangelands of southern Ethiopia, where woody tree density is consistently increasing from about 40% in middle of 1980s (Coppock, 1994) to more than 60% (Hasen-Yusuf, 2014). The consistent increase of bush and shrubs density resulted in shifts in the balance between woody plants and grass species (Anagssa et al., 2012a).

However, direct measurement of tree biomass that involves harvesting and weighing trees is time consuming, costly and impractical method, especially when dealing with numerous species and large sample areas. For this, allometric equations are useful tools for accurate estimate of biomass (Lu et al., 2002).

Thus, allometric equation whichrefers to a regression model that fitted the total tree biomass and its components (e.g., stems, branches, leaves or roots) to easily measurable independent variables such as diameter at breast height (e.g., DBH), (Basuki et al., 2009).

Thus, the dendrometric parameters of all of the trees are measured and the allometric equation is then used to estimate the stand biomass by summing the biomass of individual trees (Kuyah et al., 2012). However, there are very a few allometric equations available to estimate the above-ground biomass of woody plant encroachment in this region (e.g., Hasen-Yusuf et al., 2013).

Furthermore, use of generic allometric equations developed somewhere have limited applicability particularly for drylands woody species due to different growth form and architecture (Kuyah et al., 2012).This showed that the sites specific allometric equations are more accurate in predicting the biomass of woody species on the local level as it takes into account the site effects (Yuen et al., 2016).

### 1.5.5. Potentials of rangelands in sequestering carbon

Rangelands are estimated to store greater than 10% of the terrestrial biomass carbon and up to 30% of the global soil organic carbon (Grace et al., 2006; Derner and Schuman, 2007). Furthermore, improved rangeland management practices increased the carbon sequestration potential that raged from 0.35–0.55 Gt-C.yr$^{-1}$by 2030 worldwide (Tennigkeit and Wilkes, 2008) depending on a wide range of factors suchgrazing fire , land use and landscapes (Neely et al., 2009).

Likewise, Ethiopian rangelands, with wide ranges of plant composition and management systems alsohold great potential for carbon sequestration.A few study (e.g., Kibebew and Tessema, 2016) on carbon stocks under different grazing systems in selected pastoral areas of Ethiopiashowed that total SOC stock (mean ± standard deviation) in Somali region rangedfrom 58.622±9.296 to 160.536±31.477t ha$^{-1}$, in Afar region, the mean carbon stock was 29.519±17.771 t ha$^{-1}$.

In the same study indicated above, it was also reported that the total carbon stocks that ranged between 60.962±14.110 and 81.697±20.248 tha$^{-1}$ for Borana zone of Oromia region. A study conducted by Niles et al. (2010) in selected sites of Borana rangeland also reported higher carbon stocks in enclosure as compared to that of the open- grazed areas.

Moreover, numerous reviews and studies on soil carbon sequestration in Ethiopia (Girmay et al., 2008; Sheferaw et al., 2013)also elucidated the potential of increasing SOC pools through different land restorative measures of degraded soils, bur their data are limited to specific sites and mostly focused on highlands.

In general, the carbon (C) input to the soil is related to net primary production (NPP), which in turn varies with climate, land cover, plant species composition and soil type

(Sharma et al., 2012), and hence information on the carbon sequestration potential of Ethiopian rangelands is imperative.

### 1.5.6. Opportunities of carbon sequestration in rangelands

People living in dryland areas faced many challenges, such as climate variability, land land degradation and drought and poverty (IPCC, 2007; Mureithi et al., 2014).On the other hand, arid and semi-arid rangelands are thought to hold potential for carbon sequestration (Tennkgeit and Wikes, 2008)

Therefore, improved land management practices that can enhance carbon sequestration in soil systems can enhance resilience of the ecosystem and provide an important incentive for pastoralists (Neely et al., 2009).

Carbon finance projects in rangelandshave already begun transactions in many countries like USA and Australia, and the potential of rangelands for carbon sequestration deserves significant to contribute to environmental protection and livelihood diersfifcation option in developing countries received attention in recent times (Perez et al., 2007).

The Kyoto Protocol opened the possibility for less developed countries to receive payments for carbon offsets based on land use, and hence African communities may benefit rather than only suffer from these global changes (Tieszen et al., 2004).

Accordingly, there is increasing need to link carbon sequestration in pastoral rangelands to local livelihoods through promotion of carbon-credits and reduce vulnerability associated with climate variability and change (Follett and Reed, 2010).

For example, (Reid et al. (2004) reported that if carbon is valued at $10 per ton and with modest improvement management can gain carbon stock by 0.5 t C ha$^{-1}$ yr$^{-1}$, which translates into \$50 yr$^{-1}$ (with population density in pastoral areas estimated at 10 people km$^{-2}$ or 1 person per 10 ha), and this gain would bring about 15% increases in income, where half of the pastoralists in Africa earn less than \$1 day$^{-1}$ or about \$360 yr$^{-1}$.

In context of Ethiopia, Shiferaw et al. (2013) showed that land restorative measures increased SOC pools that ranged from 1–3.2 Mg ha$^{-1}$ yr$^{-1}$, while land conversion particularly to arable land resulted in depletion of 5.4 Mg ha$^{-1}$yr$^{-1}$.Girmay et al. (2008) also reported about 0.066–0.154 Mg ha$^{-1}$yr$^{-1}$ (0.24-0.55 Mg CO$_2$ e ha$^{-1}$yr$^{-1}$) SOC stocks withimplementation of improved rangeland management practices.

However, carbon finance in rangelands is constrained by regulations of eligibility criteria as suggested in the Kyoto Protocol (Tennigkeit and Wilkes, 2008), among others: (i) the exact rate of carbon stock change per unit area after adoption of the practice, (ii) the time required for new steady-state levels to occur, and (iii) the total area over which the intervention carried out.

Yet, study is needed tofill the knowledge gap in terms enprical evidence on the potential of carbon stored in the system required to develop strategies in carbon sequestration projects. Moreover, there are a number of priority actions that need to be undertaken as the pastoral wasys of land use system based on communal resources pools may constraints in carbon markets trade as livelihood diversification option.

## 1.6. MATERIALS AND METHODS

### 1.6.1. Description of the study area

#### *1.6.1.1. Location and topography*

The study was conducted in Yabello district, located in Borana administrative zone, southern Ethiopia.Yabello district is located in the central part of Borana rangelands, located between 4°30’55.81”and 5°24’36.39”N latitude an d37°44’14.7”and 38°36’05.35”Elongitude, and covers a total area of 5,556 square kilometers.

Borana rangelands (also known as the Borana plateau) cover approximately 95,000 km$^2$ and extends from 4º to 6°N latitude and 36º to 42°E longitude, is slightly undulating and ranging in altitude from 1000 to 1500 m above sea level (m.a.s.l.) with peaks up to 2000 m.a.s.l.

The present studies were carried out across three sites namely: Dida-Hara, Dharito and Obda inYabello district (Fig.1.1).Yabello is the capital town of the Borana zone and located 570 km south of Addis Ababa. The altitude of the study area ranges from 1260 to 1928 meter above sea level, where Obdais a major highland feature in the district.

These sites are located in North-East, South and West part of the Yabello town (zonal capital of the Borana administrative zone) in the Dawa sub-basin watershad (OWDSE, 2010).

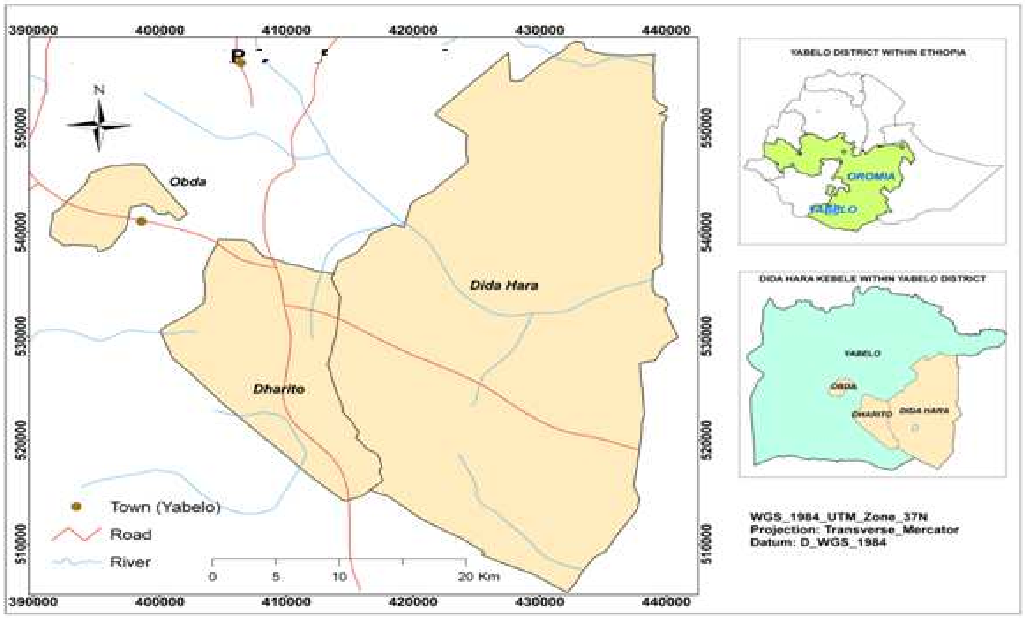

Figure1.1. Map of the study sites from Yabello district in Borana rangelands of southern Oromia

*1.6.1.2. Climate of the study area*

The Borana rangelands are characterized by arid to semi-arid climate with most areas receiving between 238 and 896 mm annual precipitation, with a high coefficient of variability that ranged between 18 and 69% (Angassa and Oba, 2007). The rainfall in Borana is bimodal, with 60% received during the long rains from March to May and 30%

during the short rains from September to November, while the remaining 10% is from the occasional rains.

The mean daily temperature in the Borana rangelands is 24°C, with a mean maximum and minimum daily temperatures of 28 and 17°C, respectively (Coppock, 1994). Drought is recurrent in the area and generally increased from once every twenty years in 1900s (Cossins and Upton, 1998) to five to six years in 2000s (Megersa et al., 2014).

#### *1.6.1.3. Soils of the study area*

The terrain of the central Borana plateau includes a central mountain range, scattered volcanic cones and craters and gently undulating and flat plains. The basement-complex mountain largely runs from north-west to south-east from Yabello to Moyale and north from Arero (OWWDE, 2010).

Soils in the Borana rangelands are developed from granitic and volcanic parent materials and their mixtures, and soils of East Africa mainly composed of the Precambrian basement complex, quaternary deposit and tertiary and quaternary volcanic parent materials followed one another (Coppock, 1994).

Soils are generally characterized by low organic matter (fertility) and deficiency in most nutrients, especially nitrogen and phosphorus, having a very low cation exchange capacity (CEC),and are prone to compaction attributed to the very old age of common parent materials (Tully et al., 2015).

The uplands and hills had well drained soils (Nitisols, Luvisols and Cambisols) having reddish brown to dark reddish brown color, while the bottom land soils (Vertisols, Fluvisols and Andosols) are characterized by dark-to-dark grayish color (Oba et al., 2000; OWWDE, 2010).

The level of electrical conductivity (EC) in soils of the Dawa sub-basin, where the present study area found, is less than 2 mmhos/cm which indicates that the soils are salt free and suitable for most crops production and/or for normal growth of bush and shrubs for grazing (OWWDSE,2010).

In general, soils in Borana rangelands are well drained and usually have equitable proportion of sand (53%), clay (30%) and silt (17%), which are sandy clay loam to sandy-loam in texture(Coppock, 1994).

#### *1.6.1.4. Vegetation in the study area*

In terms of woody vegetation, the Borana rangeland is dominated by tropical savannah with varying proportions of perennial grasses and woody plants with few areas covered by forest. Acacia species is the dominant vegetation cover in the study area.

The main encroachers of low value woody tree species are *Acacia mellifera, A. oerfata, A. drepanolobium, A. reficiens, Commiphora africana, A. brevispica, A. horrida, Aloe species, Albizia amare and h*ighly nutritive woody species are *Acacia tortilis* and *A. nilotica* (Coppock, 1994; Dalle et al., 2006).

Common perennial grass species of*Cenchrus ciliaris, Chloris myscrostachya, Cynodon dactylon, Panicum maximum, Pennisetum stramineum, Heteropogon contortus*, Sporobolus *pyramidalis* and *Themeda triandraChrysopogon aucheriPanicum turgidum* and *Panicum coloratum* and various families of forbs are also found (Angassa and Oba, 2010).

#### *1.6.1.5. Human and livestock population, and land use systems*

The Borana zone of southern Ethiopia has a human population of about 962,489, 487,024 men and 475,465 women, of which about 96,862 (40,502 female and 56360 male) are residing in Yabello district (CSA, 2008).The main economic activities are livestock production while crop farming is increasing in terms of the space it occupies (Coppock, 1994).

The major livestock species kept in the Borana lowland were cattle, goats; sheep and camel whilethe minor livestock species were equines (Desta and Coppock, 2002). As reported by Homan et al., (2008), the livestock density was estimated at 23.5 TLU km$^{-2}$ for Borana rangelands.

In a recent study (Samuel and Tredyte, 2017), livestock stocking density based on total area and livestock population was estimated at 24.2 TLU (Tropical Livestock Unit) per $km^2$ (16.5, 2.7, 1.8, 1.2 and 2 of cattle, goat, sheep, donkey, and camel, respectively). However, the number of cattle population in the Borana of southern Ethiopia followed the boom-and bust pattern (Desta and Coppock, 2002).

Generally, the open grazed areas are exposed to high grazing pressure throughout the year as the open-grazed areas were freely accessible to grazing throughout the year, whereas enclosures used during wet season. Changes in land use/land cover types such as expansion of crop farming and bush encroachment are common with consequences on the livelihood of the local communities

### 1.6.2. The study approaches

#### *1.6.2.1. Experimental sites*

The studies in this thesis were conducted in the areas by taking into account the existing grazing management systems (e.g., enclosures versus open-grazed lands), different land use systems (e.g., crop land , protected natural forest (woodland)) and exsiting prescribed fire experiments (burned sites versus unburned areas).

Accordingly, three sites namely Dida-Hara, Dharito and Obda Pastoral Associations (PAs) were selected through systematic random sampling technique. The Dida-Hara site was included in all studies**,** while the other two sites (Dharito and Obda) were used only for the study in **chapter III.**

The sites represented landscape types depending on vegetation structure and different land use systems to systematicaaly assess for carbon stocks dynamics. In selection rocess, a preliminary survey followed by mapping at landscape (sentinel level) was carried out were carried out. Knowledgeable community elders and experts working in the area were participated in the process.

#### *1.6.2.2. Experimental design*

In this study, we adopted a paired-site design approach which also applied in several research studies (e.g., Murphy et al., 2003; Novara et al., 2012)for studying soil organic carbon dynamics. Accordingly, adjacent sites, with similar slope, elevation, and land aspect were selected through systematic randmonsampling technque to fix the experimental plots.

A transect line of at least 500 m long was established in each adjacent site. Along a transect line, sampling plots (30x30m size each) with nested' sub-plots were establishedfor collection of response variables on soil and vegetation samples.

A systematic randomized samplingtechnique was employed to account for variability of soils and vegetation.However, the number of plots allocation along a transect line was flexible to adjust a small sample size in each study and control pseudo-replication.

The general approach and sampling methodology followed in data collection, preparation and analysis of vegetation and soil samples were presented the next sub-sections. Furhtermore, some of the specific design approach and sample collection methods specific to each study were presented in each material and method section of the individual study.

#### *1.6.2.3. Soil sample collection, process and laboratory analysis*

Random soil samples were collected at 0-5 5-15 and 15-30 cmsoil depths for the studies in **chapterII andV,** and from 0-15 and 15-30 cm depths for the study presented in **chapter III** using standard auger (Eijkelkamp Agrisearch Equipment, BV). Ten to fifteenauger points (sub-plots) were allocated in systematic randomized patterns placed the square angled wire (25 cm x25 cm) on a proper place within the plot.

The sub-plots were well spaced at minimum of 15 m distance from each other to account for heterogeneity in soil and vegetation characteristics within the site. However, the distance between the sub-plots was not drawn to the standard scale.During soil sampling, litter or any other residuematerials were removed from the soil surface by hand before taking the soil samples.

The collected soil samples from each auger points were mixed thoroughly in a bucket to form a composite sample for each depth and plot. Finally, thecomposite soil samples of about 1 kg for plot by soil layer were placed in a zip-locked nylon bag and labelled for further laboratory processes.

As number of plots and depths of sampling varied with the individual studies, specific sampling techniques were presented under the materials and methods section of the individual studies.

### Soil Bulk density determination

Soil bulk density was determined by recovering mass soil from the respective soil layers using auger (internal diameter of 7.6 cm) (Aynekulu et al., 2011). Accordingly, cumulative mass of soil was at three to four spots (sub-plots), in which the first sampling point was near the center of the main plot, and the other sub-plots (auger points) were locatedin systematic random pattern within the plot established for random soil sample collection.

A sampling metal plate was prepared for this purpose to help as guide for augering and to prevent the collapse of the auger near the soil surface. Accoridngly, the sampling plate was first pressed firmly onto the soil surface and place the auger at center of hole of the plate and started pushing the auger straight downwards by standing on either side of the plate.

Hence, the first depth layer was augered and all of the soil materials from inside and outside of the auger and any soil that fell onto the sampling plate were transferred into a bucket with the shovel without disturbing the shape of soil aggregates. The same procedure was applied in collecting the mass of soil remaining soil layers.

Finally, the entire mass soil augered from the sub-plot by depth layer was transferred into bag with zip-locked for further processing.The same auger was used throughout soil sampling to avoid any change in volume of soil. Depth restriction, if any, was also recorded.

In the laboratory, composite soil samples were air-dried by spreading each on paper in a ventilated soil sample preparation room and sub-samples (200-250 gm based on the massof soil augered in each depth layer) oven-dried at 105$^o$C for 48 hours to constant weight, and gravimetric water content on sub-sample basis and used determine the total dry weight soil mass.

The remaining soil mass was air-dried and passed through 2 mm sieve and the coarse fragments (> 2 mm), and weighed separately. Bulk density was finally calculated as the ratio of the mass of oven dried soil mass to its auger volume corrected for coarse fragment and used in the calculation of SOC and TN stocks.

Similarly, composite soil samples were air dried; ground to pass through 2 mm size sieve and samples were further sieved to 0.05 mm size for determination of SOC and total nitrogen. Laboratory analysis for composite soil samples was carried out at the Soil and Plant testing laboratory at College of Agriculture, Hawassa University following the procedures as outlined by Sahlemedhin and Taye (2000).

### Determination ofsoil particle size distribution

Soiol particle size distributions were analyized by hydrometer method (Bouyoucos, 1962). In this respect, 50 gm of air-dried soil sample was weighed into a one liter plastic bottle with stopper and 100 ml of dispersing agent (40 gm of sodium hexametaphosphate and 10 gm of sodium carbonate were dissolved in distilled water) was added.

Then, the solution was shaken on an end to end shaker for three hours, transferred into the stirring cup and stirred for 5 minutes. The dispersed soil suspension was transferred to a cylinder, and first and second hydrometer readings were taken at 40 seconds after the cylinder is set down and 2 hours after the first reading, respectively. The USDA particle size classification method was followed to assign textural classes.

### Soil pH

Soil pH was measured in a 1:2.5 mixture ofsoil to deionized water ratio. For this, 10 gm of <2 mm size dry soil sample was weighed into a 100 mL beaker and 25 mL distilled water was added, and the suspension was stirred vigorously for 20 minute (Sahlemedhin and Taye, 2000).

Then, the soil-water mixture was allowed to stand for about 30 minutes.A pH meter was calibrated with buffer solutions of pH 4.0 and 7.0. The electrode of the pH meter wasinserted partly into the settled suspension and the pH readings were recorded.

**Cation exchange capacity (CEC)**

For determination of cation exchange capacity (CEC), the soil sample was extractedusing the 1 M ammonium acetate (1M$NH_4OAc$ at pH 7.0) following the guideline of Chapman (1965). Hence, five gram of air-dried soil sample passed through 2 mm sieve was weighed into a 250 mL beakerfor determination of CEC. About 100 mL of ammonium acetate (1MNH4OAc pH 7) was added and stirred instantly with a string rod and allowed to stand overnight being covered with watch glasses.

The next day, the soaked samples were transferred onto filter funnels placed on 250 ml volumetric flasksfitted with filter paper and the mixture was washed on the filter with 25 ml ethanol 3 times. Some drops of the filter were taken into Nessler tubes and 3 drops of Nessler's reagent was added to avoid formation of precipitation.

The soil sample was saturated with sodium by washing with successive20 mL of sodium chloride (NaCl) 5 times and the filtrate was collected in 250 ml plastic bottles for determination of CEC by distillation. The leachate (NaCl percolate) in the plastic bottles was transferred to a 500 ml Kjeldahl flask and the plastic bottles were washed with 25 ml of distilled water, with small amount at a time.

Fifteen mL of 0.20 N $H_2SO_4$ was poured to distillate receiver of 250 mL Erlenmeyer flask and the distillation tip was dipped in the sulfuric acid solution. Ten ml of 1N NaOH solution was added to the Kjeldahl flask and connected immediately to distillation apparatus.

After about 75 mL of the NaCl percolate was distilled, the distillate receiver was removed from the distiller, condenser tip was rinsed and the distillate was titrated with 0.1 N $H_2SO_4$ using methyl red indicators until the color changed from purple to yellow and finally CEC was computed

### Available Phosphorus (Av.P)

Soil available phosphorus was extracted with Mehlich-3 multi-nutrient extraction method (Mehlich, 1984) at Horticoop Ethiopia (Horticultural) PLC Soil and Water Analysis Laboratory, Bishoftu, Ethiopia.

The Mehlich No. 3 extricating solution contained [0.2 N $CH_3COOH$, 0.25 N ammonium nitrate ($NH_4NO_3$), 0.015 N ammonium fluoride ($NH_4F$), 0.13 N HNO3, and 0.001N ethylenediamenetetraacetic acid (EDTA)] and measured byInductively Coupled Plasma Atomic Emission spectrophotometer (ICP-AES) at wavelength of 178.221 nm and the value was reported in mg $kg^{-1}$. A blank of Mehlich No.3 was prepared for each batch during the analysis.

### Soil organic carbon (SOC) and totan nitrogen (TN) determination

The organic carbon content of the soils was determined following method of Walkley and Black wet-oxidation (Walkley and Black, 19934); where as total nitrogen was analyzed by wet-oxidation procedure of the Kjeldahl method (Bremner and Mulvaney, 1982).

### Calculation of SOC and TN stocks

SOC stock (Mg $ha^{-1}$) in each depth was calculated as described in formula below (Aynekulu et al., 2015):

$$\mathrm{SOC} = \frac{\mathrm{OC}}{100} \times \mathrm{Bd}\,(\mathrm{i}) \times (1 - \%\,\mathrm{frag}) \times \mathrm{D} \times 100 \qquad (1.1)$$

Where, SOC-soil organic carbon stock (Mg $ha^{-1}$), OC is soil organic carbon content (%) in fine soil (fraction < 0.05 mm) determined in the laboratory (%), BD (i) =bulk density, frag (%) = weight of coarse fragment (>2 mm) determined in the laboratory divided*D*= depth soil layer (cm) and 100 used to convert the unit to Mg $ha^{-1}$.

$$\mathrm{BD(i)} = \frac{\mathrm{M}}{\mathrm{V}} \qquad (1.2)$$

Where M = the oven-dry soil mass, (g). Here, auger volume was estimated from its (radius 3.8 cm) and depth of the soil as $V\ (cm^3) = \pi \times (3.8)^2 \times$ depth layer (cm), where: $\Pi = 3.142$.

The TN stock was calculated in an analogous to SOC stock by substituting the soil organic carbon content (% OC) by total nitrogen content (%TN).The overall (0-30 cm) SOC and TN stocks were estimated by summing in each soil layer.

### *1.6.2.4. Vegetation survey and biomass estimation*

Vegetation sampling was conducted using 0.5 m x 0.5m, and a 10 x 10 m quadrat size was used for sampling of herbaceous and woody layers, respectively. Response variables included herbaceous biomass, grass basal cover, species richness, and density of woody population.

Herbaceous biomass was estimated by hand harvesting all the herbaceous materials rooted withinthe sub-plot (quadrat) to ground soil surface.The distructively harvested herbaceous samples were weighed immediately and retained for determination ofherbaceous dry biomass (oven dried at 65°C for 48 hr to toconstant weight). Samples from the sub-plots were pooled together and reported average dry biomass in gm $m^{-2}$.

Herbaceous species richness was determined by counting all herb and grass species rooted in each sub-plot, while herbaceous cover (grass and forbs) was estimated visually based on the area (soil part) covered by herbaceous base compared to bare ground in each quadrat (Oba et al., 2001).Woody plant species (mature, saplings and seedlings) within 100 $m^2$area were recorded used to estimate the density of woody vegetation.

Sampling procedure with respect to measurement of dendrometric variables to develop allometric equations for the estimating the above-ground woodybiomass was presented in c**hapter IV**.Generally, knowledgeable people and expert working in the areaparticipated during field data collection.

### 1.6.3. Statistical analysis.

Analysis of variance (ANOVA) in linear mixed models wasemployed to fit the esearch hypotheses and objectives of the studies in c**aptersII, III and V**, whereas descriptive statistics and regression models were used for the study in **hapter IV).**The details of the models and procedures were presented in each respective section of each study. The data analyses were performed using SAS version 9.3® (SAS2012).

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# CHAPTER TWO

# 2. EFFECTS OF ENCLOSURE MANAGEMENT ON CARBON SEQUESTRATION, SOIL PROPERTIES AND VEGETATION ATTRIBUTES IN BORANA, SOUTHERN ETHIOPIA

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[a]School of Plant and Horticultural Sciences, College of Agriculture, Hawassa University, P.O. Box 05, Hawassa, Ethiopia
[b]International Livestock Research Institute (ILRI), P.O. Box 30709, 00100 Nairobi, Kenya
[c] School of Veterinary Medicine, Hawassa University, P.O. Box 05, Hawassa, Ethiopia
[d] World Agroforestry Centre (ICRAF), P.O. Box 30677, 00100 Nairobi, Kenya
[e] Botswana University of Agriculture and Natural Resources, Department of Animal Science and Production, Private bag 0027, Gaborone, Botswana
* Corresponding author's address: ayana.angassa@gmail.com

## ABSTRACT

*The use of enclosure has globally gained popularity as an effective strategy to enhance soil carbon sequestration, but empirical evidence is lacking particularly in arid and semi-arid rangelands of southern Ethiopia. We investigated for differences in soil properties, more specically soil organic carbon (SOC) and total nitrogen (TN) contents and sctocks, and vegetation characteristics between enclosures and adjacent open-grazed areas, while accounting for the effects of the enclosure age chronosequenceand soil depths. Accordingly, enclosures along three age categories (< 20, 20–30 and> 30 years old) each paired with adjacent open grazed areas were selected through systematic random sampling technque. We collected soil samples at three soil depths (0–5, 5–15 and 15–30 cm), and vegetation attributes from 90 plots within 9 enclosures each paired with the adjacent open grazing sites. The results showed that SOC and TN contents and stocks increased in the enclosures as compared to the adjacent open-grazed rangelands, although the differences were not significant ($P > 0.05$) and varied along the age sequence and soil depths. We recorded an overall mean of SOC stocks of 39.6 ± 3.5 Mg ha$^{-1}$ in the younger (< 20 years old) , 40.8 ± 3.4 Mg ha$^{-1}$ in the medium (20–30 years old) and 51.0 ± 4.4 Mg ha$^{-1}$ in older (> 30 years old) age enclosures to the depth of 0-30 cm. Similarly, the SOC stock for the adjacent open-grazed areas ranged from 34.4 ± 2.5 to 47.9 ± 5.1 Mg ha$^{-1}$. In contrast, herbaceous biomass was significantly ($P < 0.05$) higher inenclosures than the adjacent open-grazed areas.The enclosuresage chronosequence further showed a general trend in increasing SOC and TN contents and stocks, and herbaceoius biomass with increasing duration of enclosures, but not linear.The study concluded that enclosuresgenerally improved the soil nutrientsand recovery of vgetation, indicates the effectiveness of enclosure management in the restoration of degraded rangelands and roles for enhancing the soil carbon sequestration potentials,withimplication for environmental management andclimate change mitigatation. However, we remain cautious with respect to the conclusiveness of these findings given the scarcity of information on soil data prior to enclosure establishment and other environmental factors, most importantly soil moisture and climatic factors may influence the carbon and nitrogen dynamics in such ecosystems other than management and needs for a comprehensive study.*

**Keywords:** Soil carbon stock, Total nitrogen stock, Rangelands, Herbaceous biomass, Borana

## 2.1. **INTRODUCTION**

Rangelands constitute the largest and most diverse land resources in the world (Reeder and Schuman, 2002), and hold great potential for carbon sequestration (Lal, 2004). According to Campbell et al. (2008), rangelands can store enormous amount of terrestrial carbon stocks both globally (36%) and in Africa (59%),

The carbon sequestration potentias in the rangelands can help to mitigate the impact of climate change (Neely et al., 2009).Moreover, sequestration of carbon in the soil system is essential for the improvement of soil quality, nutrient retention and water holding capacity to increase the net primary productivity for more carbon assimilation (Lal, 2015).

Despite the potential of rangelands for carbon sequestration, heavy grazing pressure contributed to the rapid losses of soil carbon (C) and nitrogen (N) (Reid et al., 2004). The degradation of rangeland resources is a major problem in arid and semi-arid ecosystems with significant impact on the environment and livelihoods of the pastoral communities (Oba et al., 2000; Angassa, 2014).

Previous studies (Tessema et al., 2011; Bikila et al., 2016) showed that increased grazing pressure resulted in the losses of soil carbon (C) and nitrogen (N). According to Piñeiro et al. (2010), the effect of grazing modifies the structure and function of ecosystems, affecting biomass and soil organic storage.Management practices such as community enclosures and rotational grazing can help to restore rangeland ecosystems (Nosetto et al., 2006).

Similarly, others (Howden et al., 1991; Glenn et al., 1993; Walker and Steffen, 1993) have also indicated that rehabilitation of degraded rangelands by means of enclosure and/or grazing exclusure is a key strategy in reducing the loss of carbon (C) from terrestrial ecosystem.

Moreover, several studies (Su et al., 2005; Mekuria et al., 2007; Pei et al., 2008; Steffens et al., 2008) have documented an increase in soil C accumulation in exclusions, but others (Reeder and Schuman, 2002; Nosetto et al., 2006; Shrestha and Stahl, 2008) have reportedlittle improvement in soil carbon under enclosure management as compared to the open grazing systems..

In consrast, other studies particularly in arid and semi-arid areas in Kenya and Ethiopia (e.g. Verdoodt et al., 2009; Mekuria et al., 2011; Mureithi et al., 2014) reported that soil propertiesand vegetation characteristics are improved following grazing exclusion.

In general, differences among various studies and regions are apparent because of variations in climate, soil properties and soil depth, landscape position, plant community composition and management practices (Derner and Schuman, 2007; McSherry and Ritchie, 2013).

Furthermore, the complexity of vegetation dynamics, land use practices and soil characteristics are additional inadequacies to fully understand the driving factors (Angassa et al., 2012). This suggests the need to further investigate the effect of grazing management on carbon dynamics.

In East Africa, community enclosures are extensively practiced by pastoralists to conserve standing pasture for dry season grazing (Angassa and Oba, 2010; Oba et al., 2001; Verdoodt et al., 2010).

Enclosures are defined as area of rangeland, which is enclosed by a fence with branches of thorny Acacia trees as well as traditional rules to protect vegetation from grazing and/orbrowsing with the exception of calves and sick animals (Coppock, 1994; Oba, 1998).

In some cultures, such enclosed areas may be called “exclosures” (Aerts et al., 2009). According to Angassa and Oba (2010), calves are allowed grazing inside enclosures for 3 to 4 months depending on the length of the dry season.

The use of the communal enclosures for over three to four decades in savanna rangelands of southern Ethiopia and the restored areas for their exclusive use may have positive implication to enhance the potential of carbon sequestration in the region.

However, this potential notion has rarely been investigatedso far. Previous studies that have so far conducted in the rangelands of Borana in southern Ethiopia focused only on the effects of grazing on vegetation ecology and range condition assessment.

On the other hand, few studies (Belay and Kebede, 2010; Yusuf et al., 2015; Bikila et al., 2016) reported the influence of land use and vegetation types on carbon stocks and these studies are also limited and sites specific. Furthermore, these studies did not show the effects of enclosures management along the enclosures age chronosequence on soil carbon stock.

Hence, investigating the effect of enclosures on soil carbon in Borana rangelands helps to understand the potential of rangelands for carbon sequestrationto inform policies for diversification of local livelihood options through carbon finance. The age of enclosures may also reflect the spatial separation of carbonsequestration in terms of time of restoration of vegetation states.

Therefore, the objectives of the study were: (i) to investigate the effects of management systems on soil properties, more specifically SOC, TN contents and stocks and key vegetation characteristics; (ii) to investigate how the age of enclosures influences soil properties, more specially SOC and TN contents and stocks as well as vegetation attributes

We hypothesized that (1) enclosure management would favor more accumulation of SOC and TN stocks, and herbaceous biomass than the adjacent open-grazed areas; (2) enclosures in the older age category could accumulate more soil nutrients and herbaceous biomass than the younger enclosures.

## 2.2. MATERIALS AND METHODS

### 2.2.1. Study area

This study was carried out across 9 semi-private (community) enclosures in Dida-Hara area(04°47.318′N latitude and 038°20.017′E longitude), which are located at about 30 km north-east of Yabello town in Borana rangelands, southern Ethiopia ( Fig. 1.1). Dida-Hara covers an area of about 985 $km^2$

The biophysical conditions described of the study area are given in detailsin the general materials and methods section under **chapter I**, but a few points specific to the Did- Hara are repeated here.

The topography of Dida-Hara area is flat to slightly undulating plain, with altitude range between 1260 and 1700 meter above sea level (m.a.s.l.).Before the 1970s, Dida-Hara was open perennial grassland, which was used for grazing during the rainy season. After the 1980s, range enclosures were established by the community following the development of perennial ponds that attracted semi-permanent settlements (*villag*e).

The pastoral population in Dida-Hara area is settled in semi-permanent settlementslocally known as *Ollas*. The creation of *ollas* had substantially affected the patterns of seasonal grazing to a year-round grazing (Angassa and Oba, 2010).The mean livestock holding of household was estimated at 12.63 cattle, 11.13 small ruminants (goat and sheep) and 2.38 camels (Solomon et al., 2007).

Most of the communal rangelands are degraded due to the high stocking rate, which was estimated at 23.5 Tropical Livestock Units $km^{-2}$(1 TLU ~ 250 kg) (Homann et al., 2008). However, overgrazing in enclosure is not a threat as enclosures are seasonally grazed by calves only during the dry season and rested during the growth season. Incontrast, the communal rangelands are under continuous year round grazing systems (Angassa and Oba, 2010).

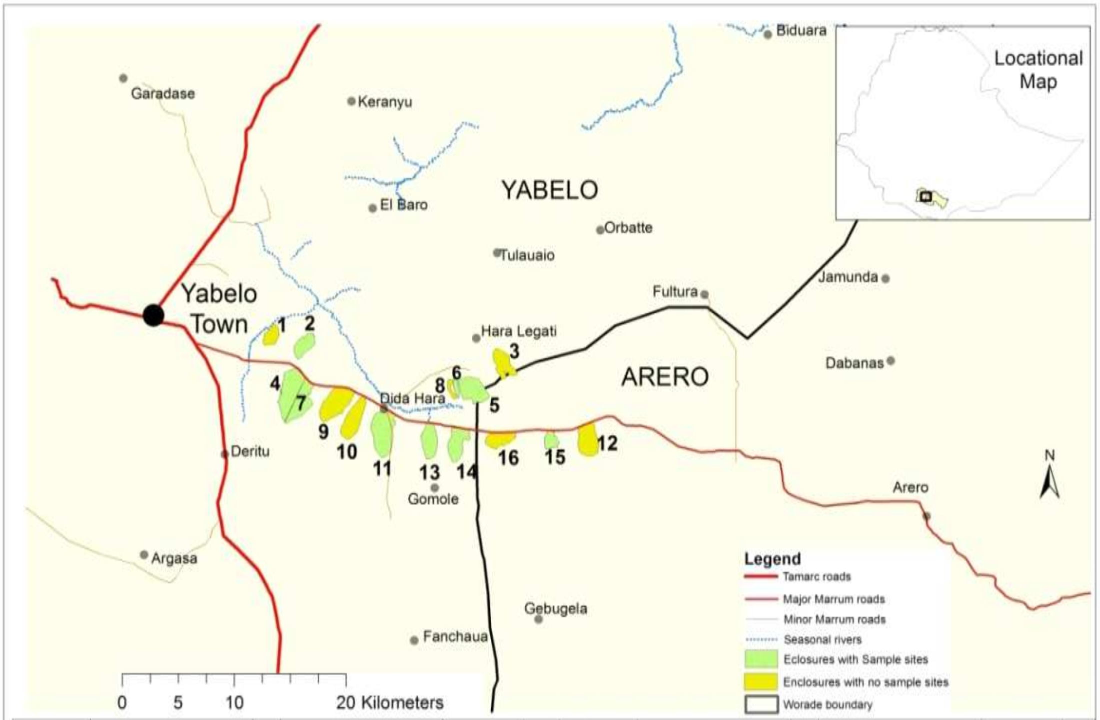
### 2.2.2. Study approaches and experimental design

Before starting the actual field sampling, field survey and mapping of enclosures were carried out in Did-Hara between May and June 2012, with technical support from the International Livestock Research Institute (ILRI, Kenya).

In the identification and mapping processes, knowledgeable community members, especially those managing the enclosures and experts working in the area were participated.

A high-resolution satellite image from Google earth® and ground-truth points collected by means of Geographic Position System (GPS) were used to delineate the boundary of enclosures following line features (fence).

Hence, a total of 16 communal enclosures were identified and mapped (Fig.2.1). Information collected during the preliminary field survey was used as a guide in selecting the study sites for the final data collection.



| Enclosure ID | Enclosure Name | Age | Block(Name of the cluster) | Sampled | Area | Enclosure ID | Enclosure Name | Age | Block(Name of the cluster) | Sampled | Area |
|---|---|---|---|---|---|---|---|---|---|---|---|
| 8 | Dhadacha | 12 | Dambala Saden | None | 0.6 | 6 | Dullacha Halake | 20 | Dambala Saden | Sampling site | 0.6 |
| 5 | Doyo Shune | 14 | Dambala Saden | Sampling site | 4.3 | 1 | Diimaa Liban | 28 | Dambala Bora | None | 1.8 |
| 2 | Duba Jatani and Gufu Jirma | 15 | Dambala Bora | Sampling site | 2.5 | 14 | Siku Muka Korma | 32 | Dambala Saden | Sampling site | 3.7 |
| 4 | Guyo Amaro | 17 | Dambala Bora | Sampling site | 5.7 | 16 | Hara Hawatu | 22 | Dambala Saden | None | 2.6 |
| 15 | Bule Harasogido(Alona) | 23 | Dambala Saden | Sampling site | 1.5 | 9 | Sanke | 33 | Dambi | None | 5.4 |
| 3 | Dullacha Liban | 20 | Dambala Saden | None | 3.1 | 11 | Jirmo and Ilman Sinphiree | 33 | Dambi | Sampling site | 6.2 |
| 7 | Doyo Dallacha | 20 | Dabmala Bora | Sampling site | 4.2 | 10 | Iliman Iyyaa | 34 | Dambi | None | 4.2 |
| 12 | Guyo Halake (Kukuba Sala) | 20 | Dambala Saden | None | 4.3 | 13 | Siku Hamsesa | 36 | Dambala Saden | Sampling site | 3.4 |

Area in Kilometers squared

Figure 2.1. Maps of enclosures and the sampling sites in Dida-Hara area from Yabello district in Borana rangelands, southern Ethiopia

Following the initial step, we selected 9 semi-private enclosuresassociated with settlements (*Ollas*) using a systematic random samplingmethod. The selected enclosures ranged from

60 to 620 ha in size, while the enclosures' age were distributed between 15 and 37 years old at the time of fielddata collection (May–June 2013).

We grouped the enclosures into threeage chronosequence: [< 20 years old (younger), 20–30 years old (medium) and> 30 years old (older)] each with three replicates (N = 3), while each age category of enclosures had corresponding adjacent open-grazed sites as control.

Therefore, age in this paper is used as anindependent variable, whereas site served as replicates.These selectedenclosures were distributed within 20 km radius in Dida-Hara area andall enclosures were selected from the uplands (> 1510 m.a.s.l.)

The experimental design paired site design approach consisted pairs of enclosuresand the adjacent open-grazed areas (as control) (Fig. 2.2a). At each paired site, a 500 m long line transect was established each in enclosure and the adjacent open-grazed site.

We checked whether the selected paired sites had similar soils and landscape position. The transect position was chosen at about 20–30 m away from the fence border between enclosure and the adjacent open grazed areas to avoid any edge effect.

At each paired site, five pairs of plots(30 m× 30 m each) were established along each transect. The plots were well spaced (separated by at least 100 m) to account for spatial variability and avoid pseudo-replication (Fig. 2.2a).

In total, ninety plots (90) were used for field data collection from nine (9) enclosures and the adjacent open-grazed (control) sites. Data collection on soil and vegetation was carried out at the end of the main rainy season between May and June in 2013.

### 2.2.3. Soil and vegetation sample collection and analysis

In each plot, we collected soil samples at three soil depths (0–5, 5–15 and 15–30 cm) from 10 sub-plots (auger points) and distributed in a randomized pattern within the plot using auger (Eijkelkamp Agrisearch Netherlands Equipment BV) (Fig. 2.2b).

A total of 270 composite soilsamples: [two grazing management systems [(enclosure vs. open-grazed areas) x three age categories of enclosures x three sites (replicates) x 5 plots (five each experimental site) x3 soil depths (0-5, 5-15 and 15-30 cm)].

In addition, soil mass was collected from three sampling points (sub-plots) in each plot for the same soil depths(Fig.2.2b).The augered mass soil samples were used to determine soil bulk density,while the samples from compositesamples were used to determine soil organic carbon (SOC) and related soil properties in laboratory.

Furthermore, three sub-plots of 0.5 × 0.5 m and a 10 x 10 m nested within the main plot at each paired site, respectively were used for sampling of herbaceous response variables (biomass estimation, species richness and basal cover) and woody density (mature, saplings and seedlings) as indicated in figure 2.2b.The details are presented under thegeneral materials and methods of section of **chapter I**

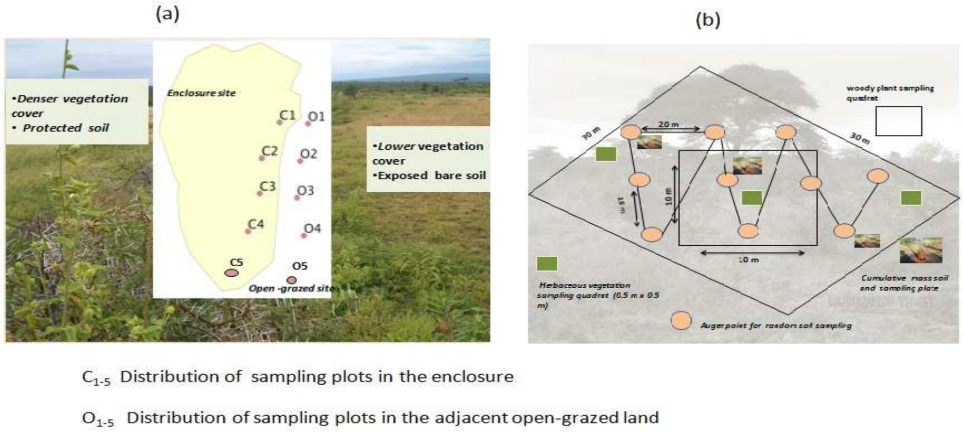


Figure2.2. The image of paired site with enclosure and the adjacent open-grazed areas (a) and diagrammatic sketch of sampling plot and sub-plot layouts (b).

2.2.4. Data analysis

Before data analysis, we tested our data for normality and homogeneity of tests. We also checked the data for outliers. The effects of management and age of enclosures across the soil depths on soil properties and nutrient stocks were analyzed using mixed model procedure.

The categorical fixed variables in the model for soil parameters included: [management (dummy variable: enclosure vs. open-grazed areas), three age categories of enclosures] at each soil depths (three levels) and to the the 0- 30 cm depth.

For vegetation attributes, the model included management along the enclosures age sequene. The sampling plot was included in the model as a random effect to control for the repeated measures. Mean comparisons were made using Tukey's test (adjusted) at $P< 0.05$. All values were reported as mean ± standard error (SE). All statistical analyses were performed using SAS version 9.3 software package (SAS, 2012).

## 2.3. RESULTS

### 2.3.1. Effects of grazing management on soil properties

The distribution of soil particle size fractions(sand, silt and clay)did not show significant ($P > 0.05$) difference between the two management systems (Table 2.1).However, the the proportion of silt and clay fractions were relatively higher in enclosure than in the adjacent open grazing lands, whereas that of sand content was inconsistent across the age sequence (Table 2.1).

With respect age effct, older enclosures had significantly ($P < 0.05$) higher in clay content than the younger enclosures, whereas higher silt content was recorded in the younger age group than the medium and older age of enclosures (Table 2.1). Overall, the proportion of sand fraction was dominant (i.e., over 43 %) in terms of textural fractions under both management systems (Table 2.1).

In the present study, soil bulk density did not show any significant ($P > 0.05$) differences betweent enclosures and the adjacent open-grazed areas across the three soil depths (Table 2.1). However, relatevly higher bulk densityunder the open-grazed areas that ranged from 1.10 to 1.47gm $cm^{-3}$, while the values were ranged between 0.90and 1.39 gm $cm^{-3}$in soils under enclosure management across the three soil depths (Table 2.1).

However, apparently more difference was observed between the medium age enclosure and its adjacent open-grazed areas in the lower depth (15-30 cm). There was an increasing trend in bulk density with increasing duration of enclosures (Table 2.1).

Our present results showed that there was generally higher soil pH in enclosures than that of the adjacent open-grazed rangelands, although results varied with the enclosures age sequence.Accordingly, the medium age enclosures at had significantly ($P < 0.05$) higher soil pHthan its adjacent open-grazed areas in the5–15 and 15–30 cm depths (Table 2.1).

Our results also showed relatively higher CEC values in enclosure than the adjacent open-grazed areas. However, only the medium age enclosures that had significantly ($P < 0.05$) higher CEC values than than the corresponding adjacent open-grazed areas across the three soil depths (Table 2.1).

Furthermore, we observed an increasing trend in CEC contents with increasing duration of enclosures, although significant difference ($P< 0.05$) was obtained only between the older (> 30 years old) and younger (< 20 years old) enclosures age  categories(Table 2.1).

Our present study results also showed that relatively higher available phosphorus (Av.P) inside enclosuresas as compared to the adjacent open-grazed areas. However, apparently more increase of Av.P values were recorded in the medium and older age enclosures in the top soil surface (0–5 cm) and medium depth (5-15 cm) , while lower in the lower depth (15–30 cm) except for the medium age enclosures (Table 2.1).

Table 2.1. Mean values of selected soil properties between enclosures along the age chronosequence and the adjacent open-grazed areas across three soil depths in Dida-Hara from Borana rangelands

| | | Grazing management systems | | | | | |
|---|---|---|---|---|---|---|---|
| Soil properties | Depth (cm) | Younger enclosures (<20 years old) | Adjacent open-grazed | Medium enclosures (20-30 years old) | Adjacent open-grazed | Older enclosures(>30 years old) | Adjacent open-grazed |
| Sand (%) | 0-5 | 44.1±1.4 | 44.7±1.3 | 47.3±1.4 | 49.3±2.2 | 46.9±1.7 | 45.5±2.4 |
| | 5-15 | 43.1±1.3 | 44.2±1.3 | 45.4±1.6 | 47.5±2.4 | 44.9±1.6 | 42.5±2.1 |
| | 15-30 | 41.8±1.4 | 44.6±1.8 | 42.6±1.7 | 46.2±2.2 | 42.9±2.0 | 43.4±2.3 |
| Silt (%) | 0-5 | **25.2±1.6** | 24.9±1.6 | 19.3±1.4 | 19.8±1.6$^{a}$ | 18.1±1.5 | 20.7±2.1 |
| | 5-15 | **25.6±1.5** | 24.3±1.4 | 19.9±1.6 | 20.9±1.9 | 20.9±1.7 | 21.7±2.1 |
| | 15-30 | **26.0±1.5** | 24.4±2.0 | 21.9±1.5 | 21.0±1.8 | 21.3±1.3 | 22.7±2.1 |
| Clay (%) | 0-5 | **30.8±1.1** | 30.4±1.1 | 33.5±1.0 | 30±.91.2 | **35.0±1.0** | 33.8±1.0 |
| | 5-15 | 31.3±1.0 | 31.5±1.2 | 34.7±0.9 | 31.6±1.2 | 34.2±0.9 | 35.9±0.7 |
| | 15-30 | **32.2±0.9** | 31.5±1.2 | **35.7±1.1** | 32.8±0.8 | **35.8±1.0** | 33.9±0.8 |
| BD (gmcm$^{-3}$) | 0-5 | 0.91±0.06 | 1.1±0.1 | 1.10±0.1 | 1.10±0.05 | 1.21±0.15 | 1.26±0.21 |
| | 5-15 | 1.10±0.10 | 1.19±0.1 | 1.16±0.1 | 1.21±0.07 | 1.13±0.08 | 1.11±0.04 |
| | 15-30 | 1.13±0.08 | 1.21±0.1 | 1.39±0.1 | 1.47±0.12 | 1.34±0.08 | 1.30±0.10 |
| pH ($H_2O$) | 0-5 | 6.2±0.1 | 5.9±0.1 | 6.3±0.2 | 6.2±0.2 | 6.2±0.1 | 5.9±0.1 |
| | 5-15 | 6.1±0.1 | 5.9±0.1 | 6.4±0.2$^{a}$ | 6.0±0.2$^{b}$ | 6.2±0.1 | 5.9±0.1 |
| | 15-30 | 5.9±0.1 | 5.9±0.1 | 6.6±0.2$^{a}$ | 6.2±0.2$^{b}$ | 6.1±0.1 | 5.8±0.1 |
| CEC (cmol$_c$ kg$^{-1}$) | 0-5 | **15.57±0.55** | 15.48±0.73 | 18.42±1.40$^{a}$ | 14.35±0.71$^{b}$ | **22.56±2.85** | 19.83±1.05 |
| | 5-15 | **17.25±0.59** | 16.25±0.96 | 18.58±0.83$^{a}$ | 14.27±1.2$^{b}$ | **21.65±0.72** | 21.63±0.55 |
| | 15-30 | **18.32±0.81** | 17.80±1.08 | 18.84±0.73$^{a}$ | 12.86±1.1$^{b}$ | **22.94±1.32** | 20.67±0.66 |
| Av.P (mg kg$^{-1}$) | 0-5 | 3.91±0.31 | 4.41±0.25 | 6.87±1.95 | 5.70±0.63 | 5.42±0.97 | 4.87±0.30 |
| | 5-15 | 3.36±0.24 | 3.13±0.17 | 4.91±1.24 | 3.13±0.17 | 3.93±1.12 | 3.21±0.28 |
| | 15-30 | 2.64±0.23 | 2.75±0.22 | 3.46±0.49 | 2.75±0.22 | 2.74±0.43 | 3.02±0.48 |

Note: BD -Bulk density, CEC- Cation Exchange Capacity, Av.P- Available Phosphorus, N=3 -replication of experimental sites (enclosures along the age chronosequence each paired with open-grazed areas).

Mean values (±SE) followed by different letters showed significant P >.0 05 between enclosure and the adjacent open-grazed areas , while in bold showed higher in the older age enclosures as compared to the youger enclosures , after Tukey's test (adjusted) at P < 0.05 significant level.

### 2.3.2. Effects of grazing management on SOC and TN contents and stocks

The present results showedthat enclosures did not significantly (P>0.05) increased SOC content and stock, but marginally improved as compared to the adjacent open-grazed areas along the enclosure age sequence and soil depths (Table 2.2).

On average, enclosures along the age chronosequence had 8.13 -11.4%, 9.0 -20.60 % and7.6 -15.0%,respectively higher SOC contents in the younger, medium and older age enclosures as compared to their corresponding adjacent open-grazed areas across the three soil depths (Table 2.2).

Hence, apparently more increase in SOC contents of 20.6% was obtained in the medium age enclosure at 5-15 cm soil depth followed by the olderage enclosures with 15.0% increase in lower (15-30 cm) depth as compared to their respective adjacent open-grazed areas, although the amount increase was not signfcant in both enclosures groups (Table 2.2).

Similarly, the difference between enclosure and its adjacent open-grazed in terms of SOC stock ranged from −0.5- 2.8, 1.2 -2.9, and 0.3- 1.4 Mg $ha^{-1}$, respectively in the younger, medium and olderage enclosures across the three soil dephs (Table 2.2).

Accordingly, the SOC stock in enclsoures under the younger age category had showedinconsistent results decrease (-3.85%) in the 5-15 cm and significantly higher (50% increase) at 0-5 cm depth as compared to thethe adjacent open-grazed areas (Table 2.2).

On the other hand, increase in SOC stocks with ranged from 10.0 -25.0% is more apparently in the medium age (20-30 years old) enclosures as compared to the adjacent open-grazed areas, although the increase didnot show significant across the three soil depths($P > 0.05$) (Table 2.2).

Furthermore, the older age enclosures had more SOC content and stock along the three soil depths, although the differences did not show significant ($P > 0.05$) (Table 2.2).Likewise, relatively higher TN content was recorded under enclosures as compared to the open-grazed areas across the three soil depths, butresults varied along the age chronosequence and soil depths (Table 2.2).

Accordingly, more apparent increase in TN contentsthat ranged from 11.8 to 21.4% wererecorded in the older age (> 30 years old) enclosures; with significantly higher (P

$<0.05$) value as compared to the adjacent open-grazed areas was recoredonly in 15-30 cm depth (Table 2.2).

Similarly, younger age enclsoures (< 20 years old) inconsistent results in terms of TN stock with both signficatnt higher (42.4% increase) in the 0-5 cm soil layer,and marginally lower (6.7%) in the 5-15 cm soil depth as compared to its corresponding open-grazed rangelands(Table 2.2).

On the other hand, we recorded more range of TN stock from 3.65 to 33.90% and14.60 to 253.9% increase in enclosures at the medium and older age categories, respectively than their respective open-grazed areas across the three soil depths (Table 2.2).

In the current study, both TN content and stock increased with increasing with duration of enclosures, but the increase was non-linear as only the older enclosures that had signfrcantly higher TN content than the medium and young enclosures, while the TN stock was only significant between the older and younger enclosure age groups, yet results varied along the soil depths (Table 2.2).

In the present study, we also found a higher C: N in enclosures management than in the open-grazed areas, but did not significantly ($P > 0.05$) vary (Table 2.1). Furthermore, results varied across the enclosure age categories each paired with adjacent open-grazed site (Table 2.1)

Table 2.2Mean values of soil organic carbon (SOC) and total nitrogen (TN) conents and stocks andCarbon to Nitrogen ratio (C: N) in enclosures along the age chronosequence and the adjacent open-grazed areas across three soil depths in Dida-Har from Borana rangelands

| Soil properties and nutrient stocks | Depth (cm) | Grazing management systems | | | | | |
|---|---|---|---|---|---|---|---|
| | | Younger enclosures (<20 years old) | Adjacent open-grazed | Medium enclosures (20-30 years old) | Adjacent open-grazed | Older enclosures(>30 years old) | Adjacent open-grazed |
| SOC (%) | 0-5 | 1.37±0.08 | 1.23±0.09 | 1.46±0.14 | 1.34±0.12 | **1.74±0.16** | 1.55±0.17 |
| | 5-15 | 1.33±0.10 | 1.23±0.07 | 1.23±0.11 | 1.02±0.09 | **1.55±0.14** | 1.44±0.14 |
| | 15-30 | 1.12±0.08 | 1.01±0.06$^{b}$ | 1.03±0.14 | 0.93±0.08 | **1.39±0.13** | 1.21±0.13 |
| SOC stock (Mg ha$^{-1}$) | 0-5 | 7.70±1.03$^{a}$ | 5.14±0.46$^{b}$ | 7.51±0.92 | 6.36±0.84 | **9.78±1.55** | 9.36±1.54 |
| | 5-15 | 12.75±1.14 | 13.26±1.66 | 14.10±1.10$^{a}$ | 11.28±1.15$^{b}$ | **16.92±1.61** | 15.71±1.69 |
| | 15-30 | 19.17±1.97 | 16.37±1.17 | 19.21±2.45 | 17.53±1.30 | **24.30±2.03** | 22.87±2.51 |
| TN (%) | 0-5 | 0.15±0.01 | 0.15±0.01 | 0.14±0.01 | 0.13±0.0 | **0.19±0.0** | 0.17±0.01 |
| | 5-15 | 0.14±0.01 | 0.14±0.01 | 0.13±0.01 | 0.14±0.01 | **0.18±0.0** | 0.15±0.01 |
| | 15-30 | 0.13±0.01 | 0.13±0.01 | 0.13±0.01 | 0.12±0.01 | **0.17±0.01$^{a}$** | 0.14±0.01$^{b}$ |
| TN stock (Mg ha$^{-1}$) | 0-5 | 0.84±0.11 | 0.59±0.06 | 0.79±0.10 $^{a}$ | 0.59±0.07$^{b}$ | **1.14±0.22** | 0.92±0.10 |
| | 5-15 | 1.38±0.11 | 1.48±0.16 | 1.58±0.14 | 1.47±0.16 | **1.95±0.15** | 1.64±0.10 |
| | 15-30 | 2.15±0.13 | 2.10±0.19 | 2.27±0.16 | 2.19±0.13 | **2.83±0.16** | 2.47±0.19 |
| C:N ratio | 0-5 | 9.9±1.1 | 9.1±0.8 | 10.3±0.9 | 10.6±0.70 | 9.4±0.6 | 9.4±0.9 |
| | 5-15 | 9.9±1.1 | 9.6±1.0.0 | 9.5±0.8 | 7.9±0.6 | 8.8±0.6 | 9.8±1.0 |
| | 15-30 | 9.2±1.1 | 9.0±1.2 | 9.0±1.1 | 8.5±0.8 | 8.9±0.9 | 9.7±1.4 |

N=3 - replication of experimental sites (enclosures along the age chronosequence each paired with open-grazed areas). Mean values (±SE) followed by different letters showed significant at $P >.0\ 05$ between enclosure and the adjacent open-grazed areas , while in bold showed significantly higher in the older age enclosures as compared to the youger enclosures , after Tukey's test (adjusted) at $P < 0.05$ significant level.

Averaged for all sites, soil organic carbonand total nitrogen contents and stocks between enclosures versus open-grazed areas along the soil depths are presented in figures 2.3 (a-d). Accordinly, our results showed that more apparent increase (12.4%) in SOC content in enclosures as compared to the open-grazed lands was obtained in 15-30 cm depth, although the difference was not significant (Fig. 2.3(a)).

Likewise, enclosure had 1.38, 1.17 and 1.98 Mg ha$^{-1}$, respectively more SOC stocks as compared at 0-5 , 5-15 and 15-30 cm depths than the adjacent open-grazed areas, respectively, but did not show any significant differences ($P <0.05$) (Fig.2.3 (b)).

These values are equivalent to percent increase of 9.0 to 20.0 % in SOC stock in the enclosures as compared to the adjacent open-grazed areas, with more apparent difference was obtained in the 0-5 cm depth, although the difference was not significant (Fig.2.3 (b)).When averaged for all sites, enclosures had accumulated 43.80±2.30 Mg ha$^{-1}$ SOC stock, whereas in the in the open-grazed areas was 39.20 ± 2.00 Mg ha$^{-1}$to the top of 30 cm depth

Similarly, TN content also showed percent increase of16.7% due to enclosure management as compared to the open-grazed areas in the lower depth (15-30 cm), although the difference were not significant (Fig. 2.3(c)), whereas accumulation of TN stock in enclosures (0.22 Mg ha$^{-1}$, i.e., 31.4% increase) as compared to that of the open-grazed areas was recorded in 0-5 cm depth(Fig.2.3 (d)).

Acoss all sites, enclosureaccumulated TN stock estimated at 5.00±0.20 Mg ha$^{-1}$ and that of the adjacent open-grazed areas with 4.5±0.2 Mg ha$^{-1}$ were recorded in this study.

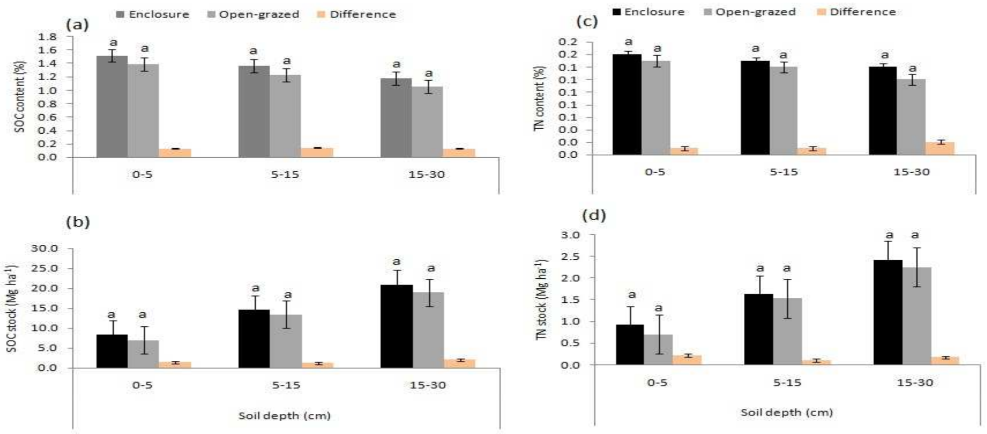


Figure2.3Mean values of soil organic carbon (SOC) content (a), SOC stock (b), total nitrogen (TN) content (c) and TN stock (d) in enclosures and the adjacent open-grazed areas across the three soil depths.Bars followed by same letters at each soil depth did not differ significantly (P >0.05), after Tukey's test (adjusted). Error bars indicate standard error of the mean (SE).

Overall, total mean SOC and TNcontents and stocks with pooled data to the 0-30 cm depth in enclosures along the sequence each paired with the open-grazed areas are presented in Figures 2.4(a-d).

Accordingly, enclosures had increased higherSOC content in the younger, medium and older age enclosuresas compared to their respective adjacent open-grazed areas with more apparent relative increase (14.3%) was obtained in enclosures at the mediumage category (20-30 years old), although the difference not significant (Fig. 2.4(a)).

On average, we recorded higher SOC stocks within enclosures (i.e., ranged from 39.55±3.45 to 50.96±4.37 Mg ha$^{-1}$) andthe values ranged from 34.75±2.32 to 47.94± 5.05 Mg ha$^{-1}$) for open grazed areas in the top 0-30 cm depth (Fig 2.4b).

The soil carbon sequestration rate in enclosures along the age sequence was estimated. Accordingly, difference between younger enclosures (average of 16 years) and medium enclosures (average of 22 years old), the the medium age enclosures had accumulated SOC at the rate of 0.2 Mg ha$^{-1}$yr$^{-1}$ as compared to younger enclosures.In the same way, enclosures in the older age category accumulated about 0.8 Mg ha$^{-1}$ yr$^{-1}$ attributable to age difference between 22 and 35 years (medium to older age categories of enclosures).

Overall, enclosures managed for more than 30 years in the study area had shoowd marginal increase of SOC stock at a rate of 0.6 Mg ha$^{-1}$yr$^{-1}$ than the younger age enclosures that ranged between 16 and 35 years (younger to older ages).

In this study, wealso found that the older enclosures (> 30 years old) had relative increase of 20% inTN content as compared to its adjacent open-grazed areas when data was pooled to 0-30cm depth, Furthermore, older enclosures had accumulated higher by 0.9 Mg ha$^{-1}$ (18% increas) in TN stock when compared with the adjacent open-grazed areas (Fig.2.4 (d)).

However, no recognizable differences were observed between the younger and medium age enclosures and their respective adjacent open-grazed areas in terms of TN contents and stocks as it was depicted in Figures 2.4(c&d).

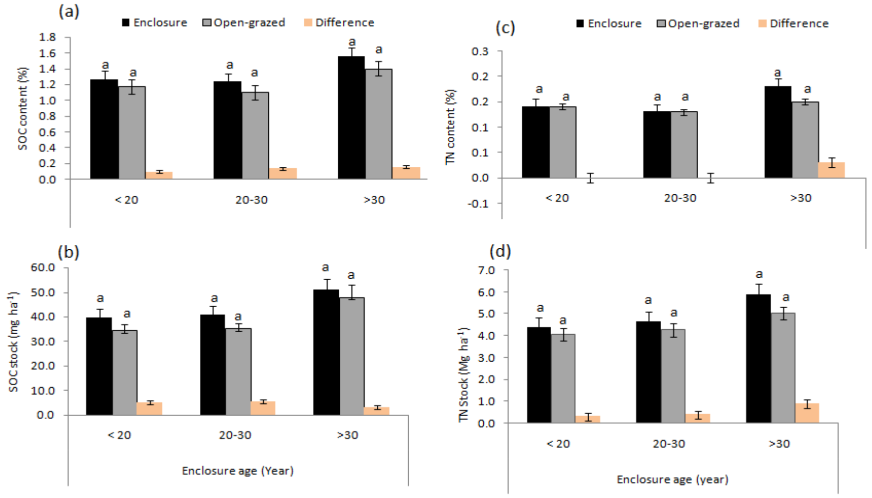


Figure 2.4 Mean values of soil organic carbon (SOC) content (a), SOC stock (b), total nitrogen (TN) content (c) and TN stock (d) in the enclosure and their djacent open-grazed lands along the sequence an in nthe top of 0- 30 cm depth.

Bars followed by similar letters at each age category of enclosure did not differ significantly ($P > 0.05$), after Tukey's test (adjusted). Error bars indicate standard error of the mean (SE).

### 2.3.3. Effects of grazing management on vegetation characteristics

The results in this study showed that enclosures had significantly ($P < 0.05$) higher herbaceous biomass, species richness, and herbaceous basal cover as compared to the adjacent open-grazed areas (Table 2.3), althoiugh results varied along the age chronosequence

Accordingly, herbaceous biomass increased by about 2.7 times in the younger, 0.9 times in the medium and 2 times in the older aged enclosures as compared to their corresponding adjacent open-grazed areas (Table 2.3).

Averaged for all sites enclosures had (difference ± SE: 71.8 ±4.8 gm m-2) herbaceous biomass as compared to the open-grazed areas when averaged for all sites (Table 2.3). In terms of age effect, the older enclosures had increased herbaceous biomas by only 30.0 gm m-2 (28.5% increases) as compared to the medium and younger enclosures (Table 2.3).

Our results also showed that enclosures had significantly higher (*P* < 0.05) species richness than the open-grazed areas. However, only the younger enclosures that had significantly (P < 0.05) higher number of species richness than its adjacent open-grazed areas (Table 2.3). The resultson herbaceoius richness with respect to age of enclosures was non-linear, which was initially, increased in the medium age and then declined in the enclosures with older age group (Table 2.3).

Our results also showed that significantly more herbaceous basal cover (difference ± SE: 20.9±3.4%) in enclosures as compared to that of the adjacent open-grazed areas. However, results showed variability along the age chronsequnce, where apparently more significant difference was obtained in the older enclosure (28.0% increase) as compared with that of the adjacent open-grazed areas (Table 2.3).

Overall, herbaceous basal cover increased with increasing enclosures age. However, more significant variation was observed between the older- and younger age enclosures (Table 2.2). In this study, we did not observe significant (*P*>0.05) difference between enclosures and the adjacent open-grazed areas in terms of woody density, and the results alsovaried along the enclosures' age chronosequence (Table 2.3).

Accordingly, enclosures had relatively higher woody density (1309 woody plants stems ha-1) than open-grazed areas (1238 number of woody plants stems ha-1 except in the younger enclosures, where slightly lower counts of woody plants were recorded (Table 2.3)

Table 2.3. Mean values of selected vegetation attributes between enclosures along the age chronosequence and adjacent open-grazed areas in Dida-Hara area of Borana in southern Ethiopia

| | | | Vegetation attributes | | | |
|---|---|---|---|---|---|---|
| Enclosures' age group | Management systems | N | Herbaceous biomass (g m$^{-2}$) | Herbaceous richness (number of species m$^{-2}$) | Basal cover (%) | Woody density (stems ha$^{-1}$) |
| < 20 years | Enclosure | 3 | 105.2 (11.4)$^{a}$ | 5.0 (0.2)$^{a}$ | 25.04(5.2)$^{a}$ | 1167 (174) |
| | Open-grazed | | 28.6 (6.4)$^{b}$ | 3.0 (0.5)$^{b}$ | 7.5 (2.23)$^{b}$ | 1573 (222) |
| 20-30 years | Enclosure | 3 | 105.4 (19.7)$^{a}$ | 5.0 (0.4) | 31.5 (5.0)$^{a}$ | 1307 (282) |
| | Open-grazed | | 56.1 (8.9)$^{b}$ | 4.0 (0.5) | 14.8 (3.3)$^{b}$ | 1067 (166) |
| > 30 years | Enclosure | 3 | 135.5 (24.7)$^{a}$ | 5.0 (0.17) | 45.0 (4.98)$^{a}$ | 1453 (393) |
| | Open-grazed | | 43.6 (4.81)$^{b}$ | 4.0 (0.5) | 16.7 (3.13)$^{b}$ | 1073 (90) |
| Overall | Enclosure | 9 | 115.4 (11.2)$^{a}$ | 5.0 (0.15)$^{a}$ | 33.9 (3.11)$^{a}$ | 1309 (169) |
| | Open-grazed | 9 | 43.6 (4.8)$^{b}$ | 3.0 (0.30)$^{b}$ | 13.0 (1.75)$^{b}$ | 1238 (102) |

Mean values (±SE) followed by different superscript letters are significantly different between enclosure and open-grazed lands, after Tukey's test (adjusted) at $P<0.05$. N- Replication of enclosures along the age chronosequence each paired with open-grazing systems.

## 2.4. DISCUSSIONS

### 2.4.1. Effects of grazing management on soil properties

Absence of significant difference between enclosures and open-grazed areas in terms of soil particle distribution, which is also in agreement with the findings of Angassa et al. (2012) that indicated that texture is a permanent soil property that would not be altered in short period and grazing management might have minimal effect in this regard.

However, the inconsistent results (i.e., decrease in silit and increase in clay) with respect different age of enclosures in the present study, perhaps due to weathering. Furthermore, the higher sand fraction, low silt and clay contents in the open-grazed areas that might have resulted from the process of accelerated soil erosion that facilitated by the poor ground cover (Oba et al., 2000).

Generally, the results of the current findings in terms of sand proportion sandy-clay loam textural class agrees with the report by Coppock (1994), who indicated that most parts of the Borana rangelands are characterized by sandy and shallow soils (53.0 % sand, 17.0 % silt and 30.0 % clay).

The average bulk density of the soil in the study sites did not show any significant difference between enclosures and open-grazed areas. However, a slightly higher soil bulk density in the open-grazed areas could be attributed to the effect of soil compaction as a result of continues grazing. Similarly, Mureithi et al. (2014) recorded a significantly lower soil bulk density in enclosures than in the open-grazing lands in Baringo district of Kenya.

Furthermore, the observed increase in soil bulk density with the advanced age of enclosures could be associated with the level of management practices by the individual settlements where the enclosures be affiliated with. A study conducted by Yusuf et al. (2015) in Borana rangelands of southern Ethiopia has also indicated variation in soil bulk density across different intensity of bush encroachment.

Our study results suggesting that although the bulk density was not found to be significantly different between the two management systesm, variation in bulk density did have an important influence on SOC stock.

Ingeneral, soil bulk density is a dynamic property highly dependent on soil conditions at the time of sampling. Hence, it may be necessary to calculate SOC stocks in a manner that is less subject to fluctuations to an equivalent sample mass method (fixed mass) (e.g., Lee et al.,2009).

A slightly higher pH in enclosures than in the adjacent open-grazed areas might be attributed to the effects of CEC and SOC within enclosures that in turn trap base cations. In a semi-arid savanna of Ethiopia, Tessema et al. (2011) reported that grazing pressure did not significantly affect soil pH, although to some extent higher pH value was found in the light grazed areas.

The discrepancy in the results of soil pH along different age of enclosure management is most likely related to past management practices (range burning and cultivation).

In this study, the overall mean pH values in enclosures (5.9 to 6.6) and open-grazed lands (5.8 to 6.2) across the three soil depths fall within the moderately acidic soils according to the ratings of Landon (1991), which could be attributable to occurance of heavy rainfall duringmain rainfall season preceded our sampling time leach away base forming cations from the surface soils, despite the arid and semi-arid of climatic condtions and limited rainfall in the Borana rangelands.

Inconstrst to our findings Angassa et al. (2012) reported that soils of the Borana rangelands are moderately alkaline with higher values of pH for grazed (7.56) and ungrazed (7.38) areas, which still are good enough for the growth of diverse plant species.

The results of the present study indicated that the CEC values are higher in enclosures than in the open-grazed areas, followed similar trends of pH and SOC. Organic matter usually increases the available negative charges in the soil, hence increasing the CEC.

A study conducted in semi-arid rangelands of Kenya (Mureithi et al., 2014) indicated that enclosures can accumulate more cations than open-grazed areas. The same authors also indicated that soils with high organic matter content might have higher cation exchange capacity due to more accumulation of litter from excessive above-ground biomass in enclosures.

Generally, the results of CEC values in the current study fall within the range of 15-25 cmolc kg-1 (i.e., medium) according to the ratingclassification by Landon (1991). Angassa et al. (2012) also found similar ranges of CEC values in Borana rangelands of southern Ethiopia.

Although not significant, relatevly higher amount of available phosphorus was observed in the medium age enclosure than the open grazing system, whereas this was reverse at the young age enclosure, similar trend with that of CEC and pH. In general, phosphorous is made available to the plants if the pH is in between 6.0 to 7.0, where our results in soil pH values fall within.

Angassa et al. (2012) also reported significantly higher available phosphorus in the ungrazed sites in the same study which is some what similar trend with our findigs in the present study. Furthermore, Mekuria and Aynekulu (2011) also reported a significant increase in terms of available phosphorus in enclosures in semi-arid areas of Tigray in northern Ethiopia.

In geral, our results on available phosphorus (Av.P) under both management systems corresponded to a very low range (0-15 ppm) based ratings by EthioSIS (2014).The probable explanation for the very low Av.P values might be due tothe low phosphorus status inthe parent material of the soils in the study areas.

As indicated by Quinton et al. (2001) phosphorous is normally lost through erosion and decrease in grass cover due to animal grazing and trampling, but animal excreta has long been recognized as an important pathway in the phosphorus cycle in grazed pasture.

### 2.4.2. Effects of grazing management on SOC and TN contents and stocks

Although the differences between enclosures and the adjacent open-grazed rangelands in terms of SOC and TN contents and stocks did not show significant to meet our expected hypothesis, relatively more accumulation of SOC and TN contents and stocks in the enclosures than adjacent open-grazed lands was observed.

The implicitly assumption of this study was that enclosures and the adjacent open rangelands have had comparable initial conditions such that changes in soil quality, more specifically SOC and TN stocks dynamics are the consequence of enclosure establishment and the respective management differences .

The current increase in SOC content under enclosure as compared to the adjacent open-grazed areas might be attribute to more herbaceous biomass and litter fall as well as the dense fibrous rooting systems of woody trees favors soil organic matter formation and more built up carbon storage in the soil.

In line with our findings in the present study, Angassa et al. (2012) also reported that enclosures had moderately more SOC content (0.89 %) than the open-grazed areas (0.86 %) in the same study area. In contrast to our present findings, Rathjen (2012) found a significantly higher below-ground carbon allocation in the open-grazed areas than in enclosures in the same study area.

Others (Reeder and Schuman, 2002; Dabasso et al., 2014) also reported that exclusion of livestock enhanced the growth and development of annual and perennial grasses which have dense fibrous rooting systems conducive to soil organic matter formation and accumulation.

Reeder et al. (2004) also argued that plant root residues are the primary source of soil organic matter that helps to improve more accumulation of below-ground biomass that in succession enhances soil organic carbon.

Our results showed an increase (5.24, 5.66 and 3.06 Mg $ha^{-1}$) in SOC stocks, respectively under the younger (<20 years old), medium (20-30 years old) and older (>30 years old) enclosure age sequence as compard to their paired adjacent open-grazed sites.

Similar trend of increases by 6.6, 9.6, and 10.6 Mg $ha^{-1}$ in SOC stocks, respectively in the 0-15 cm depth, after 15, 18, and 23 years of enclosure establishment was reported by Verdoodt et al.( 2009) in semi-arid rangelands lands of Kenya .

Although not significant SOC content and stocks were apparently more under enclosures at the medium age (20-30 years old), whereas the older age (>30 years old) had higher TN content and stock than their respective adjacent open-grazed areas.

However, our results in terms of SOC and TN stocks are in contrast with previous study on exclosures in the central and northern highlands of Ethiopia (Girmay et al., 2009), who reported an increase in SOC stock of 22.6 Mg C $ha^{-1}$ in the 0-15 cm depth after 8 years of exclosure establishment (i.e., on average, 2.8 Mg $ha^{-1}$ $yr^{-1}$).

More studies (e.g., Su et al., 2005; Yong-Zhong et al., 2005; Pei et al., 2008) also reported a significant increase in SOC and TN following grazing exclusion in a desert steppe of the in degraded lands of Inner Mongolia, northern China.

Our results on the differences between enclosures and open-grazed areas in terms of SOC and TN stockswere apparently more in the top surface (0-5 cm), while SOC and TN contents in the lower (15-30 cm) depth, although some discprepnacies were observed along the age chronsequence.

In linewith, Jobbagy and Jackson (2000) stated that the vertical distribution of SOC in different ecosystems might be synchronized by the type of vegetation through the root-shoot allocations along the soil profile. Lal (2004) also explained that in savanna ecosystems biological activity (especially termites) are important factor influencing soil structure and soil C is equally distributed within the soil profile.

On the other hand, Jobbagy and Jackson (2000) argued that loses of SOC is more sensitive in the upper most of the soils due to the impact of land management systems. Furthermore, Fultz et al. (2013) also highlighted that the depth interval as small as 0-5 cm has a very important implication in terms of capturing the very minimal changes in SOC and TN stocks due to the impacts of management and land use changes.

Other study (Shrestha and Stahl, 2008) also indicated a highly significant change in terms of SOC accumulation at 0-5 cm depth of the soil after 40 years of grazing exclusion in a semi-arid sagebrush steppe of Wyoming.

The SOC and TN contents and stocks increased with increase in the duration of enclosures, although it did not show linear to meet our second hypothesis which stated significant increase in SOC and TN stocks with advanced in the age of enclosure.

The differences in SOCsequestration indicates an increasein SOC stock which averaged each 0.3 Mg ha$^{-1}$ yr$^{-1}$ in the younger (average of 16 years old) and medium age (average of 22 years old) enclosures, whereas for older age enclosures (average of 35 years old) it was estimated at 0.1 Mg ha$^{-1}$ yr$^{-1}$.

The lack of linear increase along the age sequnece was possibly an indicator of vegetation succession and of the partial removal of organic matter input through harvesting of grass and forb biomass. However, the different ages of enclosures management could play significant role in regulating ecosystem serviceswithin rangelandsthrough enhancement of vegetation restoration, soil erosion control and carbon storage (Mekuria, 2013).

The overall lack of significant net effect of enclosure management on carbon sequestration can be understood as a balance from initial input at early age of enclosure (degradation level) , which is difficult to separate the effects of enclosure as management and the age sequnee, most importantly in the absence of soil data prior to enclosures establishement.

Dean et al. (2012) also indicated that the potential of rangelands for soil carbon sequestration may be hindered by uncertainty on the direction and magnitude of changes in soil organic carbon associated with shifts in woody vegetation cover.Moreover, Conant et al. (2001) aslo showed that the response of SOC to grazing is highly variable across various ecosystems.

Our results indicated that the overall mean of SOC stock, across the studied ages of enclosures (15-37 years old) ranged between 39.6 to 51.0 Mg ha$^{-1}$ within the 0-30 cm depth falls within the range of 20 and 80 Mg ha$^{-1}$reported for tropical woodland and savanna ecosystems by Alam et al. (2013). This finding also agrees with the report by Yusuf et al. (2015) in the same study area.

The overall, the low accumulation of SOC and TN stocks in the present study might be attributed loose enclosure management and human interference like grass cutting activities and lower inputs of primary productivity due to recurrent drought and low

### 2.4.3. Effects of grazing management on vegetation characteristics

The observed higher accumulation of herbaceous biomass in enclosures thanthe adjacent open-grazed areas in the current study agrees with the findings of Angassa and Oba (2010),which indicated enclosures accumulated significantly more above-ground herbaceous biomass than open-grazed in southern Ethiopia.The decline in herbaceous biomass in the open-grazed areas might be due to year-round continuous grazing (Wairore et al., 2015).

It seems that the accumulations of herbaceous biomass in enclosures (115.4 gm $m^{-2}$) and open-grazed plots (44.0 gm $m^{-2}$) in the current study are very low as compared to the finding of Oba et al. (2001), who reported high accumulation of herbaceous biomass (440 gm $m^{-2}$) for enclosure and continuously grazed areas (172 gm $m^{-2}$) in northern Kenya.

This suggests that the variations in sites and climatic conditions most likely contributed to the lowaccumulation of biomass. Low primary productivity and poor condition of semi-arid ecosystems were also noted elsewhere in the world (Milchunas and Lauenroth, 1993; Abebe et al., 2006).

The results of our findings indicated that the age of enclosures had no significant effect on herbaceous biomass. Similar observation was also reported byAngassa and Oba (2010) inBorana rangelands of southern Ethiopia.

Our findings in terms of herbaceous species richness were comparable with the findings of previous studies (Aerts et al. 2006; Angassa and Oba 2010; Verdoodt et al. 2010), who recorded higher herbaceous species within enclosures most probably because of reduced disturbance in enclosures.

The same authors have also pointed out that an increase in species richness to some extent, suggesting that short-term grazing exclusion could promote herbaceous species richness, while long-term grazing exclusion may not contribute to species diversity.

Overall, our results indicate that the cover of herbaceous plants in the open-grazed areas was very low,indicating that the impacts of heavy grazing pressure on rangeland vegetation and conversely the role of enclosures in restoration of rangeland vegetation. A similar study by Zhao et al. (2011) in the Inner Mongolia in northern China also showed that heavy grazing is followed by severe damage on vegetation cover.

The current study also indicates that woody plant density was relatively lower in the open-grazed areas than in the enclosures. This may support the notion that heavy grazing is not the only factor in driving bush encroachment in arid and semi-arid grazing areas (Angassa and Oba, 2008).

On the other hand, others (Mekuria andYami, 2013; Wairore et al., 2015) have reported the reduction in woody plant density, which could be linked to heavy grazing pressure in the communal areas.

## 2.5. CONCLUSIONS

Our results indicated that enclosures had relatively higher SOC and TN contents and stocks as compared to the adjacent open-grazed lands, although the results varied along the age sequence and soil depths. Hence, the medium age (20-30 years old) enclosures more apparently higher SOC content and stock, while the older age enclosures (>30 years old) showed a higher in TN content and stock as compared to their adjacent open-grazed rangelands.

Furthermore, soil carbon and TN contents and stocks increases with duration of enclosures though the increase was not linear. On the other hand, enclosure management promoted the recovery of vegetation as manifested by significantly higher herbaceous biomasses as compared to the open grazing system.

Thus, the data generated in the current study can inform policy makers and local land managers about the roles played by enclosures in terms of forage preservation, potential for carbon sequestration, livelihood diversification and ecosystems services.

The study suggested that there needs for a comprehensive study, most importantly soil moisture and climatic factors may influence the carbon and nitrogen dynamics in such ecosystems other than grazing management

## 2.6. REFERENCES

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# CHAPTER THREE

# 3. ASSESSMENT OF LAND USE CHANGE AND ITS EFFECT ON SOIL ORGANIC CARBON AND TOTAL NITROGEN STOCKS IN SAVANNA RANGELANDS, SOUTHERN ETHIOPIA

Submitted to Geoderma (ELSIVER): Kenea Feyisa[a], Sheleme Beyene[a], Ayana Angassa[b], and Ashenafi Burka[c] Ermias Aynekulu[d]. Assessment of Land Use Change and Its Effect on Soil Organic Carbon and Total Nitrogen stocks in savanna rangelands, southern Ethiopia

[a]Hawassa University, School of Plant and Horticultural Sciences, Ethiopia

[b]Botswana University of Agriculture and Natural Resources, Department of Animal Science and Production, Botswana

[c]Wondo Genet College of Forestry and Natural Resource, Shashemene , Ethiopia

[d]World Agroforestry Centre (ICRAF), United Nations Avenue, P.O. Box 30677-00100, Nairobi, Kenya.

## Abstract

*The influence of land use/land cover changes on the dynamics of carbon and nitrogen is a major concern in the framework of global policy agenda to address the impact of climate change and food security. We studied land use/land cover changes and its impacts on soil organic carbon (SOC) and total nitrogen (TN) contents and stocks in different land use types in the Borana rangelands of southern Ethiopia. Sequential satellite imageries of 1976, 1986 and 2013 were used to detect land use/land cover changes. The impacts of land use changes on the property of soils were also studied using four land use systems namely: woodland, communal grazingland, enclosure and cultivated land. Soil samples were collected at two soil depths (0–15 and 15–30 cm) in four land use types representing four line transects with three replication across different sites. The results generally showed the dynamics of land use changes in the study sites over the last 37 years (1976-2013). The results revealed a considerable reduction in woodland cover (-56.2%), communal grazingland (-38.8%) and enclosure (-27.3%) from the bench mark in 1976. In contrast, cultivated land was increased by 946.5% (more than 10 fold), while bushland cover was increased by 260.3% (7 times) in 2013 as compared to the situation in 1976. Overall, about 70% of the land use/land cover types in the study sites were vulnerable to chang, wheras the remaining balance was persistent to change over the last 37 years. This suggests an increase in ecological changes in the study area over the last decades. The results showed that changes in land use systems significant influenced ($P < 0.05$) SOC and TN contents and stocks. Accordingly, woodland had the highest SOC stock (55.94±3.41 Mg ha$^{-1}$) and cultivated land contained the lowest (38.10±2 .39 Mg ha$^{-1}$) in the 0-30 cm depth. We recorded a loss of about 319 Gg SOC and 30.65 Gg TN stocks, which mainly due to conversion of communal grazingland and woodland to cultivated land. Incontrast,the conversion of the communal grazingland and cropland to woodland and enclosures partially counter balance SOC and TN losses, although the results might be better understood as potential changes. We suggest further research that can address variability in carbon flux over time as influenced by land use changes to fully understand the long-term dynamics of carbon.*

**Key words:** Land use change, Mapping, Soil properties, Southern Ethiopia

## 3.1. INTRODUCTION

Land use/land cover changes are induced by natural processes and anthropogenic factors as a result of human beings interactions with their natural environment for survival (Lambin et al., 2001). The Borana rangelands of southern Ethiopia have been extensively used as communal grazingland by pastoralists for millennia, traditionally managed with the use of periodic fire until the early 1970s (Coppock, 1994).

However, there has been a dramatic shift from prime grazingland to cultivation, left unused because of bush encroachment, and highly degraded leading to different land use/land cover types in the region (Angassa and Oba, 2008).

Land use/land cover change is among the most important determinants of SOC stock status, other factors being similar, as they govern the long-term patterns of vegetation, frequency of removal and amount of organic matter (OM) added to the soil systems, and thus threatening ecosystem functions like soil fertility and carbon sequestration (Novara et al., 2012; Stringer et al., 2012).

Rangelands stored about 10–30% of the global soil carbon, besides the above-ground carbon storage in trees, bushes, shrubs and grasses (Grace, et al., 2006; Derner and Schuman, 2007). Hence, there is great potential for carbon sequestration in above-and below -ground pool that might play a key role in the mitigation of climate change effects (IPCC, 2007; Tennigkeit and Wilkes, 2008; Neely et al., 2009).

Rangelands comprise more than 60% of East Africa savanna ecosystems and the potential of these rangelands to sequester carbon under different management practices has been reported in numerous studies (Batjes 2004; Vågen et al., 2005; Grace et al., 2006). However, recent studies reported significant land use change in the rangelands of Africa as a result of increased food scarcity and demand for more lands particularly through expansion of cultivation to cope with economic hardships (Hoffman and Vogel 2008; Reid et al., 2014).

Estimating shifts of carbon due to land use change is a key process in determining the impacts of disturbances on carbon storage in ecosystems. For example, conversion of rangelands to cropland resulted in a loss of 95% of the above-ground and up to 60% of the below-ground carbon (Guo and Gifford, 2002; Reid et al., 2004). Woomer et al. (2004) also reported that land degradation can lead to a loss of up to 6 and 13 Mg ha$^{-1}$, vegetation biomass and soil carbon stocks, respectively.

Hence, land use changes can cause a change in land cover and an associated change in carbon stocks through alteration of the structure and composition of vegetation (Bolin and Sukumar, 2000). Lack of appropriate policies that govern natural resources management contributed to land use conversion from the African continent (Homewood et al., 2004).

However, the impact is hard to assess due to lack of information. For example, a review and synthesis of carbon balances for Africa from recent researches (Ciais et al., 2011) elucidated that inadequate *insitu* data and a high uncertainty have made it difficult to characterize the carbon stocks or fluxes in the region.

Moreover, the complexity of vegetation dynamics, land use practices and soil characteristics are additional inadequacies to fully understand the effect of land use systems on soil nutrient dynamics (Grace et al., 2006; Angassa et al., 2012).Generally, land use/land cover change is a dynamic process and its effects on soil properties and nutrient dynamics are varied due to factors such as soil type, climate, topography, vegetation characteristics and land management practices (Bolliger et al., 2008;Cantarello et al., 2011).

In contrast, restoring the degraded land to the natural vegetation can improve soil productivity and carbon sequestration in soils (Girmay et al., 2008; Lipper et al., 2010; Demessie et al., 2015). However, land resource and environmental decision makers require quantitative information on the spatial distribution of land use types and their conditions as well as temporal changes on carbon stock dynamics.

One way to reconstruct a better understanding of the past and present changes in land use and its impact on carbon storage potentials is through remote sensing data combined with ground based field surveys (Evrendilek, et al., 2011; Vågen et al., 2013).The savanna rangelands in southern Ethiopia comprise extensive rangelands with a unique feature of landscapes and supports pastoralism, small scale farming and other ecosystem functions (Coppock, 1994; Homann et al., 2008).

However, since the last four decades these rangelands have been experiencing considerable changes in patterns of land uses (Oba et al., 2000).As indicated by Coppock (1994), landuse change in the Borana plateau had begun around the 1960s and dramatic increase in bush cover, introduction of range enclosures, expansion of crop farming from the 1970s onwards (Dalle et al., 2006' Angassa and Oba, 2008).The potential grazingland is being cultivated, closed-off for purposes of private use during the last 30 years (Tolera and Abebe, 2007).

Some recent studies (Abate and Angassa, 2016; Ellias et al. 2015) showed that the Borana rangelands had undergone substantial changes during the past three to four decades, attributed to the shift in the environment or production system. Earlier study by Angassa et al. (2012) also indicated that land use and management systems can significantly affect the status of numerous soil properties in the same study area.

Furthermore, a few recent studies (e.g. Yusuf et al., 2015; Aynekulu et al., 2017; Feyisa et al., 2017) also showed that improved land use management practices such as enclosure improved carbon sequestration potentials and associated soil properties and vegetation attributes as compared to open-grazedlands in same study area.

Other (e.g. Bikila et al., 2016) also reported that prescribed burned areas have contributed to enhance carbon sequestration potentials and associated soil properties and vegetation attributes in Borana rangelands of southern Ethiopia

However, these studies are limited to few sites and specific land use types to represent the ast and heterogonous landscapes of the Borana plateau. On the other hand, there is no reason to assume that the current land use change will stop with the current government policies and increasing population pressure, rather it is likely to continue.

Therefore, the objectives of this study were to: (1)assess land use changes using the Landsat data for the period between 1976 and2013;(2) investigate the effects of different land use systems on SOC and TN contents and stocks;(3) estimate changes in SOC and TN stocks due to land use changes.

We hypothesized that (i) the magnitude and rates of changes in land use/land cover in the study sites is very high with further increase in recent years; (ii) SOC and TN contents and stocks significantly varied in different land use types, and (iii) the dynamics in land use brings considerable changes in SOC and TN stocks over the last decades.

## 3. 2. MATERIALS AND METHODS

### 3.2.1. Study area

Our study sites from Yabello districtof Borana rangelandsin southern Ethiopia were located between 4.57° and 5.06ºN latitudes, and 38.16° and 38.47º E longitudes with an area of about 125873 ha and covering three pastoral associations (PAs), namely, Dida-Hara, Dharito and Obda (see the maps in Fig. 1.1).The study sites were found in altitudes ranging from 1478 to 1928 meter above sea level (m.a.s.l).

In this study, pastoral associations (PAs) locally called *Kebele* refers to the smallest unit of local government within a district. These sites were selected through systematic random sampling technique to represent different land use types, among others, enclosures and open-grazed rangelands, crop farming and a few pockets of protected natural forest (woodland).

Expansion of bush encroachment was also among the most noticeable changes occurred in the study sites.In the identification processes; knowledgeable community members and experts working in the area were participated.The details of biophysical characteristics including climate, soil, vegetation and livestockproduction in the study area were described under the general materials and methods section **(chapter I).**

### 3.2.2. Research methodology

### *3.2.2.1. Data source, satellite image acquisition and pre-processing*

The Landsat multispectral scanner (hereafter MSS), thematic mapper (TM) and Enhanced Thematic Mapper Plus (ETM+) images of the years 1976, 1986 and 2013, respectively were used in the study.

The Landsat MSS (acquired on 25 January 1976), TM (acquired on 05 January 1986) and ETM+ (acquired on 30 March 2013) downloaded from the United States Geological Survey (USGS) with sensor characteristics of path/raw (168/056-57).The MSS had a spatial resolution of 79 m, while the TM and ETM+ both were of 30 m ground resolution.

All satellite images were configured to World Geodetic System 1984 (WGS 84) and Universal Traverse Mercator (UTM) zone 37N) specific to southern Ethiopia and georefrenced to known ground control points (GCP). Satellite scenes were acquired during the same season in order to minimize the influence of seasonal variations on LULC analysis.

Ancillary data such as shape file of Yabello district, Google Earth images®and different LULC features and their location points were recorded using a Global Positioning System (GPS) instrument.A maximum root mean square error of 0·5 pixels was achieved for Landsat MSS image.

Using the nearest neighbour method, images were resampled into a pixel size of 30 × 30 m Atmospheric correction was not applied in this study, because the adopted post-classification comparison approach compensated the variation in atmospheric conditions and vegetation phenology for different dates, mosaicking and subsetting of the image on the basis of Area of Interest (AOI) (Lillesand  and Kiefer, 2000).

Finally, color composite bands of RGB 4, 3, and 2 for the MSS and TM, whereas RGB 5, 4, and 3 band combinationforETM+ imageswere used and clipped so as to include the land use/land cover (LULC) classes indentified in the training sites and proceed to classify and quantify the  images in ERDAS IMAGINE® version 2014software.

#### *3.2.2.2.* Image pre-processing*and accuracy assessment*

For LULCclassification, a classification scheme, with six LULC classes namely, woodland, grazingland (the communal rangelands), enclosure, cultivated land, bushland and bareland cover types across the three study sites (Dida-Hara; Dharito and Obda).

The nomenclature of LULC classes was based on field observation and discussion with community elders and local experts as well as previous studies (e.g., Tsegaye et al., 2010; Elias et al., 2015; Abate and Angassa, 2016) ,with minor modification to fit to the objectives of the study (see Appendix 3.1).

Both unsupervised and supervised classification methods, with the Maximum Likelihood Classifier Algorithm (Pradhan et al., 2010) methods were employed to classify the individual images independently were used Classifications of imageries.

In the unsupervised classification, images were classified into pre-defined land use/land cover classes identified by the user by delimiting polygons around representative sites and clipping the region of interest (ROI) assigned at random of natural groupings based on the spectral characteristics with the highest similarity (Lillesand and Kiefer, 2000) and the results were used as a guide for the selection of training sites for supervise classification

Then, images were further classified using supervised classification using analyst-defined to determine the characteristics of each LULC types and overcome the problems of mixed pixels and misclassification. Filtering was not carried out in this study.This is because filtering might mislead by generalizing the classification to the extent that some mapping units for instance cultivated area and bareland covers could be underestimated.

Accuracy assessment determines the quality of the information derived fromthe remotely sensed data using error matrix and Kappa statistic (Congalton and Green, 2009). The stratified random sampling, where the number of points was stratified to the LULC types, was adopted in order to reduce the bias effect. Hence, a total of 250 pixels random points were selected for LULC mapping for theyears 1976, 1986 and 2013.

The visual interpretations of satellite images were conducted by analyst, groud reference data for the training sites and Google earth® image. A confusion matrix was produced as cross- tabulations of the classified data versus the reference data and was used to assess the classification accuracy. Afterwards, overall accuracy, user's and producer'saccuracies and the kappa statistic (*Khat* = Coefficient of Agreement) were performed for each lULC class in each year (Congalton and Green, 2009) from ERDAS.

### *3.2.2.3. Land use/land-cover change analysis*

The LULC change analysis was done using postclassification comparison method where Landsat image for each year (1976, 1986 and 2013) was classified and labeled independently, and then comparison was made using an overlay procedure (Lu et al., 2004).

The magnitudes and rates of changes in terms of LULC classes were determined using the following variables: total area (TA), changed area (CA) and change extent (CE)

The average change extent (CE %) was calculated following the formula adopted by Abate and Angassa (2016)

$$\mathrm{CE} = \frac{\mathrm{A}_2 - \mathrm{A}_1}{\mathrm{A}_1} \times 100 \qquad (3.1)$$

Where:
CE = Change extent (%)
$A_1$ = Amount of land use/land cover type in time 1 ($T_1$)
$A_2$ = Amount of land use/land cover type in time 2 ($T_2$)
$T_1$= study period at initial time
$T_2$= study period at final time.

A cross-tabulation module detection method was used to detect land use /landcover change (LULC) through which a LULC change matrix was produced, which enables to knowthe nature and spatial distribution of land use changes.The change/conversion matrix for three periods: 1976–1986, 1986–2013 and 1976– 2013 were also used to drive the gains and losses for land covercategories.

The gains for each class were derived by subtracting the persistence from the column total (final state) and the loss for each class was derived by subtracting the persistence from the row total (initial state). The land use/land cover analyses were carried out in ERDAS IMAGINE®version 2014. The land use/land cover classification for three study sites was computed altogether.

### *3.2.2.4. Experimental design for field data collection*

A paired-site design approach was used in this study (detailed under general materials and methods section in **chapter I**). Accordingly, four different land use types (woodland, cultivated land and open-grazed-rangelands [communal], and enclosure [protected rangelands]) from adjacent sites, with similar slope, elevation, and land aspect were selected.

The selected land use types were apparently representatives of the current land use systems in the area and their influence on carbon and nitrogen stocks dynamics and related soil properties were assessed and replicated three times across three sites (Dida-Hara; Dharito and Obda) in order to capture the variability in terms of soils and management practices of land use.

According to on-site discussion with local community elders, crop fields received no farm input such as fertilizer, since they were brought to cultivation. Field data collection was carried out between July and August 2013.

Hence, a total of 12 paired sites = [(4 land use types) ×3 site (replicates)] were used in this study. At each replicate site, we established 10 sampling plots (30 × 30 m each) along a 500 m long line transect in each of the four land use types.

The plots were well spaced and put at an interval of 50 m along a transect line to account for the spatial variability of the soil, while keeping, 20  to 30 m distance away from the land use perimeter to avoid edge effects.

### *3.2.2.5. Soil sampling and Laboratory analysis*

In each plot, random soil samples were collected using auger (Edelman auger, 7.6 cm in diameter) at 15 randomly distributed auger points (sub-plots) from soil depths of 0–15 and 15–30 cm. Hence, a total of 240 composite soil samples :[( 4 land use types) ×3 site (replicates) × 10 plots × 2 depths] were collected across the study area.

In addition, cumulative masses of soil samples were collected from four sampling points (replicates) in each plot at the similar depth range using auger with a diameter of 7.6 cm, clearly marked to determine the soil bulk densit (Aynekulu et al., 2011).A cumulative soil mass sampling plate was used to collect any soil that fell out of the auger during sample collection.

The details of the procedures soil samples processing, laboratory analysis, calculation of SOC and TN stocks were presented in detailsunder general materials and method section in **chapter I).**

### *3.2.2.6. The SOC and TN stocks change due to land use changes*

There is a lack of long term monitoring carbon stocks changes attributable to land use /land cover changes  in many arid and semi-arid parts of Africa (Reid et al.,2004; Cias et al.,2010), as this aslo the case in Borana rangelands of southern Ethiopia.

Thus, comparisons were made between adjacent patches of land with different land cover, and a known history of use, with similar climate, slope aspect and altidue.  In this study, we did not set the base line and it was assumed that the SOCof the same land use type remains constant over thestudy periods.

The four land use types selected from adjacent sites were similar with respect to climatic factors, the type of soil, slope, elevation, and drainage, but under different land use systems. On-site observation and discussions with land owners showed that no substantial differences in soil management practices have occurred over the last 30 years

Thus, change in SOC and TN stocks over time was due to changes in land use types.Inthis respect, when a ha of agiven land use /land cover (e.g., woodland) was converted to other land use /land cover types, in this case (cultivated land), change inSOC stock was eastimated as the difference in SOC stock under cultivated and that of the woodland then multiplied by the mangnitude of area of woodland converted to cultivated land and viceversa (Carré et al., 2010).

$$SOC_{c} = (SOC_{i} - SOC_{j}) \times A_{ij} \qquad (3.2)$$

Where, SOCc is the SOC stocks change caused by the transfer out of and/ or transfer in of land use types i and j using the land use transition matrices, $SOC_i$ and SOCj are the mean SOC stocks of land use types i and j, Aij is the area converted from land use type i to land use type j during the study period. Then the same calculations were done for the TN stocks.

### *3.2.2.7. Statistical analysis*

Prior to data analysis, we tested our data to set for normality by producing a histogram and a normal probability plot of residuals. The differences in SOC and TN stocks and related soil properties in different land use types were analyzed using one-way analysis of variance (ANOVA) in linear mixed effect procedure for each depth (0-15 and 15-30 cm) and to 30 cm soil depth (i.e., comparing land uses as the fixed effect and plots as the random effect).

Comparison of means among the four land use/land cover changes at each soil depth was carried out using the Tukey’s HSD test (adjusted for p-significance at $P < 0.05$). Results were reported as mean ± standard error, unless otherwise specified

## 3.3. RESULTS

### *3.3.1. Land use /land cover changes and accuracy assessment*

The land use/land cover maps were classified correctly and independently according to the selected land use classes including cultivated land, woodland, communal grazingland (the communal rangelands),enclosure, bush land and bareland for each period (1976, 1986 and 2013).

Results on accuracy assessment for each classified land cover types showed an overall accuracy of 85, 85.83 and 91.67% and the corresponding Kappa coefficients of 0.74, 0.81 and 0.90, respectively, for the three image maps using the Landsat MSS for 1976, TM for1986 and ETM+ for 2013.

The results also showed a high rate of agreement between the user's accuracy (UA) and producer's accuracy (PA) for the woodland, bushland, communal grazingland and enclosure. In the case of bareland, users' accuracy and producer's accuracy were 68 and 57.14%, respectively, and with users' accuracy of 76% and producers' accuracy of 75% for the cropland (Appendix 3.1).

The dynamics and rate of land use/land cover analysis for the study sites were obtained by interpretation of Landsat data (MSS, 1976, TM, 1986 and ETM+ 2013) as presented in Table 1 and Figures 3.1(a-c).According to the produced LULC mapsthere were six major land use/landcover classes with a total area of 125,873.16 ha.

However, the significant spatial expansion in cultivated and bush land cover types and the Computed percentages of LULC classes showed that in the year 1976, the communal grazingland comprised the largest area coverage (53.9%) followed by grassland (enclosure) (22.3%), while the remaining balances were covered by woodland (12.6%), andcultivated, bushland and bareland occupied o about 2.03, 4.8 and 4.4% that order (Table 3.1).

In the year 1986, the area coverage of cultivated land and bush cover were increased to 25.0 and 17. %, respectively, while the communal grazing, woodland and eclosures were declined  to 36.1, 7.2 and 8.8%, respectively as compare to the base year ( 1976) (Table 3.1).

In the year 2013, the area under enclosure in the grassland increased to 16.2%, whereas the area coverage under communal grazinglands and woodland declined as compared to the base year (1986).

Overall the last 37 years (1976-2013), the proportion of bushland showed substantial increase (260.3%)and that of cultivated land increased more than tenfold (945%) as compared to the base year (1976) (Table 3.1)

Table 3. 1. Magnitude and rate of land use/land cover changes (ha) across the three periods (1976, 1986 and 2013) in the study areas from Yabello district in Borana rangelands

| | 1976 | | 1986 | | 2013 | | 1976-1986 | | 1986-2013 | | 1976-2013 | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| LULC class | TA(ha) | (%) | TA(ha) | (%) | TA(ha) | (%) | CA (ha) | CE (%) | CA (ha) | CE (%) | CA (ha) | CE (%) |
| Woodland | 15864.4 | 12.6 | 9069.1 | 7.2 | 6948.2 | 5.5 | -6795.3 | -42.8 | -2121 | -23.4 | -8916.2 | -56.2 |
| Bushland | 6013.6 | 4.8 | 22187.5 | 17.6 | 21664.7 | 17.2 | 16173.9 | 269 | -522.8 | -2.4 | 15651.1 | 260.3 |
| Grazingland (communal) | 67893 | 53.9 | 45421.7 | 36.1 | 41533.5 | 33 | -22471.3 | -33.1 | -3888.2 | -8.6 | -26360 | -38.8 |
| Enclosure(grassland) | 28060.8 | 22.3 | 11040.1 | 8.8 | 20407.5 | 16.2 | -17020.7 | -60.7 | 9367.4 | 84.9 | -7653.3 | -27.3 |
| Cultivated land | 2556.6 | 2 | 31415 | 25 | 26754.8 | 21.3 | 28858.4 | 1128.8 | -4660.2 | -14.8 | 24198.3 | 946.5 |
| Bareland | 5484.7 | 4.4 | 6739.8 | 5.3 | 8564.5 | 6.8 | 1255 | 22.9 | 1824.8 | 27.1 | 3079.8 | 56.2 |
| Total | 125873 | | 125873.2 | | 125873 | | | | | | | |

Notes: TA –total area, CA-change area, CE-change extent

General patterns of the LULC class identified and different classes were represented as different colours in each image, making it easy to identify (Fig. 3.1a-c).  Across the LULC maps, the arewa was dominated by the grazingland (communal). The LULC map of 1976 (Fig.3.1a) also illustrated that the predominant types of LULC class was grazingland (the communal rangelands) followed by enclosures and woodland.

According to the LULC map for the year 1986(Fig. 3.1b), it was found that cultivated land increased in magnitude. Furhtermore, the LULC map of 2013 also showed that there was progressive increase for bushland cover, whereas woodland was dramatically reduced (Fig.3.1c)

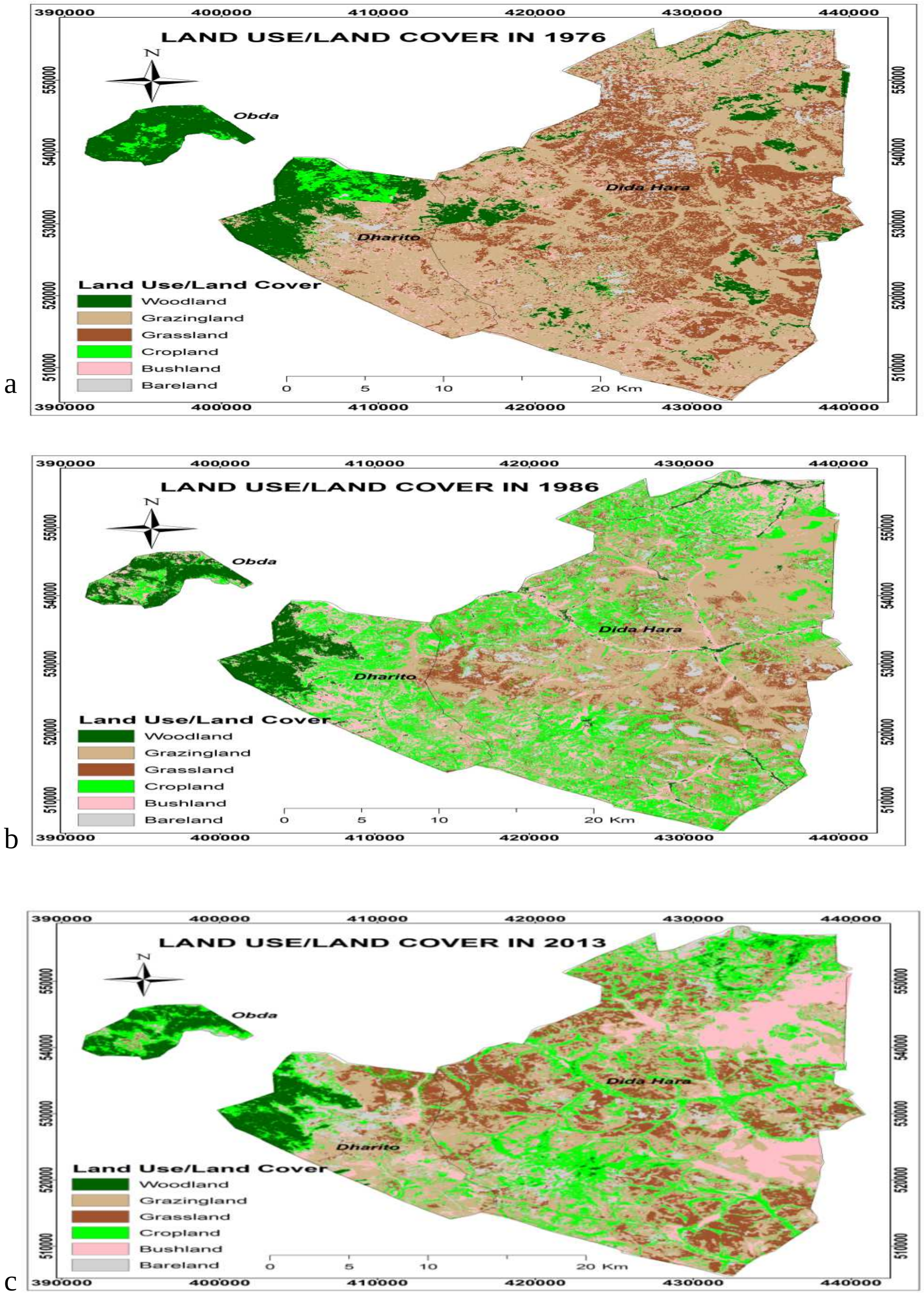

Figure 3.1Maps showing land use /land cover for the periods of 1976, 1986 and 2013 in the study sites from Yabello district in Borana rangelands of southern Ethiopia

### *3.3.2 Land use /land cover transition matrix*

The LULC change detection statistics for three periods (i.e., 1976- 1986, 1986-2013 and 1976- 2013) in the studysites are presented in Table 3.2(a-c). The results showed the dynamics of land use /land civer transtion from one category to another, with bushland, cultivated land and bareland were the net change gainers, whereas woodland and the communal grazingland were the net change losers.

On the otherhnad, the bolded diagonal elements from Table 3.2 (a-c) indicated the proportions of LULC classes that were static (persisted) and hence the net change to persistence (Np) closer to the value of ‘zero,” indicates a higher tendency to persist to change.

Accordignly, major conversion of of land was recorded from grazingland (the communal rangelands) to cultivated land (18,280.10 ha), bushland (15,180.20 ha) and enclosureduring the period from 197 to 1986 (Table 3.2a). During the same period, however, about 33% of total of the study area persistent to change, while the remaining balance (67%) of the land use types was vulunerable to change 1976 and 1986, (Table 3.2a).

Likewise, during the period from 1986–2013 grazingland further converted to bushland (13393.8 ha), enclosure land (8985.62 ha) and cultivated land (6589.19ha).During the same time period,the maganitude and rate of changes in land use/land cover change was found high with a total of 72% of the total areas in the study sites showing internal trading (Table 3.2b). During the same period (1986-2013), however, bush and cultivated lands had negative and closer to zero values of Np, while for enclosure the value was positive and relatively higher Np value (Table 3.2b).

Overall the last 37 years (1976 to 2013), cultivated gained (26082.36 ha), mainly through conversions of the communal grazingland (61.76%), woodland (15.3%), and enclosure (15.2% (Table 3.2c).The net change to persistent ratio (Np) for bushland (positive) and cultivated land (positive), whereas it was negative and closer to zero for the remaining LULC classes).

Furthermore, the communal grazing land was the most persistent, suffered the highest loss followed by woodland, whereas bush and cultivated lands showed the lowest persistence, with both have also with lowest amount of loss. Overall, the largest proportion (70%) in the study sites experienced land use /land cover dyanmcis, whereas only about 30% (i.e., sum of the diagonal elements) showed perisistance to change (Table 3.2c).

Table 3.2. Transition matrix showing land use /land cover changes for the periods of 1976, 1986 and 2013 in the study sites from Yabello district in Borana rangelands of southern Ethiopia

| | **(a)1976-1986** | | | | | | |
|---|---|---|---|---|---|---|---|
| | WL | BuL | GL | GR | CL | BaL | 1986 total |
| WL | **7,090.49** | 49.59 | 1,717.02 | 73.71 | 118.26 | 20.07 | 9,069.14 |
| BuL | 2,678.48 | **1,423.59** | 15,180.20 | 1,608.93 | 832.14 | 464.13 | 22,187.47 |
| GL | 2,764.73 | 1,311.03 | **25,700.66** | 13,727.20 | 105.39 | 1,812.69 | 45,421.70 |
| GR | 801.12 | 342.54 | 4,698.18 | **4,535.87** | 38.25 | 624.15 | 11,040.11 |
| CL | 2,200.86 | 2,769.48 | 18,280.10 | 5,403.33 | **1,360.01** | 1,401.21 | 31,414.99 |
| BaL | 328.73 | 117.36 | 2,316.87 | 2,711.79 | 102.51 | **1,162.49** | 6,739.75 |
| 1976 total | 15,864.41 | 6,013.59 | 67,893.03 | 28,060.83 | 2,556.56 | 5,484.74 | **412, 73.11 (33%)** |
| Gain | 1,978.65 | 20,763.88 | 19,721.04 | 6,504.24 | 30,054.98 | 5,577.26 | |
| Loss | 8,773.92 | 4,590.00 | 42,192.37 | 23,524.96 | 1,196.55 | 4,322.25 | |
| Net change | -6,795.27 | 16,173.88 | -22,471.33 | -17,020.72 | 28,858.43 | 1,255.01 | |
| Np | -0.96 | 11.361 | -0.87 | -3.75 | 21.22 | 1.08 | |

| | **(b) 1986-2013** | | | | | | |
|---|---|---|---|---|---|---|---|
| | WL | BuL | GL | GR | CL | BaL | 2013 total |
| WL | **5595.13** | 1026.14 | 28.89 | 0.2 | 297.16 | 0.65 | 6,948.17 |
| BuL | 406.14 | **3439.17** | 13393.8 | 1073.66 | 3052.55 | 299.34 | 21,664.66 |
| GL | 140.52 | 6473.63 | **14578.1** | 5260.39 | 12672.2 | 2408.63 | 41,533.47 |
| GR | 12.22 | 1102.45 | 8985.62 | **3134.41** | 5007.53 | 2165.31 | 20,407.54 |
| CL | 2805.51 | 8935.38 | 6589.19 | 836.76 | **7303.16** | 284.81 | 26,754.81 |
| BaL | 109.62 | 1210.7 | 1846.1 | 734.69 | 3082.39 | **1581.01** | 8,564.51 |
| 1986 total | 9069.14 | 22187.47 | 45421.7 | 11040.11 | 31414.99 | 6739.75 | **35630.98 (28%)** |
| Gain | 1,353.04 | 18,225.49 | 26,955.37 | 17,273.13 | 19,451.65 | 6,983.50 | |
| Loss | 3474.01 | 18748.3 | 30843.6 | 7905.7 | 24111.83 | 5158.74 | |
| Net change | -2,120.97 | -522.81 | -3,888.23 | 9,367.43 | -4,660.18 | 1,824.76 | |
| Np | -0.38 | -0.15 | -0.27 | 2.99 | -0.64 | 1.15 | |

Table 3.2 (continued)

| **(c) 1976-2013** | | | | | | | |
|---|---|---|---|---|---|---|---|
| | WL | BuL | GL | GR | CL | BaL | 2013 total |
| WL | **5,953.34** | 24.12 | 854.89 | 16.79 | 90.7 | 8.33 | 6,948.17 |
| BuL | 1,990.43 | **475.01** | 15,196.37 | 3,250.67 | 231.03 | 521.19 | 21,664.66 |
| GL | 2,139.53 | 2,606.18 | **22,497.74** | 11,309.59 | 789.62 | 2,190.78 | 41,533.47 |
| GR | 1,298.61 | 1,316.18 | 9,035.30 | **7,398.36** | 410.4 | 948.69 | 20,407.54 |
| CL | 3,993.37 | 1,092.85 | 16,107.89 | 3,969.41 | **672.45** | 918.83 | 26,754.81 |
| BaL | 489.13 | 499.25 | 4,200.84 | 2,116.01 | 362.36 | **896.92** | 8,564.51 |
| 1976 total | 15864.41 | 6013.59 | 67893.03 | 28060.83 | 2,556.56 | 5,484.74 | **37893.82 (30%)** |
| Gain | 994.83 | 21,189.65 | 19,035.73 | 13,009.18 | 26,082.36 | 7,667.59 | |
| Loss | 9,911.07 | 5,538.58 | 45,395.29 | 20,662.47 | 1,884.11 | 4,587.82 | |
| Net change | -8,916.24 | 15,651.07 | -26,359.56 | -7,653.29 | 24,198.25 | 3,079.77 | |
| Np | -1.5 | 32.95 | -1.17 | -1.03 | 35.99 | 3.43 | |

Notes: WL-woodland, BuL-bushland, GL-grazing land, GR-Enclosure, CL-cultivated Net change = gain−loss, Np refers ratio of net change to persistence ratio (i.e., net change/ bolded diagonals of each class).All the figures in the table are in hectare except Np which is a ratio

### *3.3.4. Effects of land use systems on selected soil properties, SOC and TN contents and stocks*

The results showed that sand fraction was lower in woodland and higher in the communal grazingland (Table 3.3).Conversely; we found that the clay fractions were higher in the woodland plots and lower in both the cultivated and communal grazingland.

The results of this study aslo revealed that significant ($P < 0.05$) difference was observed only between woodland and cultivated land in terms of clay fraction in both soil depths, whereas no significant difference in terms of sand and silt fractions. Overall, texture of the soil for the investigated land use types was categorized under sandy clay loam, with sand fraction being the dominant (Table 3.3).

The results showed no significant ($P > 0.05$) difference in terms of the overall mean soil bulk density between land use types. However, we recorded a significantly higher bulk density value in the communal grazingland than in the cultivated land (15-30 cm) depth (Table 3.3).

Furthermore, the mean value for the bulk density was higher in the communal grazingland than the value recorded in the woodland (Table 3.3). Generally, the mean value for bulk density showed a decrease in the communal grazingland, enclosure, cultivated land and woodland in that order (Table 3.2).

Higher amounts of soil SOC contents were e observed at two depths of soil for the woodland and enclosure than that for the grazing and cultivated land use types (Table 3.2). Moreover, woodland had higher SOC content than enclosure and the grazingland had higher SOC content than cultivated land, but did not differ significantly across the two soil depths (Table 3.3).

Generally, there was not significant ($P > 0.05$) difference between woodland and enclosure in terms of SOC stock, although the value was relatively higher for the woodland. Likewise, the communal grazingland had a higher SOCstock than the cultivated land, but with no significant difference in both soil depths (Table 3.3).

The results showed a significant ($P < 0.05$) difference in total nitrogen (TN) content (%) between woodland and cultivated land soils, as well as woodland and enclosures. However, there was no significant difference between the woodland and communal grazingland in the 15-30 cm soil depth (Table 3.3). Overall, we found a higher TN content in the communal grazingland than in the enclosures and cultivated land use types (Table 3.3).

Similarly, the results showed that the mean total nitrogen (TN) stock was significantly ($P < 0.05$) higher in the woodland than in the other land use types across both soil depths. Moreover, enclosures had higher TN stock than cultivated land with no significant difference in both soil depths (Table 3.3).

The carbon to nitrogen (C: N) ratio was significantly ($P < 0.05$) higher in enclosures than in the communal grazingland and cultivated land in both soil depths. Moreover, enclosures had relatively higher C:N ratio than the woodland, communal grazingland, and cultivated land across soil depths (Table 3.3).

Table 3.3. Comparisons of selected soil properties, SOC and TN contents and stocks in different use types at two soil depths in the study sites from Yabello district in Borana, southern Ethiopia.

| soil parameters | depth (cm) | Woodland | Grazingland | Enclosure | Cultivated land |
|---|---|---|---|---|---|
| Sand (%) | 0-15 | 50.9±2.5 | 56.9±4.5 | 54.20±3.1 | 58.7±2.7 |
| | 15-30 | 50.0±2.5 | 56.6±4.4 | 54.0±3.0 | 58.1±2.7 |
| Silt (%) | 0-15 | 19.4±1.6 | 19.2±3.0 | 21.2±2.0 | 17.10±1.7 |
| | 15-30 | 18.8±1.6 | 19.5±3.1 | 20.0±1.9 | 16.3±1.8 |
| Clay (%) | 0-15 | 29.7±1.6$^{ab}$ | 23.9±1.1$^{c}$ | 24.6±1.2$^{bc}$ | 24.2±1.0$^{c}$ |
| | 15-30 | 31.2±1.5$^{ab}$ | 23.9±1.4c | 26.0±1.3b | 25.6±0.9$^{b}$ |
| BD (gm cm$^{-3}$) | 0-15 | 1.45±0.08 | 1.59±0.07 | 1.50±0.06 | 1.54±0.07 |
| | 15-30 | 1.54±0.08$^{c}$ | 1.68±0.06$^{ab}$ | 1.63±0.06$^{bc}$ | 1.58±0.07$^{bc}$ |
| SOC (%) | 0-15 | 1.390.10$^{ab}$ | 0.83±0.10$^{c}$ | 1.24±0.10$^{b}$ | 0.83±0.07$^{c}$ |
| | 15-30 | 1.23±0.10$^{ab}$ | 0.79±0.10$^{c}$ | 1.07±0.10$^{b}$ | 0.73±0.07$^{c}$ |
| SOC stock (Mg ha$^{-1}$) | 0-15 | 29.15±1.86$^{ab}$ | 22.97±1.98$^{cd}$ | 25.80±1.57$^{bc}$ | 19.94±1.30$^{d}$ |
| | 15-30 | 26.79±1.79$^{ab}$ | 22.82±2.130$^{cd}$ | 24.23±1.60$^{bc}$ | 18.16±1.23$^{d}$ |
| TN (%) | 0-15 | 0.16±0.01$^{a}$ | 0.13±0.01$^{b}$ | 0.13±0.01$^{b}$ | 0.13±0.01$^{a}$ |
| | 15-30 | 0.14±0.01$^{ab}$ | 0.12±0.01$^{bc}$ | 0.11±0.01c | 0.10±0.01$^{c}$ |
| TN stock (Mg ha$^{-1}$) | 0-15 | 3.81±0.25$^{ab}$ | 3.25±0.26$^{b}$ | 3.09±0.18$^{bc}$ | 2.84±0.19$^{c}$ |
| | 15-30 | 3.71±0.21$^{ab}$ | 3.20±0.24$^{b}$ | 3.08±0.20$^{bc}$ | 2.74±0.18$^{c}$ |
| C:N ratio | 0-15 | 8.73±0.49 | 6.68±0.48 | 10.26±0.78 | 7.50±0.35 |
| | 15-30 | 8.41±0.48 | 6.62±0.54 | 9.70±0.76 | 7.22±0.46 |

Notes: SOC-Soil organic carbon, TN- totals Nitrogen, BD- Bulk Density, C: N- ratio of Carbon to Nitrogen. Mean values (±SE) followed by different letters within a row indicate a significant difference at $P< 0·05$ different between different land use types at each depth (tested by one-way ANOVA with proc mixed model, Tukey's (adjusted) test at $P< 0.05$).

Overall, the total SOC content across thje 0-30 cm depth showed significantly higher under woodland (1.31±0.07%) enclosure (1.16±0.06%) than in the communal grazin gland grazingland (0.81±0.06%) and cultivated land (0.79±0.05%) (Fig.3.2a).

Likewise, ttheSOC stocks in the 0-30 cm depth showed that woodland (55.90±3.4 Mg ha$^{-1}$) and enclosure (50.03±3.03 Mg ha$^{-1}$) had higher SOC than the communal grazing land

(45.79±4.00 Mg ha$^{-1}$) and cultivated land (38.10±2.47 Mg ha$^{-1}$).However, significantlydifference in terms of SOC and stocks was obtained between woodland and communal grazingland, woodland and cultivated lands, as well as between enclosures and cultivated lands in terms (Fig.3.2 (a&b).

The mean total nitrogen (TN) contents in the 0- 30 cm depth were higher under woodland (0.15±0.04%), followed by communal grazingland and enclosures woodland each (0.12±0.04%) and than the cultivated land (0.11%±0.04). However, significantly higher TN content was found only in the woodland as compared to the remaining land use types (Fig.3.2c).

The results showed a decline in TN stocks (0-30 cm depth) from the woodland towards the communal grazingland, enclosures and cultivated land in that order. However, a significantly ($P < 0.05$) higher TN stock was observed in woodland than in the cultivated land use type (Fig. 3.2d).

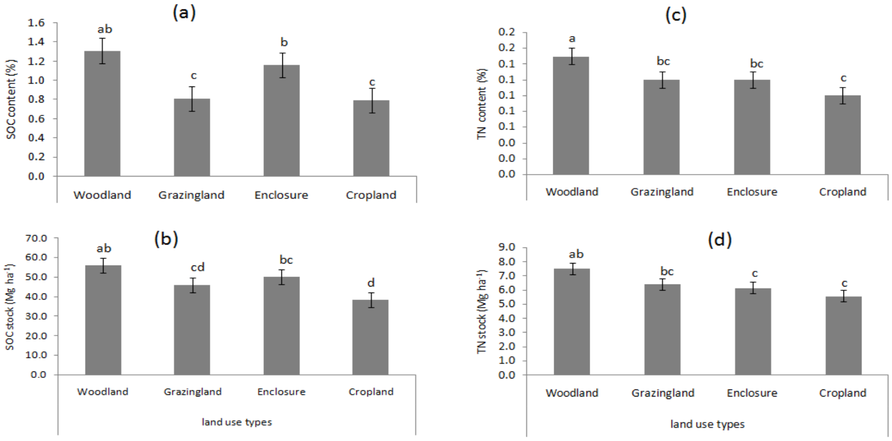

Figure3. 2. Mean comparisons of SOC and TN contents and stocks between different use types when the data pooled to 30 cm soil depths

Note: Different letters above bars indicate significan ($P < 0.05$) between different land use types, after Tukey's test (adjusted) at $P < 0.05$ signficant level. Error bars indicate standard error of the mean.

### *3.3. 4. Estimated SOC and TN stock changes due to land use change*

Based on the area weighted average of the four land use types, the total SOC stocks were estimated at 5.50, 4.34 and 4.33 Tg ,respectively for the year 1976, 1986 and 2013 (Fig.3.3). Likewise, the total TN stocks across the same periods were: 0.74, 0.60 and 0.59, respectively (Fig.3.3).

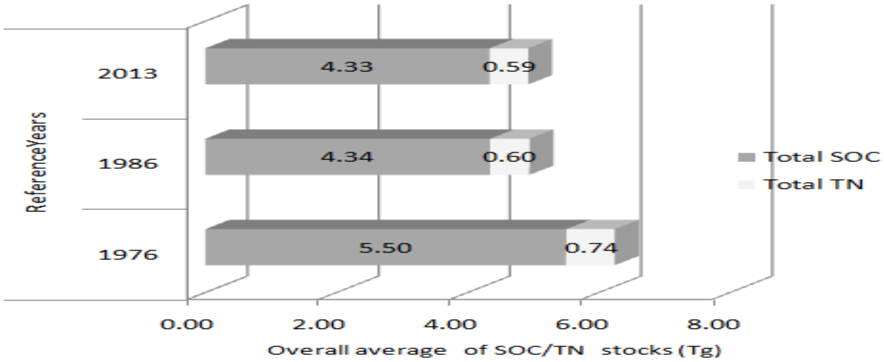

Figure3. 3. Total SOC and TN stocks estimated across the different land use types (based on the areaweighted coverage) in the study area for the period of 1976, 1986 and 2013

Nonetheless, LULC changes resulted in potential loss of SOC stock estimated at 319.51 (equivalent to 7.02 Tg $yr^{-1}$) and TN stocks of 30.65Gg (rate= -0.32 0.10 Gg $year^{-1}$) due to land use change for the period between 1976 and 2013(Table 3.4)

Conversely, there was again in terms of SOC stock estimated at 59.63 Gg (rate= 1.61 Gg $yr^{-1}$) due to conversion of the communal grazingland to woodland and enclosure as well as cultivated land to the remaining land uses types from 1976 to 2013 (Table 3.4).

Likewise, our results showed a total of 18.80 Gg gain (0.5Gg $yr^{-1}$) in TN stocks between 1976 and 2013. During the same period, there was also a gain (increase) in TN stocks with the conversion of cropland and enclosure to woodland and grazingland cover types (Table 3.4). Accoridngly, conversion of communal grazingland to cultivated lands accounted for 39% (123.87 Gg) followed by conversion of woodland and enclosures to cultivated land accounted for 37% (118.44 Gg) of the total loss (Table 3.4).

During the transfer out of land use types, the woodland resulted in carbon loss, whereas during the transfer in cropland wwas carbon loss, whereas both carbon gain and loss for the transfer out and in enclosure and the commual grazingland were observed in this study (Table 3.4).

Thus, results showedthe potential changes in terms of both positive (gain) and negative (loss) inSOC and TN stocks, which entirely based on weighted area conversion as estimated in LULCconversion matrix

Table 3.4. Potential change (gain/loss) in soil organic carbon (SOC) and total nitrogen (TN) stocks due to land use change/conversion from 1976 to 2013 in the study area (Yabello district), Southern Ethiopia

|  |  | 1976-2013 | Mean difference (j-i) (Mg ha$^{-1}$) |  | Relative change (%)* |  | 1976-2013 Total change (Gg) |  |
|---|---|---|---|---|---|---|---|---|
| land use type (i) (from) | land use type (j) | Changed area (ha) | SOC | TN | SOC | TN | SOC | TN |
| Commual grazingland | Woodland | 854.89 | 10.11 | 1.08 | 22.08 | 16.77 | 8.64 | 0.92 |
|  | Grassland (Enclosure) | 9,035.30 | 4.24 | -0.34 | 9.26 | -5.28 | 38.3 | -3.07 |
|  | Cultivated land | 16107.89 | -7.69 | -0.85 | -16.79 | -13.2 | -124 | -13.69 |
|  | Net change |  |  |  |  |  | -76.9 | -15.84 |
| Woodland | Grazingland | 2139.53 | -10.11 | -1.08 | -18.09 | -14.36 | -21.6 | -2.31 |
|  | Grassland (Enclosure) | 1298.61 | -5.87 | -1.42 | -10.5 | -18.88 | -7.62 | -1.84 |
|  | Cultivated land | 3993.37 | -17.8 | -1.93 | -31.84 | -25.66 | -71.1 | -7.71 |
|  | Net change |  |  |  |  |  | -100 | -11.86 |
| Enclosure(grassland) | Grazingland | 11,309.59 | -4.24 | 1.42 | -8.47 | 23.28 | -48 | 16.06 |
|  | Woodland | 16.79 | 5.87 | 0.34 | 11.73 | 5.57 | 0.1 | 0.01 |
|  | Cultivated land | 3969.41 | -11.93 | -0.51 | -23.85 | -8.36 | -47.4 | -2.02 |
|  | Net change |  |  |  |  |  | -95.2 | 14.04 |
| Cultivated land | Grazingland | 789.62 | 7.69 | 1.93 | 20.18 | 34.53 | 6.07 | 1.52 |
|  | Woodland | 90.7 | 17.8 | 0.85 | 46.72 | 15.21 | 1.61 | 0.08 |
|  | Grassland (Enclosure) | 410.4 | 11.93 | 0.51 | 31.31 | 9.12 | 4.9 | 0.21 |
|  | Net change |  |  |  |  |  | 12.6 | 1.81 |
| Total gain |  |  |  |  |  |  | 59.6 | 18.8 |
| Total loss |  |  |  |  |  |  | -320 | -30.65 |
| total Net change |  |  |  |  |  |  | -260 | -11.85 |
| SOC /TN acculation rate in time intervals (Gg yr$^{-1}$) |  |  |  |  |  |  | -7.02 | -0.32 |

Notes: WL-Woodland, GL-Grazingland, GR-Enclosure (grassland), CL-Cultivated land, SOC- soil organic carbon, TN –total nitrogen. Negative value showed a decrease in SOC and TN stocksfor due to land use change , Gg –Giga gram= 1000 Mega gram

* Relative change (%) Land type (j) - land use type (i)/land use type (j) x 100,

## 3.4. DISCUSSIONS

### *3.4.1. Lad use /land cover changes and accuracy assessment*

The overall land use/land cover classification accuracy and kappa statistics for those Landsat images of 1976 to 2013 generally suggest a good conformity between the classification and land use/land cover categories. The accuracy level was within the 85% to 90% classification accuracy standards used by Chen (2002) and Kiage et al. (2007).

The present results indicate that the conversion of one land use type to another over the last 37 years is not linear rather it is dynamic in nature. Generally, an increase in the cover of cultivated land and bushland are through the reduction of communal grazing land and savanna woodland.

Among the study areas, Dida-Hara and Dharito were well known for their good stands of savanna grasslands in the past as used for extensive communal grazing areas during the dry season by the local communities. Currently, these areas are under stress due to increased pressure from cultivated land and yearly grazing that contributed to the dynamics of land use changes.

Currently, these areas are under stress due to increased pressure from cultivated land and yearly grazing that contributed to the dynamics of land use changes. Conversion of the communal rangelands to crop cultivation is an emerging issue through land use change in the study areas.

This type of land use change is seen by many as a strategy for livelihood diversification of the local communities. Abate and Angassa (2016) have reported the situation of land use conversion towards cropping as one of the mitigation strategies against the economic hardship associated with increasing population pressure and recurrent drought in Borana, southern Ethiopia.

According to Elias et al. (2015), crop cultivation is usually practiced through shifting land use that involves clearing of woodlands where the land remains abandoned after one or two cropping seasons and eventually replaced by shrubs.

The current results are consistent with the findings of others (Tsegaye et al., 2010; Belay et al., 2013) suggesting a substantial reduction in the cover of savanna rangelands and woodland in arid and semi-arid areas of Ethiopia.

A similar study (Maitima and Mugatha, 2009) has shown that the majority of land use changes in Africa as a result of the conversion of natural vegetation to farmlands and human settlements. The observed dynamics in terms of various land use types and rate of change over the study period could be a result of anthropogenic factors.

Reid et al. (2004) have also indicated that agricultural expansion related to government policy has been one of the key driving forces for the dynamics of land use/land cover changes especially in East Africa and many tropical countries (Mugagga et al., 2012).

A substantial increase in terms of bush cover dynamics observed in the current study is in line with previous studies (Dalle et al., 2006; Angassa and Oba, 2008), that suggest that multiple set of factors including suppression of fire and heavy grazing as well as episodes of climatic factors could be among the drivers of changes.

Thus, the continued decline in woodland cover over the last four decades perhaps linked to the ever increasing human exploitation of woody plants for different purposes (i.e., charcoal making, fuel-wood collection, construction and land clearing for crop production).

In consistent with the present findings, previous studies (Tsegaye et al. 2010; Debeko et al., 2018) have associated the reduction in vegetation cover in different parts of Ethiopia to the increasing demand for socio-economic needs due to demographic pressure. Moreover, Tsegaye et al. (2010) also reported that woodland vegetation in the Afar Region of Ethiopia declined by 8.1% between 1972 and2003.

The observed increase in the trend of bareland in this study is concurrent with the findings of Elias et al. (2015), suggesting that a rapid expansion of bareland into key grazing areas is probably attributed to the expansion of farming activities, shrinkage of grazingland followed by heavy grazing and loss of vegetation cover.

### *3.4.2. Land use/land cover transition matrix*

The results of the current study indicate a transition from one land use type to other categories, which accounts for 70% of the study areas and this is in line with the findings of others (Tsegaye et al., 2010; Abate and Angassa, 2016) from arid and semi-arid rangelands of Ethiopia.

Overall, the present findings have shown a rapid increase in the expansion of cultivated land and bush cover in the study areas with increased shrinkage and deterioration of the communal rangelands. A study by Belay et al. (2013) from the Hammer district of south-western Ethiopia has reported a similar pattern of land use/land cover changes.

The same authors have reported an increase in woody vegetation cover (i.e. woodland, bushland and shrubland) with the reduction in grasslands. Overall, there is a rapid increase in the expansion of cultivation and bush cover at the expense of the communal rangelands in the study areas.

As a result, the dynamics of land use changes could greatly affect the status and quality of soil properties, carbon and nitrogen content in the system. For example, Angassa and Oba (2008) have suggested the re-introduction of prescribed fire and selective clearing of invasive woody plants for the reclamation of degraded rangelands.

### *3.4.3. Effect of land use on soil properties, SOC and TN contents and stocks*

Despite the similar proportion of soil particles, clay fraction was higher in woodland soils than other land use types. This suggests that the presence of trees may be in reducing the fine fraction through erosion, which has already demonstrated by significantly higher SOC under the woodland and enclosure land use types.

Similarly, Kiflu and Beyene (2013) also reported that there was no significant difference in sand content although the silt and clay fractions were significantly varied between different land use types in southern Ethiopia. Moreover, changes in land use patterns also have an indirect impact on the variation in soil texture through its effects in terms of exposing the soil to agents of erosion (Mesele, 2006).

The findings indicate that the textural class of the soil was sandy clay loam across the adjacent land use types, suggesting that soil texture is an inherent soil property that is not influenced within a short period of time. However, Brady and Weil (2002) argued that pedogenic processes such as erosion, deposition, eluviation, and weathering can change the soil texture over a long period of time.

The current findings indicate no significant difference in terms of soil bulk density between the different land use types. In contrast to the present findings others (e.g., Lemenih, et al., 2005; Mesele, 2006) have reported that soil bulk density significantly varied with land use types because of differences in land management and land use histories.

In the current finding, we recorded more mean value for soil bulk density in the open-grazed rangelands than in the woodland might be attributed to the impact of heavy grazing, soil degradation and poor land management. Zhang et al. (2014) also reported an increase in soil bulk density as a result of heavy grazing especially in degraded rangelands.

In line with our findings, Mesele (2006) also has reported higher values of soil bulk density in grassland (1.40 gm cm$^{-3}$) and degraded bushland (1.85 gm cm$^{-3}$) in Borana rangelands of southern Ethiopia. However, the normal range of bulk density for mineral soils is ranging from 1.3-1.4 gm cm$^{-3}$ (Bohn et al., 2001).

The probable reasons for the low SOC content and stock in the cultivated land as compared to protected woodland and enclosures in the present study could be due to increased disturbance through cultivation and compaction that may be reducing organic matter input into the soil system. Our findings agree with the results by others (Yimer et al., 207; Fuetal, 2010) that suggest that SOC concentrations are lower in croplands.

Furthermore, the lower SOC content and stock under open-grazed land might be because of increased grazingland/or browsing pressure that lead to reduced organic matter input. Recnt studies from the same study areas (Aynekulu et al., 2017;Feyisa et al., 2017) have similarly indicated that the open-grazed rangelands have accumulated lower SOC content and stocks than enclosures most likely due to the impact of continuous grazing in the open-grazed areas.

In consistent with the current findings, Dabasso et al. (2014) also found higher SOC stock in shrubland followed by grassland in a semi-arid pastoral area of Kenya. Liu et al. (2010) also argued that the type of vegetative cover is one of the factors influencing soil organic carbon content.

Generally, the mean SOC contents in the 0-30 cm depthin cultivated land (0.79%) and open-grazed land (0.81%) fall within a very low range, while that of enclosures (1.16%) and woodland (1.31%) fall within the medium range according to the rating by EthioSIS (2014).

We found a relatively higher TN content and stock in the protected woodland than in the communal grazingland, enclosures and farmland. This could be due to the presence of leguminous woody trees that might have the capacity to fix atmospheric nitrogen resulting in increased TN in the soil system.

Similarly, other studies from southern-central and rift valley highland Ethiopia (Lemenih et al., 2005; Yimer et al., 2007) reportedconversion of a native forestland to cropland that contributed to dramatic decline in SOC and TN. However,a relatively higher TN content and stock under the communal grazingland as compared to enclosures and cropland land might be due to the addition of N through urine and feces under continues grazing and/or browsing by livestock.

A similar study from southern-central Ethiopia (Demessie et al., 2013) aslo found more SOC and TN stocks in naturally managed vegetation as compared to that of the converted lands to agricultural activities. Another study by Zhang et al. (2013) in the Loess Plateau of China reported that grassland and forest has generally the highest SOC and TN storages, whereas the SOC and TN storages are the lowest in croplands.

The SOC stocks recorded in the cultivated land (38.1 Mg $ha^{-1}$) and woodland (55.90 Mg $ha^{-1}$) are generally in agreement with the findings of Alam et al. (2013), indicating that the SOC for tropical woodland and savanna ecosystems are in the range of 20 to 80 Mg $ha^{-1}$.

In contrast, Dabasso et al. (2014) have reported higher value of SOC stocks (ranging from 78.93 to 107.22 Mg ha$^{-1}$) within the soil depth of 0-30 in different vegetation types of semi-arid regions of Kenya.

Others (Grace et al., 2006; Mureithi et al., 2014) have also stated that soil organic carbon in response to land management practices are site specific. However, the same authors stated that the interaction of natural and anthropogenic factors with historical land use change particularly in East African savanna are rare and needs an in-depth study.

In general, the results of the present study indicate that soils under the natural vegetation of rangelands are threatened by the permanent human interference such as overgrazing and expansion of crop farming, whereas restoration of degraded rangelands and protection of natural forest (woodland) can also enhance carbon sequestration.Lal (2015) has stated that one of the most straight forward pathways to mitigate soil degradation in arid and semi-arid drylands is to maintain or replenish SOC concentrations above the critical level of 1.1 to 1.5%.

Although the the C:N ratio did not vary among the land use types, it was found to be higher in enclosures as compared to the remaining land use types, with more apparent varation in the top layer of the soil (0-15 cm). In linewith this finding, Nega and Heluf (2013) have also indicated that C: N ratio did not vary in terms of land use types rather the ratio of C: N varied between soil depth

However, our results on C:N ratio values across the four different land use types at both soil depths were found to in the low range when compred to the optimum range of the C:N ratio, which is expected to be about 10:1 to 12: $10:1$ in mineral soils (Landon, 1991). Carbon-nitrogen (C:N) ratio is an index of nutrient mineralization and immobilization whereby low C:N ratio indicates higher rate of mineralization (Brady and. Weil, 2002).

### *3.4.4. Changes in SOC and TN stocks due to land use changes*

A basic assumption of this study is that the dynamics of SOC and TN stocks is triggered by the impact of land use/land cover changes. Accordingly, the present findings indicate that the conversion of woodland to cultivated land resulted in losses of SOC stock [17.8 Mg ha$^{-}$

$^{1}$ (31.84%)] and TN stock [1.93 Mg ha$^{-1}$(25.66%)] in the 0-30 cm depth. Similarly, we recorded a loss of SOC stock [17 (24%)] and TN stock [8.0 Mg ha$^{-1}$ (13%)] as a consequence of the conversion of communal grazingland and enclosures to cultivated land.

Our findings are in agreement with previous reports (e.g. Post and Kwon, 2000; Guo and Gifford, 2002; Murty et al., 2002), indicating that the conversion of native vegetation to cropland causes a reduction in SOC by 25–42% in the top layer. SOC can be rapidly lost due to enhanced carbon decomposition and erosion that brought about by soil disturbance (Lal, 2005).

Furthermore, the conversion of communal rangelands to cultivated land on average contributed to the reduction of SOC stock (20.5%) and TN stock (10.5%). A similar study in southern Ethiopia by Demessie et al. (2013) indicates that the conversion of natural forest to cultivated land has adversely affected the status and dynamics of SOC and TN contents and stocks.

On the other hand, the conversion of communal grazingland to protected woodland and enclosures improved SOC stock (16-22%) when compared to SOC stock under the communal grazingland. The probably reason for the improvement in SOC stock might be due to the restoration of degraded rangelands through vegetation management.

In consistent with the current findings Guo and Gifford (2002) have also stated that SOC stocks is increased (+19%) following the conversion of cropland to pastureland. Girmay et al. (2008) also confirmed that carbon sequestration is increased in the soil system as a result of the restoration of degraded lands. This suggests that the dynamics of land use change can influence the status and dynamics soil carbon and nitrogen.

Overall, the current findings indicate that land use conversions triggered a loss of SOC stocks[2] [319.51 Gg (0.10 Mg ha$^{-1}$yr$^{-1}$)] and gain of SOC stocks [59.63 Gg (0.02 Mg ha$^{-1}$yr$^{-1}$)] with net loss of 299.88 Gg (73.4 x 10$^{-3}$ Mg ha$^{-1}$yr-$^{1}$) over the last 37 years.

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[2] SOC and TN accumulation (Mg ha$^{-1}$ yr$^{-1}$) rate was estimated based on time period of 37 years (1976-2013) and divding by the total area of the four land use types under gone transition dervided from LULC analysis.

This estimation indicates a significant loss in terms of SOC stocks from woodland (31.4%) and enclosures (29.8%) out of the total SOC stocks recorded in the study sites. Deng et al. (2016) have also indicated an average loss of 0.39 Mg $ha^{-1}$ $yr^{-1}$ inSOC stock as a result of land use conversion.

Others (Houghton et al., 2012; Munoz-Rojas et al., 2015) also indicate that land use/land cover changes have a significant effect on the dynamics SOC stock accumulation. Similarly, Bolin and Sukumar, (2000) have stated that change in land use can cause alteration in the amount of carbon stocks. However, Guo and Gifford (2002) argued that conversion of one land use type to other form of land use might result in either as carbon source (loss) or carbon sink (gain).

In the context of the current study, figures in this paper would be useful insights on the dynamics of carbon caused by land use/land cover changes in the study areas. However, because of limitations of data, SOC and TN stocks were hypothesized as constant regardless of the changes expected over the last 37 years. Thus, the changes in SOC and TN stocks due to land use/land cover changes in the present study might be better referred to as potential changes.

The loss in TN stock was attributed to the expansion of cultivated land through the conversion of communal grazingland, which is estimated at 13.69 Gg (54.8%), followed by woodland (7.11 Gg or 25.1 %), whereas the gain (increase) was observed from the conversion of cropland and enclosures to other land use types. For example, others (e.g., Houghton et al., 2012; Deng et al., 2016) showed hat the lack of long-term monitoring of the magnitude and direction of SOC stock dynamics might make it difficult to come up with solid conclusion as the dynamics of SOC stock could be influenced by land use types, management and other disturbances.

### 3.4.5. CONCLUSIONS

The results of the present study showed the dynamics of land use changes in southern Ethiopia over the last 37 years. The findings highlighted that an increase in the coverage of cropland, bushland and bareland was an emerging issue at the expense of communal grazingland. Conversely, woodland and communal grazingland showed a constant decline in terms of area coverage over the last 37 years. Generally, differences in land use types greatly influenced SOC and TN stocks.

The results showed a decrease in SOC content and stocks in woodland, enclosures, open-grazed rangelands (communal rangeland) and cultivated land. Conversion of woodland and communal grazingland to cultivated lands resulted in higher loss of SOC and TN stocks. Overall, conversions of communal grazingland and cultivated land to woodland and enclosures partially balance SOC and TN stocks losses in the soil system of the study area.

We showed that restoration of degraded rangelands and protection of natural woodlands can improve soil quality and enhance carbon sequestration in savanna grassland system. However, long-term carbon and nitrogen data are required to better understood gains and losses in SOC and TN stocks as a result of land use changes.

Generally, the current findings are useful to inform policies makers and land owners in the implementation of limproved management practices in savanna rangelands of southern Ethiopia. We suggest further study understand the carbon dynamics for climate change mitigating.

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# CHAPTER FOUR

# 4. ALLOMETRIC EQUATIONS FOR PREDICTING ABOVE-GROUND BIOMASS OF SELECTED WOODY SPECIES TO ESTIMATE CARBON IN BORANA RANGELANDS, SOUTHERN ETHIOPIA

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[a]Hawassa University, College of Agriculture, P.O.Box 05, Hawassa, Ethiopia

[b]Hawassa University, School of Veterinary Medicine, P.O.Box 05, Hawassa, Ethiopia

[c]International Livestock Research Institute (ILRI), P.O. Box 30709, 00100 Nairobi, Kenya

[d]World Agroforestry Centre (ICRAF), P.O. Box 30677, 00100 Nairobi, Kenya

*Corresponding author: e-mail: ayana.angassa@gmail.com

## Abstract

*Species-specific allometric relationships which relate the biomass of individual tree components to easily obtainable measurements are essential for comparative assessments of standing biomass and biomass allocation strategies. We developed species specific equations to predict aboveground biomass (AGB) of ten woody species in Borana rangelands of southern Ethiopia. Destructive harvesting was employed to obtain dry weights of aboveground biomass by components, which were related to the diameter at stump height (DSH), diameter at breast height (DBH), tree height (TH), and crowndimentsions [(crown area (CA) and crown volume (CV)].The predictive performance of these five different predictors were assessed by introducing aloneand or in combination wof two or more of them using adjusted coefficient of determination (Adj.*$R^2$*), Akaike information criterion (AIC) and mean square error (MSE). Models used linear natural log transformation of the common powre form of allometric equation.Top ranked models were those using combination of DSH (stem diameter at 30 cm above the ground) with crown volume (CV) showed the best fit as demonstrated their the highest coefficient of determination and higgly significant (adj.*$R^2 > 0.80$*;* $P < 0.001$*), except in three woody species (A.brevispica, Rhus natalensisi and C. Africana). Furthermore, the best fit models also varied within biomass components of the same species and between woody species. Likewise, the mixed-species allometric equations for a group of woody species was best fit by a combination of three predicators (DSH-TH-CA), with adj.*$R^2$*ranged between 0.84 and 0.90 across the AGB compoentents. Hence, our species-specific allometric equations could be adopted for the indirect biomass estimation in semi-arid savanna ecosystem of southern Ethiopia. The mixed species allometric models will also give a good opportunity when species-specific equations are not available and contribute to estimation of the biomass and carbon stock in woody vegetations of East African rangelands.***

**Keywords**: Acacia species, Biomass Borana; Carbon stock; Regression models

## 4.1. INTRODUCTION

The invasion of woody plant species in arid and semi-arid rangelands is a common problem worldwide (van Auken 2009; Archer and Predick, 2014).

Savanna rangelands have become increasingly encroached by woody plant species in recent decades, which have been particularly more evident in African savannas (Oldeland et al., 2010; Kgosikoma et al., 2012). These invasions of bush encroachment into savanna rangelands have been triggered by environmental and anthropogenic factors (Angassa and Oba, 2007; Coetzee et al., 2008; Knapp et al., 2008).

Furhtermore, increases in the global carbon dioxide ($CO_2$) concentration may further benefit the establishment of $C_3$ woody species at the expense of $C_4$ grasses in arid and semi-arid ecosystems (Wigleyet al., 2010; Throop et al., 2012).

In many parts of East African rangelands, large numbers of invasive woody plants have been identified (Coppock, 1994; Young et al., 1998; Oba et al., 2000; Brooks et al., 2004).The thickening of woody vegetation that competes with the herbaceous forage has negatively affected the grazing capacity of many rangelands (Scholes and Archer, 1997; Solomon et al., 2007).

Until the 1970s, the Borana rangeland management system in southern Ethiopia was considered as one of the finest grazing lands among East African rangelands (Coppock, 1994; Oba, 1998).By the mid-1980s, about 40 percent of the Borana rangelands had already been affected by bush encroachment (Coppock, 1994), which is approximately equal to a density of 2400 plants per hectare, is considered as a borderline between the encroached and non-encroachedcondition (Roqueset al. 2001).

Hence, previous study (Dalleet al., 2006) reported thatthe cover and density of woody plants have crossed the threshold level to 52 percent with mean total woody plant density of 3014 stems per hectare. Similarly, Angassa and Oba (2008) have also reported a total density of 3995 woody plant stems per ha in the heavily encroached rangelands of Borana in southern Ethiopia.

The spread of woody vegetation cover in the Borana rangelands has attributed to numerous factors, among others, suppression of fire (Oba et al., 2000), increased grazing pressure (Coppock, 1994), exclusion of browsers (Angassa and Oba, 2010), establishment of permanent ponds in the wet season grazing rangelands and expansion of settlements (Oba, 1998) and episodic climatic events (Angassa and Oba, 2007) have played a major role in the proliferation of invasive woody species.

This in turn threatens thelivestock production particularly grazers and livelihood of pastoral communities (Oba and Kotile, 2001; Angassa and Oba, 2008). In contrast, the invasion of woody vegetation in savanna rangelands is generally thought to contribute to the increase in carbon (C) sequestration potential in the systems (Hughes et al., 2006;González-Roglich et al., 2014).

In savannas, above-ground carbon (C) stocks increases as the proportion of trees increase relative to grasses and can be associated with positive attributes in multiple ecosystems functions such as reduced nutrient loses from soil erosion processes and enhances nutrient recycling (Howard et al., 2012).

Hence, woody plant encroachment in savannas has the potential to alter carbon and nitrogen sequestration stocks over the long-term, which have regional or global environmental implications (Archer et al.,2004).This carbon accumulation appears to be a function of enhanced below and above-ground net woody vegetation primary productivity (NPP) and organic matter stabilization in protected soil aggregates (Knapp et al, 2008).

In some countires, for Australian (Gifford and Howden, 2001) and tropical American savannas (Asner al., 2003; Lettet al., 2004) the net gain in carbon sequestration through woody plant encroachment were estimated.

However, with increased density of woody plant encroachment in many parts of Sub-Saharan Africa, either species specific or generalized allometric equations for predicting the above-ground biomass are rarely available that could be used to estimate carbon sequestration are rarely reported (Henry et al., 2011).

Furthermore, little information is available regarding the above-ground biomass and carbon stock potential of woody species growing in arid and semi-arid ecosystem of southern Ethiopia.Thus, to quantify the net gain in C stock due to woody plant proliferation in such ecosystems reliable estimates of the standing biomass are required.The above-ground woody biomass can be quantified by destructive harvest (direct method).

This method involves harvesting of all the trees in the known area and measuring the weight of the different components of the harvested tree like the tree trunk, leaves and branches (Gibbs et al., 2007) that can be costly and impractical, especially when dealing with numerous species and large sample areas (Xiao and Ceulemans, 2004).

The second method of tree biomassestimationisthenon-destructive methodor an allometric equation (indirect method) that is ultimately based on harvested sample trees (Montèset al., 2000; Chaveet al., 2005) relate to measurable variables (e.g., diameter at breast height, total tree height, and woody density).

Hence, the use of allometric prediction of biomass compartments is often the preferred approach since it is less time consuming and less expensive than direct measurements (Vashum and Jayakumar, 2012). However, the models have been found to vary from species to species, and for a given species in different ecological regions of the world (Litton and Kauffman, 2008; Yen and Lee, 2011; Alvarez et al., 2012; Vahedi et al., 2014).

Therefore, allometric equations developed for East Africa are exclusively focused on forestry systems (Kuyah et al., 2012) and are species-specific (Okelloet al., 2001). Furthermore, developers of these models often caution against extrapolation beyond their study sites.

This might be due to the spatial heterogeneity of landscapes and variability of environmental factors (i.e., soil type, soil nutrients, climate, disturbance regime, succession status and topographic position) and genetic variation among woody species (Cole and Ewel, 2006; Litton and Kauffman, 2008). It has been postulated that generic equations may lead to systematic errors of up to 400 percent at the site level.

Therefore, locally developed and well-implemented models may be a better alternative and are expected to provide less uncertainty than generic equations (Chaveet al., 2014). In arid and semi-arid ecosystem of southern Ethiopia, either species specific or generalized allometric equations to estimate the above-ground biomass and carbon sequestered in the woody vegetation are rarely available except a study by Hasen-Yusuf et al (2013).

However, that study has only addressed limited number of woody species at specific site, where its direct application would result in higher uncertainties in heterogeneous landscapes with varied vegetation types across the Borana plateau.

Therefore, the aims of this study were to: (1) develop species-specific allometric models to predict the above-ground biomass for ten key woody species common to the savannas of southern Ethiopia; (2) develop a generalized allometric equations for a group of woody species that can contribute to estimate the biomass and amount carbon that can be sequestered in woody vegetation of the savanna rangelands of East Africa.

We hypothesized that: (i) the biomass prediction models would vary depending on the growth form and architecture of the woody species; (ii) For some of the woody species, use of the combinations of two or more dendrometric variables will improve the model fit, while it could also make the model complex and sources of bias in some woody specie

## 4.2. MATERIALS AND METHODS

### 4.2.1. Study area

The study was conducted in Borana lowlands of southern Ethiopia between July and August 2013 (locally called cool dry season) when plants reach at their peak growth and with full leaves to capture the crown architecture (branching patterns). This particular study was carried out in Dida-Hara (04º47.318'N and 038º20.017'E), which is located at about 30 km east of Yabello town.

The oral history showed that about four decades ago, Dida-Hara area was known for its open savanna grassland with very few or no tree covers. The name Dida-Hara derived from the open grassland plain with no tree. Dida-Hara was used for wet season grazing while livestock used to move to the well zone during the dry season (Angassa and Oba, 2008). The rests of general description of the study area including climatic factors (e.g., rainfall), soil and vegetation can be referred in details from the section of general materials and methods under **chapter I**.

### 4.2.2. Selection of woody species

The woody plants species were sampled from within enclosures and open-grazed rangelands in Dida-Hara area. Within this study area, measurement sites were established within 10 x 10 km radius. Initially, a reconnaissance survey was conducted and contacts with the local leaders and expert working in the area.

Both vegetation and socioeconomic data were collected to capture important information in order to address the specific objectives of the study. Within the extent of this area, we identified and ranked about 15 dominant woody tree species.

However, since local administrators and communities strongly control cutting of woody trees, we limited the number ofsampled trees only to 10 dominant species include: *A. brevispica, A. nilotica, A. drepanolobium. A. etbaica, A. tortilis, A. bussei, A. seyal, Rhus natalensis, Lannea rivae, Commiphora africana*.

The selected sampled woody species were found to be most relevant for our research as they accounted for about 60 % of the vegetation coverage in the study area and across the Borana rangelands. They also included both non-invasivespecies (*A. tortilis, Lannea rivae, A. nilotica*) and invasive species (*A. brevispica, A. drepanolobium, A. bussei, A. seyal, A. etbaica and Commiphora africana*).*A.drepanolobium*was considered as serious encroacher adversely affecting the productivity of the Borana rangelands (Dalle et al., 2006).

These woody species grow intermixed, although specific stands of some of these species alsooccurred in patches (*pers. obs.* Kenea), and with a wide range of services mainly for construction purposes, used browse plant as alternative feed sources and supplementary feeding for coping up feed shortages in both dry and wet season, edible fruit, fuel and charcoal making and as environmental indictors (Dalle et al., 2006; Terefe et a., 2011)

### 4.2.3. Field data collection and dry weight estimation

The allometric equations were developed based on 10 species (15 for each species x 10= 150 sampletrees) were selected through systematic random sampling to ensure the inclusion of wider possible  spanning a size range representative of the dominant size classes of height and diameter classes growing the study area.

The destructive sampling of the species was carried out within similar topographic and soil conditions both from within the enclosures and adjacent open-grazed sites as some are abundant inside the enclosures (e.g., *A. tortilis, A. etbaica, Lannea rivae and A.seyal*) and others are predominantly found in the open-grazed and poorly drained soil of valley bottoms (e.g., *A. drepanolobium and A. bussei*).

Measurements and distructive harvesting took place during the short dry months of July to August 2013, when the woody species are at their full leaves to capture the crown architecture (branching patterns). Furthermore, each plant was checked for any signs of previous damage by herbivore, humans and or any irregularity (i.e., hollow stem, broken branches) prior destructive harvesting.

We recorded local names and corresponding scientific names were obtained with the aid of knowledgeable elder and expert working in the areas, and available literature. Measurements collected prior to harvesting includeddiameter at stump height (DSH at 30 cm), diameter at breast height (DBH at 1.30 m), and tree height and crown dimensions.

The diameter at stump height was included due to the fact that many woody species in the area have multi stemming patterns especially beyond 30 cm from the ground surface. We also included crown widths as the crown shape and growth structure varied among the woody species.

We used diameter tape (d-tape) to measure DSH and DBH (in cm), with the tape held horizontally and tightly at tree stem. Tree height (TH , m) from the base of the main stem to the top of leading leafy shoot was measured using 5 m long metal stick marked at 50 cm intervals and any leftover length at the base of the tree was measured using measuring tape.

The crown edges of woody plant were first located visually.Then, the two crown diameters (taken in two perpendicular directions, i.e., north–south and east–west) with the longest axis at its widest point (d1) and the one perpendicular to the longest axis (d2) at the same height were measured using measuring tape with the help of two persons standing at each margin of the canopy.

Then, the crown area (CA) and canopy volume (CV) inm werecalculated using the formula of an ellipse (Dietz and Kuyah, 2011):

$$\mathrm{CA} = \Pi \times (d1/2) \times (d2/2) \qquad (4.1)$$

and

$$\mathrm{CV} = 2/3 \times \Pi \times (d1/2) \times (d2/2) \times \mathrm{TH} \qquad (4.2)$$

Where, TH is total tree height that extends from the base of the plant to the tallest photosynthetically active material and d1 and d2 are the diameter readings taken at 50% of plant height with d2 perpendicular to d1 (Thorne et al., 2002).

After dendrometric measurements, trees were cut down at ground level using an ax and separated into bole (main stem ) and branches > 2 cm in diameter), branches (diameter < 2 cm), dead wood (attached dead wood), and leaves.The stem and larger branches were cut into smaller pieces to facilitate weighing in the field and twigs and leaves were collected into separate bundles.

Branches, live and dead thin branches including leafy shoots, both of which hereafter referred as ‘‘branches’’ and stem to achieve three main response variables: ‘‘branches’’ (including all leafy shoots), ‘‘stem’’ and ‘‘total’’ (including both branches and stems).

Stem was considered as the main upright portion of the plant below the first branch and the branches were considered all woody portions of the canopy above the stem (N’avar et al., 2004). The fresh weights of the smaller stem and branch sections, as well as that of the leaves, were determined in the field on a 300-kg capacity of a hanging field-scale (precision 200 g, Germany).

The collected sub-samples of (250–300 g based on total fresh weight of each woody tree) for each stem, branch and twigs (in paper bag) to the laboratory of the Yabello Research Center of Oromia Agricultural Research Institute.

In the laboratory, the samples were oven dried for about 48 h at 105°C until constant weight was achieved and weighted to the nearest 0.00 gm to establish any change in weight, after which the dry weight was recorded. Component dry weights were calculated using respective subsample dry: fresh weight ratios and the component fresh weight field measure.

### 4.2.5. Model selection for allometric equations

Our dataset contained basic dendrometric measures of trunk diameter, DBH (1.30 m), and DSH (3o cm) and both of which have been routinely been used as predictor variables for biomass, often in combination with TH and crown dimentions (CA and CV) for use in model selection based on preliminary regression analyses with component biomass, which suggested that DSH was the most reliable diameter measure for dry biomass estimation for small size and multistems of acacias species.

To investigate the relationship between above-ground biomass and the predictor variables (i.e., DSH, DBH, TH, CA and CV) we restricted our analysis to the most common allometric forms, the power equation (Chave et al., 2005; Kuyah et al., 2012; Mascaro et al., 2011)

$$y = ax^b \qquad (4.3)$$

Where y is the component mass, x is predictor variable, and a and b are regression parameters tobe estimated from regression analysis and the log transformed linear form

$$\ln(y) = \ln(a) + b\ \ln(x) \qquad (4.4)$$

Assumptions related to homogeneity of variance, normal probability plots of residuals, and plots of standard residuals versus predicted values to test compliance with the assumptions of least-squares regression using the original (untransformed power function) and ln-transformed data (linear forn).

The power equation, or power function, is commonly used in allometric equations as it best models the heteroscedasticity (unequal error variance) which is common in allometric data.The ln-transformed form of this power function, was used to stabilise varianceas determined by Breusch–Pagan and Shapiro Wilks ($p < 0.05$). All parameters examined showed a better fit tin the ln-transformed linear model as compared to the power function.

The ln-transformed form of this power function, $\ln(y) = \ln(a) + b\ln(x)$, was decided to apply by logarithmically transformed on both sides of biomass equations (Eq.4.4) to stabilise variance and follows the logarithmic form of the simple linear equation for the fitting model and for dealing with heteroscedasticity (Overman et al., 1994)].

Similarly, Hasen-Yusuf et al. (2013) also stated a better fit of the natural log linear model as compared to the non-linear (i.e., power function) in predicting above-ground biomass from different predicator variables.

### 4.2.6. Model selection criteria for goodness of fit

We selected commonly used model formations as candidate models for evaluation, with predictor variables ranging from each of DSH, DBH, TH, CA and CV) to a combination of two or more of these predicators stepwise into the models.

This technique was used to determine whether the incorporation of two or more predicator variables improved the model performance to estimate the total AGB and components (stem, branches and total above-ground biomass) on the basis of independent predicator variables

About seventeen (17) model formations as candidate were screened across the three biomass compoentns and 3 to 4 modles per each biomass compentn were compared to choose the best fited ones.

We included all dendrometric measurements in order to compensate for highly variable architecture and branching pattern of each species, which often makes biomass prediction problematic (Cole and Ewel, 2006) .We, also developed multi-species allometric equations by using the pooled data of selected species with best fitted in their respective species specific model fittings.

The goodness of fit was determined by examining the adjusted coefficient of determination (Adj.$R^2$; higher value is better), Akaike's information criterion (AIC) and mean square error (MSE) (lower values are better). AIC was measured by the equation below:

$$\text{AIC} = 2k - 2\ln(L) \tag{4.5}$$

(Chave et al., 2005) where, k is the number of parameters in the model, and L is the maximized value of the likelihood function for the model. We compared the AIC differences between the candidate models and the chosen model with the smallest and the models performed best AIC $\Delta$AIC = 0, indicating a minimal loss of information, i.e. the model with the best statistical fit, whereas the model prediction might not be valid when $\Delta$AIC >10 (Kennth and David, 2002).

We also reported the prediction error (PRESS residuals) as an alternative statistic reflecting the accuracy of the estimator difference between the observed and predicted (small difference is better).The assumptions of multicollinearity was tested using a variance inflation factor (VIF) and value greater than 10 (VIF>10) is an indication of potential multicollinearity among in-dependent variables (Bihamta and Chahouki, 2011).

Accordingly, we systematically omitted the predictor variable that showed a strong correlation coefficient (r ≥0.90) with another predictor, while being less strongly associated with the biomass in our model development. Further, we excluded predicator variables with the value of VIF>10 from the model.

Thus, DSH was strongly correlated with DBH, and CA was strongly correlated with CV for all species and carry was taken when the predicators used together in the same model.

Finally, the correction factor for the systematic bias introduced by this natural log (ln) transformed data was computed to be applied where back transformation to power biomass function is required and calculated as (Baskerville, 1972; Mascaro et al., 2011)

$$CF = \exp^{(MSE/2)} \qquad (4.6)$$

Where, CF is a correction factor for back transformation of the value f biomass estimated based on long transformed (natural) to normal, MSE is the mean square error obtained by the least-squares regression for each species and biomass components. All statistical analyses were carried out using PROC REG in SAS 9.3® (SAS, 2012).

## 4.3. RESULTS

### 4.3.1Data used in development of allometric equations

The average above-ground dry biomass data on stem, branches and total biomass and field level dendrometric measurements for 10 dominant woody plant species (15 samples of each species (n=150)are represented in Appendix 4.1.The moisture content of species was ranging between 22.28 % for *Acacia brevispica* to 66.48 % for *Commiphora africana*. For each tree, we used the sampled dry weight data to calculate the total dry weight of each fraction of biomass.

Thus, a total of 150 woody plants (15 of each woody species) from a range of size and diameter classes were harvested for the development of allometric equations for estimating the above-ground biomass components for each species The mean diameter at the stump height of each species harvested surpassed 4 cm and the largest value was 10.54±1.20 cm for *A. tortilis* followed by *A. nilotica (*9.63±0.76 cm) (see Appendix 4.1).

### 4.3.2. Specific allometric equations

The goodness of fit models developed for ten key woody plant species, corresponding parameters, correction factors and statistical descriptors are presented in Appendix 4.2.The goodness of fit analysis for the models across allspecies indicated that the species-specific regressionmodels relating biomass with selected predictor variables measured in the field

Thus, for these data the parameterestimates for the intercept and slope together with the overall fit of the models are significantly different from zero ($P<0.001$) except for threespecies (i.e., *C.Africana, A.brespsica and Rhusnatalensis*). However, the number and type of predictors, i.e., the woody species dendrometric measurements required by the equations to achieve the best fit differed across species (Appendix 4.2).

Among the singlepredicator variable, the model with DSH was the bestpredictor of AGB for *A. drepanolobium, A. seyal and A.nilotica* (branches biomass) (adj.$R^2$>0.90 and thevalues of MSE<0.15,lower delta AICc values<10), while model with DBH alone was best fitted inpredicting the total AGB and branches biomass for A.*tortilis and A. bussei* (adj.$R^2$>0.82).

On the other hand, TH wasbest fitted in predicting the stem biomass (adj.$R^2$ = 0.93, delta AICc = 5.62) for A.tortilis.Crown area (CA) also gave relatively best fit in predicting the branch biomass (Adj.$R^2$ = 0.77, deltaAICc value = 5.85) for *A.bussei* (Appendix 4.2)

On the other hand, crown volume (CV) alone showed best fit for *A. bussei and A. drepanolobium*in total and stem biomass components with goodness of fit statistics (Adj.$R^2$ that ranged between 0.79 and 0.90, MSE<0.30 and delta AICc values ranged between 4.65 and 8.63). Our findings also showed that crown area as single predicator considered least as best fitted in a few species and biomass components (Appendix 4.2).

A combination of DSH and CV selected as best performing models in predicting AGB for *A. seyal, A. bussei, A. drepanolobium and A. etbaica and Lannea rivae*. Improvements in the performance of the models in terms of increase in adj. $R^2$, and decrease the values of AIC and MSE as compared to models with individual predicator.

Likewise, the combination of DBH and TH showed superior performance fitted in all biomass components for A. tortilis and *A. nilotca* with increase adj. $R^2$, and decrease the values of AIC and MSE as compared to when used alone in model (Appendix 4.2).

In general, best-performing models based on single predicator variable and combination of two or more predicators have comparable low delta AICc values (less than 10), similar coefficient of determination values (adj.$R^2$) and comparable MSE estimates. For models that performed equally, the most parsimonious model (simplest form of the predictor variable combination) was selected as the best-performing model.

Further, the prediction error sums of squares (PRESS) was used as validation criterion in determination of the final best model fitted to the biomass equations (Appendix 4.2). Our findings showed that species were explained by models requiring a combination of two predictor variables to obtain a good fit for sound estimates of the AGB (adj.$R^2$>0.80), whereas others species (i.*e., Acacia brevispica, Rhus natalensis and Commiphora africana*) showed poor fit and even no fit possible (Appendix 4.2).

Moreover, model fits were generally better for total AGB and branch biomass as compared to stem biomass (Appendix 4.2).However, ours results that fit the totaland component biomass versus prediction modelsshowed variations with the type of woody species and component biomass as some models were best predicted based on single predictor, and others required two more combination of these predicator variables to achieve the best goodness of fit that supported our hypothesis.

Thus, in-depth descriptions of each species-specific allometric biomass models for each species are as follows:

*Acacia bussei:* A combination of DBH and CV selected as the final best fitted to total AGB and stem biomass equations, because the model has lower values MSE (32–63 % decrease) and higher adj. $R^2$ (4–11 % increase) and lower PRESS value as compared to the single use of DBH and CV, while a combination of DBH and CA chosen as the final best fitted to the branch biomassequation (Table 4.1),with higher adjusted $R^2$ (7–9 % increase) and lower value of MSE (30–36 % decrease) as compared when used in single form (Appendix 4.2).

In addition, total dry AGB regressed against each dendrometric measurement, best explained except height (TH) showed the poor relation (adj. $R^2$ of 0.47 as compared toDSH (adj.$R^2$ = 0.91) DBH (adj.$R^2$ = 0.87), CA (adj.$_R{}^2$ = 0.86) and CV (adj.$R^2$ = 0.90) (Fig. 4.1).

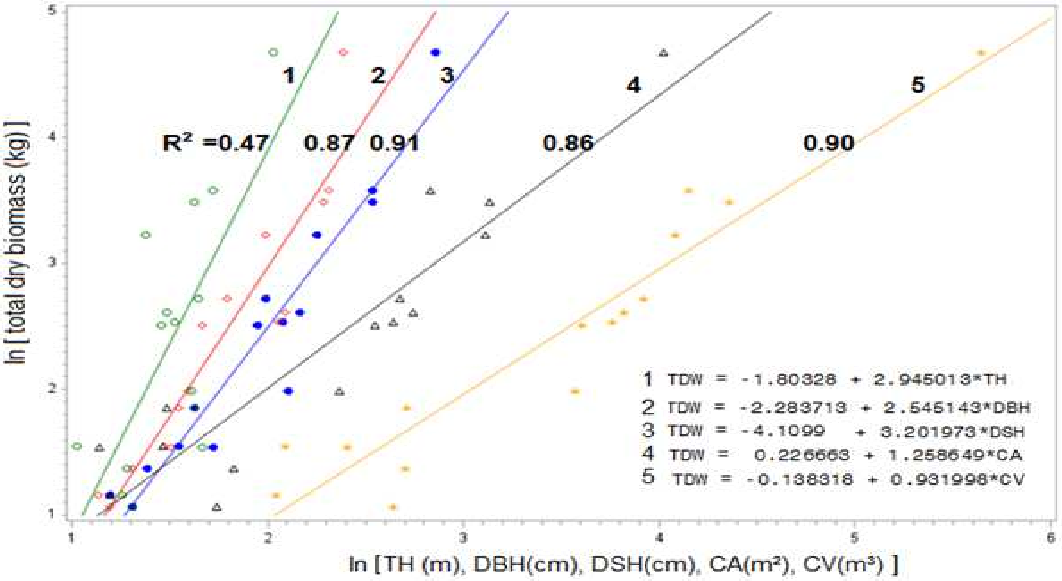


Figure 4.1. Regression lines (overlay) showing total dry biomass versus height (TH:1), diameter at breast height (DBH: 2) diameter at stump height (DSH: 3), crown area (4) and crown volume (5) for *Acacia bussei.*

*Acacia drepanolobium:* The combined use of DSH and CV was chosen as the best of fit to total AGB and stem biomass (Table 4.1), with better goodness statistics such as lower value in MSE (38–73 % decrease), delta AICc (6.4–18.6 decrease) and higher adj. $R^2$ (2–8 % increase) as compared to the single use.

For the branch biomass component, a combination of DSH and CA improved the model fit with slightly decreased delta AICc (2.14–19.75) and MSE values (0.02–0.22) and increased dj.$R^2$ (1–13%) as compared when used in single form (Appendix 4.2). Furthermore, as it was observed from regression (overlay) graph in Figure 4.2, there was strong relationship between all predicators versus the total AGB as explained best with adj.$R^2 > 0.80$.

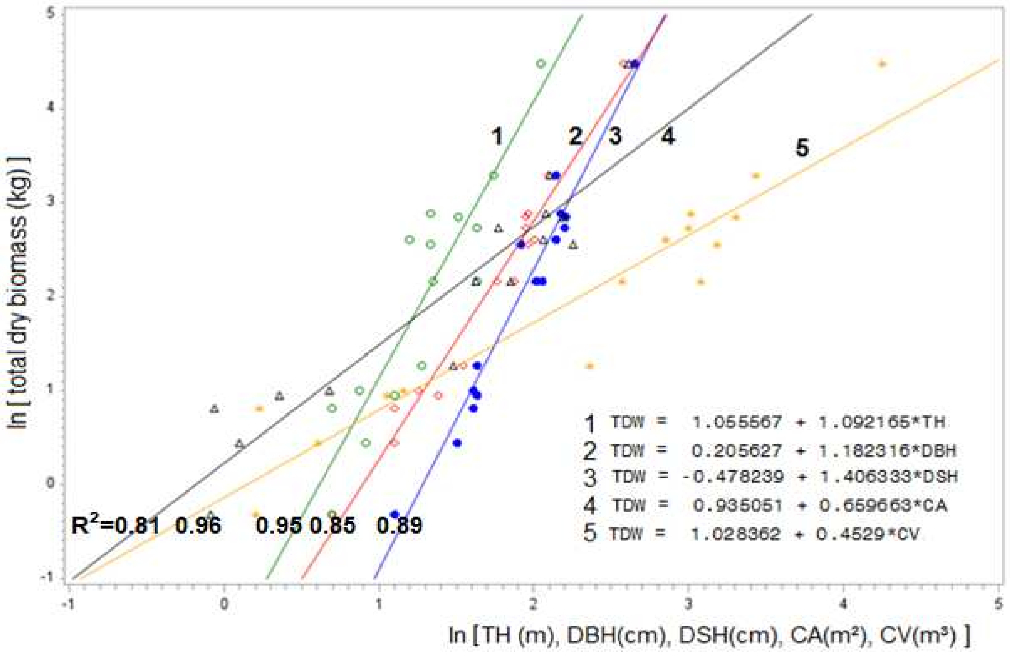


Figure 4.2. Regression lines (overlay) showing total dry biomass versus height (1), diameter at breast height (DBH: 2) diameter at stump height (DSH: 3), crown area (4) and crown volume (5) for *Acacia drepanolobium*

*Acacia etbaica:* For this species, the chosen final best biomass equation is with combination of DBH and CV that had decreased the delta AICc (e.g. by 13.32) and MSE by 50 % and increased adj. $R^2$ ranging from 0.75 to 0.84 for total AGB, while branch biomass was best estimated by a combination of DBH and TH with adj.$R^2$ of 0.68 chosen as final model equation (Table 4.1)

However, the total variation in total AGB explained byindividualpredicators was generally weak as it was revealed graphically, the minimum for TH (adj.$R^2$ = 0.39) and the maximum for CV (adj.$R^2$ = 0.67) (Fig.4.3). Hence, incorporation of the crown dimensions into the model significantly improved the predictive power and accuracy of the model.

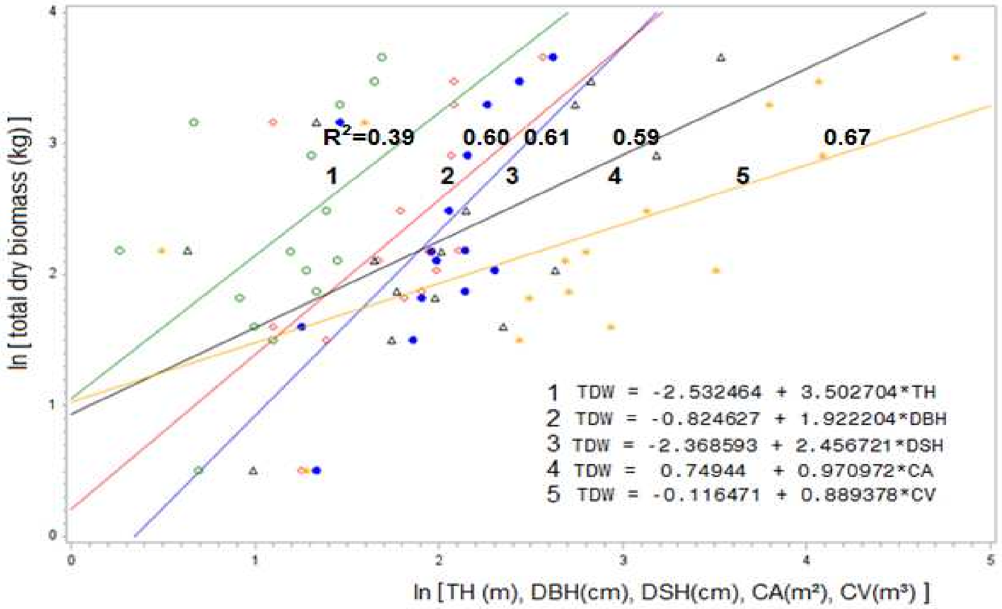


Figure 4.3. Regression lines (overlay) showing total dry biomass versus height (1), diameter at breast height (DBH: 2) diameter at stump height (DSH: 3) crown area (4) and crown volume (5) for *Acacia etbaica*

*Acacia nilotica:* As shown in Appendix 4.1, the model that uses a combination of DBH and CV (for total AGB) and DSH and CV (for branch biomass) had brought little improvement in terms of model performance criteria over the use DSH alone, whereas for stem biomass, the inclusion of TH to model with DSH and CV improved the adj.$R^2$that ranged between 0.18 and 0.28 and decreased the value of MSE by 0.09 when compared with that of the model fit using DSH and TH alone (Table 4.1).

The regression of total AGB against individual predicators showed that both diameters have strong correlation with the total above-ground biomass (DBH, adj.$R^2$ = 0.94) and (DSH, adj.$R^2$ = 0.92) (Fig. 4.4).

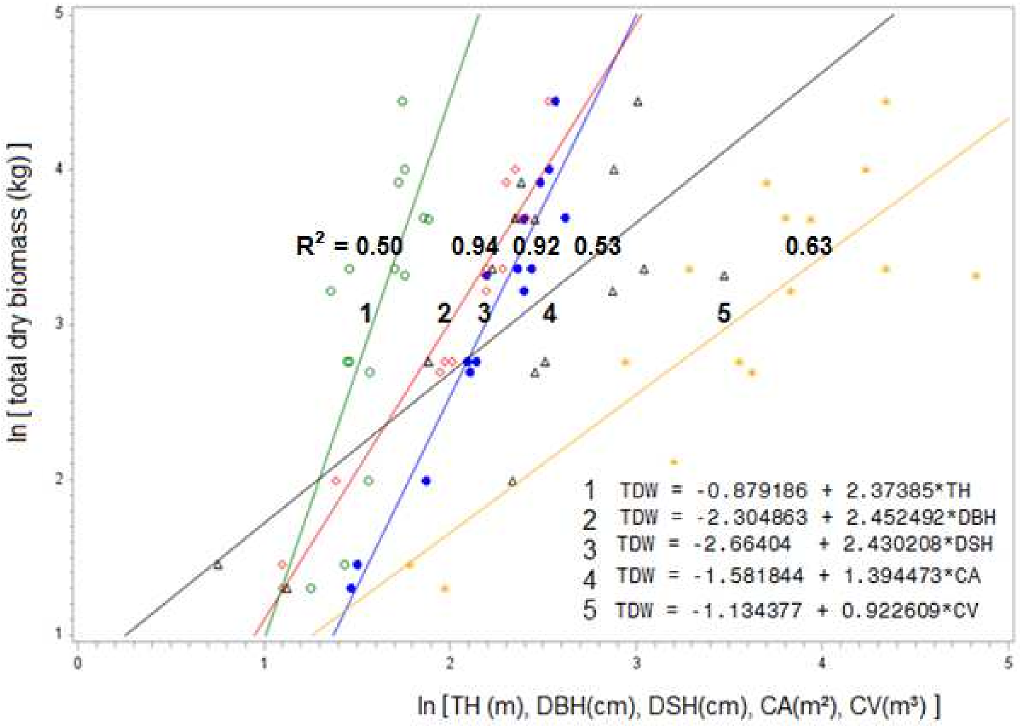


Figure 4.4. Regression lines (overlay) showing total dry biomass versus height (1), diameter at breast height (DBH: 2) diameter at stump height (DSH: 3), crown area (4) and crown volume (5) for *Acacia nilotica*

*Acacia seyal:* For this species, each biomass component required different combination of predicators as for total AGB (DSH and CV), for stem biomass (DSH and TH) for branch biomass (DSH and CA) with all models forms showed significant improvement with adj. $R^2$>0.95 (Appendix 4.2, Table 4.1) and goodness of fit parameters over the models with single of these predicators.

Moreover, strong relationships between the predicators (DSH, DBH, TH, CA and CV) and total AGB were observed from regression graphs with adj.$R^2$ ranging between 0.81 and 0.97 (Fig. 2.5).

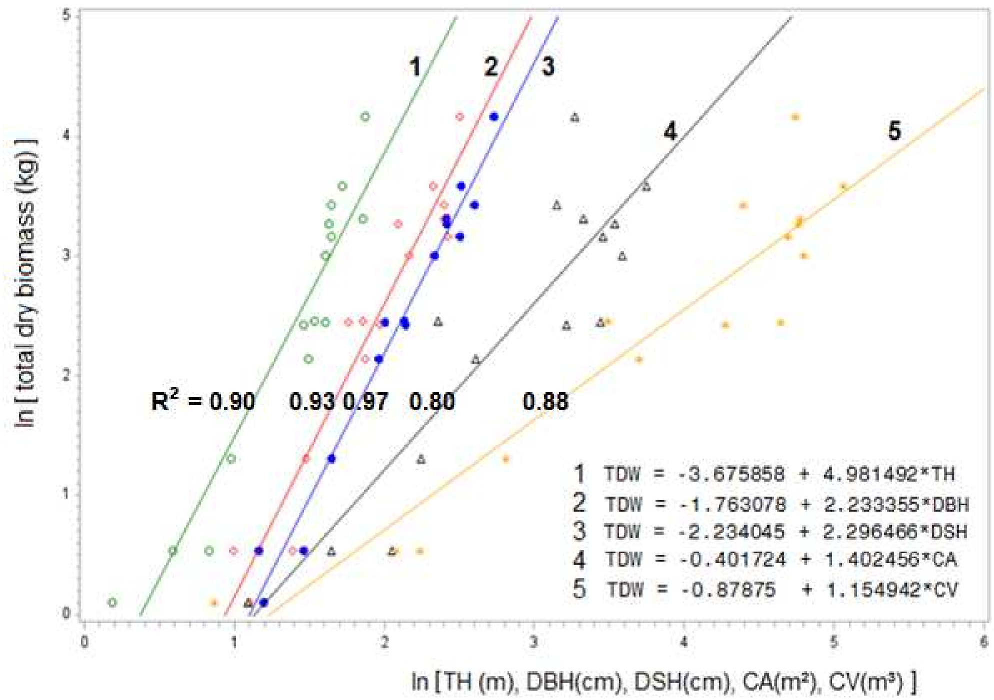


Figure 4.5. Regression lines (overlay) showing total dry biomass versus height (1), diameter at breast height (DBH: 2) diameter at stump height (DSH: 3), crown area (4) and crown volume (5) for Acacia seyal

*Acacia tortilis*: A combination of DBH and TH best explained all AGB components with adj.$R^2$>0.90 chosen as the final model fit biomass equation (Table 4.1), with 6 % increase in adj.R2 and values of MSE< 0.08 (about 58 % decrease) and delta AICc (11.62) and smaller values of PRESS as compared to models with single of predicators (Appendix 4.2). Moreover, individual predicators explained the variation in the total AGB as shown in Fig.2.6: DSH (Adj.$R^2$ = 0.92), DBH (adj.$R^2$ = 0.90) CA (adj.$R^2$ = 0.78), CV (adj.$R^2$ = 0.84) and TH (adj.$R^2$ = 0.84).

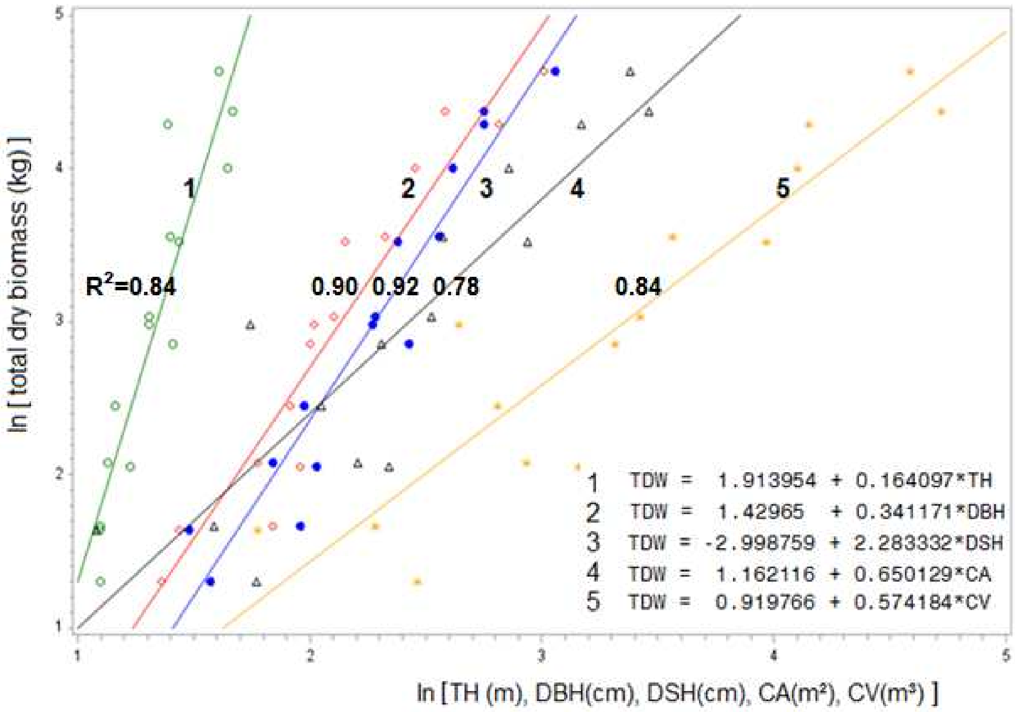


Figure 4.6. Regression lines (overlay) showing total dry biomass versus height (1), diameter at breast height (DBH: 2) diameter at stump height (DSH: 3), crown area (4) and crown volume (5) for *Acacia tortilis*.

*Lannea rivae:* For this species, the predictions of aboveground biomass best fitted only by the combination of DSH and CV. In general, a weaker fit was recorded for stem biomass model (adj.R2 = 0.70) as compared to total AGB and branch biomass models (adj.$R^2$ > 0.80) (Table 4.1).

*Rhus natalensis:* A combination of CV and TH showed relatively betterfit for estimating total AGB and branch biomass with (adj.$R^2$ = 0.62) as compared to the single use of CV (adj.$R^2$ = 0.59) and TH (adj.R2 = 0.43–0.50). However, stem biomass was less predictable to any of the predictor variables as presented by lower goodness of fit (adj.$R^2$<0.37 and higher MSE>0.30) (Appendix 4.2).

*Acacia brevispica:* For this species, relatively better model fittings (adj.$R^2$ = 0.62), was obtained in total AGB (DBH and CV) and for branch biomass (DBH and CA), where as for stem biomass poor fit of model equations (adj.$R^2$ = 0.35) and larger values of MSE = 0.88 (Appendix 4.2)

*Commiphora Africana:* For this species, either single variable or combination of any predicators did not fit to all the biomass categories. However, a slightly better fit was obtained (adj.R2 = 0.47) when interaction term was added (i.e., DBH and CA and DBHTH (Appendix 4.2).

*Mixed species allometric equations*

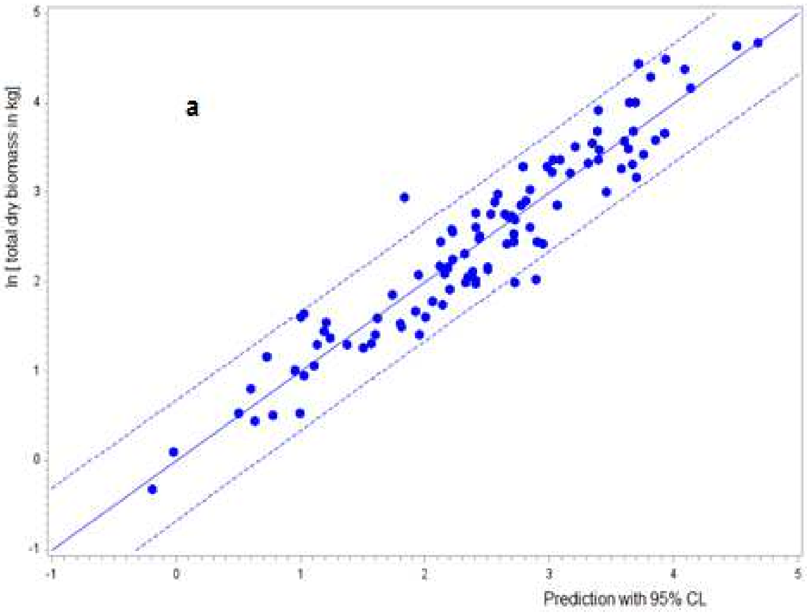
The coefficients and fit statistics from fitting the biomass prediction model to the pooled data of seven species show that the P-values for the intercept and the slope are significantly different from zero ($P < 0.0001$) (Appendix 4.3).

Similar to species specific models, the two-variable models (DSH and CA) had showed good fit for estimating the AGB (adj.$R^2 > 0.82$, $P < 0.0001$) for group of species. Diameter at stump height (DSH) alone also showed a good fit in stem biomass in this model development (adj.$R^2$ = 0.77) (Appendix 4.3).

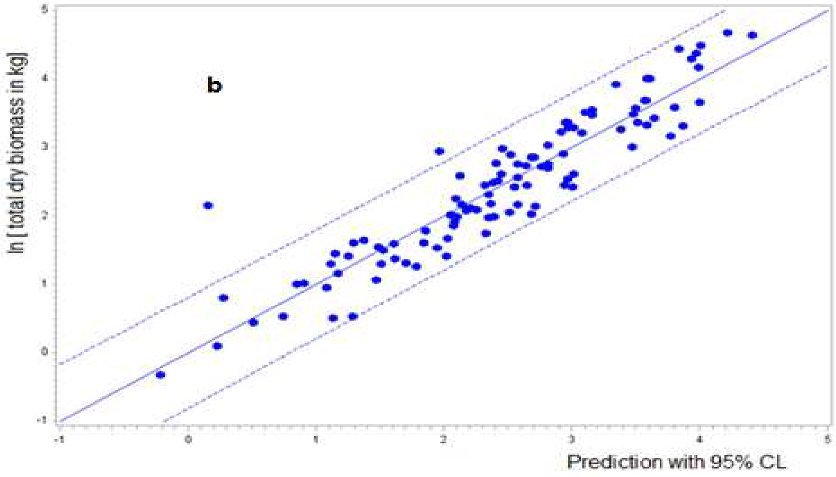
However, the optimal chosen best fits of the model was obtained with the combination of three predicator variables (DSH, TH and CA) (Table 4.2). The coefficient of determination (adj.$R^2$) increased from 0.83 to 0.90 (for total AGB), and from 0.82 to 0.89 (for branch biomass) and the values of delta AICc and MSE decreased (Appendix 4.3).

This good fit that is provided by the biomass prediction model to the data is confirmed by the regression fit of the total AGB against the predictors (DSH, TH and CA)(adj.$R^2$ = 0.90, at 95 % CI) (Fig.4.7a). Similarly, when DSH was substituted by DBH the total variation of AGB explained by the model was high (adj.$R^2$ = 0.86; Fig.4.7b).

In generalized allometric equations development, crown volume (CV) was systematically excluded from model due to the fact that it is highly correlated with other predicators as confirmed by VIF value>10.



ln (TDW)= -2.313 +1.531×ln (DSH)+ 0.670×ln(TH)+0.311×ln(CA) , Adj.$R^2$=0.904



Ln (TDW) = -1.586 +1.197 ×ln (DBH) + 0.818×ln (TH) + 0.321×ln (CA), Adj.$R^2$=0.862,

Figure4.7. Regression graph with 95% confidence interval for total dry biomass versus three predicators [diameter at stump height (DSH), TH and CA (a)] and [diameter at breast height (DBH), TH and CA (b)]

Table 4.1. Allometric equations for biomass estimation for selected woody species: The best fit model equation

| Name of species | Component biomass | Allometric equation | Adj.$R^2$ | MSE | P-value | CF |
|---|---|---|---|---|---|---|
| *Acacia bussei* | Total | $\ln (Wt) =-1.569 +1.119 \times \ln (DBH)+ 0.581 \times \ln (CV)$ | 0.94 | 0.06 | <.0001 | 1.03 |
| | Stem | $\ln( Ws )=-3.259+1.629\times \ln (DBH)+ 0.434\times \ln (CV)$ | 0.85 | 0.19 | <.0001 | 1.12 |
| | Branch | $\ln (Wb) =-1.650+1.460\times \ln (DBH)+ 0.348 \times \ln (CA)$ | 0.86 | 0.16 | <.0001 | 1.08 |
| *Acacia drepanolobium* | Total | $\ln (Wt) =-2.933+2.1822\times \ln(DSH)+ +0.336\times Ln (CV)$ | 0.97 | 0.05 | <.0001 | 1.02 |
| | Stem | $\ln (Ws) = -3.484+1.809\times \ln (DSH)+0.425\times \ln (CV)$ | 0.94 | 0.09 | <.0001 | 1.05 |
| | Branch | $\ln (Wb) =-3.797+2.557 \times \ln (DSH)+0.333\times \ln (CA)$ | 0.95 | 0.08 | <.0001 | 1.04 |
| *Acacia etbaica* | Total | $\ln (Wt) = -0.985+0.909\times \ln (DSH)+0.546\times \ln (CV)$ | 0.75 | 0.11 | <.0001 | 1.06 |
| | Stem | $\ln (Ws) =-2.124+ 1.181\times \ln (DBH)+0.786\times \ln (TH)$ | 0.68 | 0.22 | <0.0032 | 1.12 |
| | Branch | $\ln (Wb) =-2.022+0.914 \times \ln (DSH)+0.680 \times \ln (CA)$ | 0.84 | 0.15 | <.0001 | 1.08 |
| *Acacia nilotica* | Total | $\ln (Wt) = -2.963+2.191\times \ln (DBH)+ 0.740\times \ln (CV)$ | 0.93 | 0.06 | <.0001 | 1.03 |
| | Stem | $\ln (Ws) =-4.477+1.951\times \ln (DSH)+ 2.384\times \ln (TH) - 0.511 \times \ln (CV)$ | 0.86 | 0.09 | <.0001 | 1.05 |
| | Branch | $\ln (Wb)= -3.230+2.096 \times \ln (DSH)+ 0.354\times \ln (CV)$ | 0.93 | 0.08 | <.0001 | 1.04 |
| *Acacia tortilis* | Total | $\ln (Wt) = -3.043+1.4011\times \ln (DBH)+ 2.266\times \ln (TH)$ | 0.95 | 0.06 | <.0001 | 1.03 |
| | Stem | $\ln (Ws) = -4.837+ 0.710 \times \ln (DBH) +3.897\times \ln (TH)$ | 0.96 | 0.05 | <.0001 | 1.03 |
| | Branch | $\ln (Wb) =-3.148+ 1.741\times \ln (DBH)+ 1.503 \times \ln (TH)$ | 0.93 | 0.08 | <.0001 | 1.04 |
| *Acacia seyal* | Total | $\ln (Wt) =-2.444+1.913 \times \ln (DSH)+0.224\times \ln (CV)$ | 0.98 | 0.03 | <.0001 | 1.04 |
| | Stem | $\ln (Ws) =-2.900+ 1.719 \times \ln (DSH)+0.466 \times \ln (TH)$ | 0.96 | 0.05 | <.0001 | 1.04 |
| | Branch | $\ln (Wb) =-3.406+ 2.059 \times \ln (DSH)+0.378\times \ln (CA)$ | 0.98 | 0.04 | <.0001 | 1.05 |
| *Lannea rivae* | Total | $\ln (Wt) = -3.148 +1.925 \times \ln (DSH)+0.469 \times \ln (CV)$ | 0.81 | 0.05 | <.0001 | 1.03 |
| | Stem | $\ln (Ws) =-4.225+ 2.333\times \ln (DSH)+0.390 \times \ln (CV)$ | 0.7 | 0.09 | <0.0008 | 1.04 |
| | Branch | $\ln (Wb) = -2.781+ 1.698 \times \ln (DSH)-0.601 \times \ln (CV)$ | 0.81 | 0.06 | P<0.001 | 1.07 |
| Mixed species | Total | $\ln (Wt)=-2.31+1.53\times \ln(DSH)+0.675 \times \ln(TH)+ 0.310\times \ln (CA)$ | 0.9 | 0.11 | <0.001 | 1.06 |
| | Stem | $\ln (Ws) =-3.280+1.60 \times \ln (DSH)+ 0.660 \times \ln (TH)+ 0.170\times \ln (CA)$ | 0.84 | 0.17 | <0.001 | 1.09 |
| | Branch | $\ln (Wb) =-2.87+1.530 \times \ln(DSH)+ 0.67 \times \ln (TH)+ 0.370 \times \ln (CA)$ | 0.89 | 0.14 | <0.001 | 1.07 |

Note: Wt-total aboveground biomass, Ws-stem biomass and Wb-branch biomass, the best fit model waschosen from Appendix 4.1 and 4.2. Adj.$R^2$- adjusted coefficient of determination, MSE-mean standard error, CF – correction factor. DSH- diameter at stump height (cm), DBH- diameter at breast height (cm), TH -total height (m), *CA* crown area ($m^2$) and CV-crown volume ($m^3$)

## 4.4. DISCUSSIONS

### 4.3.1. Species-specific allometric equations

The selection of regression models in estimating the above-ground biomass and its components for sample woody species was based on the goodness of fit statistics such as the adjacent coefficient of determination (adj. $R^2$ values), the mean standard error, and Akaike information criterion.

Accordingly, the diameter at stumheight (30 cm), diameter at breast height (DBH) andcrown volume (CV) alone or in combination provided the best fit to the data from individual species.However, a comparison of the models developed inthis study showed that the form of these variables used varied across the woody species which could be due to difference s in species in their canopy architecture and branching patterns.

Hence, a combination of DSH and CV, and DBH and CV best fittedin predicting the AGB for *A. seyal, A. nilotca andLannea rivae*. In line with other few authors (e.g., Tietema, 1993; Hasen-Yusuf et al., 2013) also used a combined stem ankle andcanopy related as best p predictor variables in above-ground woody biomass.

Other earlier models published from tropical forestand shrub lands, however, have used only circumferenceor cross-sectional area of the stem alone as best model prediction (Okello et al., 2001; Henry et al., 2011).

In the present study, wealsofound that model with DBH-H shows best fitted for predicting the AGB and its component for some species with umbrella-like canopy structure and with a relatively higher dry biomass (e.g., *A. tortilis* and *A.nilotca*) . In line with, previously published equations forlarge trees (e.g., Brown et al., 1989; Chave et al., 2005; 2014) also reported strong dimensional relationshipsbetween biometric variables (DBH-H) and theAGB.

In contrast, Kuyah et al. (2016) have found that with DBH alone as the predictor variable and was alsobetter than models that include height and crown areaas additional predictor variables.However, in some other models developed from a combination of DBH and TH improved performance in estimatingthe AGB of shrubs and small trees within their specificsize range variability (i.e., DBH<5 cm) (Litton andKauffman, 2008).

In our present study, we found anadj. $R^2$ value ranging between 0.79 and 0.94 for models developed from DBHalone. Similarly, Chamshama et al. (2004) also reported an $R^2$ value of 97 % in miombowoodland in Tanzania using the model equation of $\ln B = 0.01559 + 2.796 \times \ln DBH$.

In general, for woody plants with single stems , DBHis animportant predictor of total tree biomass but inwoodlands where branching of the stem below breast height is common, diameters of stump values (basal area at ankle height at 5–10 cm above ground level) are often used (Tietema , 1993).

Furthermore, the use of models where tree biomass is determined from DBH has a practical advantage because most of theinventories include DBH measurements which areeasy to carry out accurately measure in the field (Segura andKanninen, 2005).

The total rank for contribution of each predicatorvariables shows that crown area and total treeheight are not as accurate as the stem diameters (DBH and DSH) and crownvolume to predict total above-ground biomass in selected woody species which could be irregularity and variable crown shapes across the woody species and individual trees harvested.

Different studies from somewhere else in different region also used a different combination of predicator variables based on specific location and vegetation structure. For example, Chave et al. (2005) used a combination of trunk diameter, tree height and woody specific gravity, other (Sawadogo et al., 2010) used diameter atankle height, diameter at breast height and treeheight and Kuyah et al. ( 2012) also used diameter at breastheight, tree height, woody density and crown area in developing species specific allometric equation

In our present findings, the biomass prediction models for two species (*A. brevispica and Rhusnatalensis*) were found to be poor model fit and lack of total fit toall AGB for *C. africana* (adj.$R^2$ = 0.26–0.47).In contrast, Hasen-Yusuf et al. (2013) from other sites in the same region found a highly significant and best fit for the total abovegroundbiomass (adj.$R^2$ = 0.93) and stem biomass (adj.$R^2$ = 0.59) using canopy volume and stem basalcircumference for the same species. Thesame authors also reported no fit possible for the branchesbiomass of the same species.

Thislack of fit for *C. Africana* in this study might be due to its growth form where mixed stands of wooded savanna as in the case of Dida-Hara site, its growth form may make the canopy shape complicated. Furthermore, sampled trees are younger and succulent at time of harvestedas it can also be seen from its higher moisture content of 66.48 % as compared to the remaining woody plants.

In line with our present findings, Sawadogo et al. (2010) also reported poorerpredictability forsome species of Acaciaspeciesand *P.tlionningiiin*in West African savannas with bushy andcomplicated shapes, which might have also made itdifficult to accurately predict their biomass.

Therefore, developing in species specific allometric equations models at site and spices levels may accountable for differences in specific growtharchitecturean stages of growth conditionof the species (Abola et al., 2005; Kuyah et al., 2012). Furthermore, different allometric equations within the same climaticzones also been reported by Abola et al. (2005).

In general, the models we presented here showed that the total and branches biomass components were predicted relatively better as compared to the stem biomass which is in agreement with the findings of Hasen-Yusuf et al. (2013).In contrast, Sawadogo et al. (2010) reported that biomass of branches and twigs were less predictable compared to stem biomass and total biomass for woody tree species.

Hence, the growth form and location of a species needs to be taken into account (e.g., whether neighboring trees compete for light and thus, may reduce branch growth) in species-specific componentspredictions as reported by Henry et al. (2011).

### 4.3.2. Mixed species allometric equation

In this study, different mixed species allometric equations models follow some different from that of the individual species equation models particularly in terms ofthe number of variables included. Accordingly, the best fitted chosen models in mixed species across the three biomass components (stems, branches and total) were based on the combination of three predicators (DSH-TH-CA).

Similarly, Chave et al. (2005) developed mixed-species equation model using three but different combination of predicator variables (DBH-, TH- woody density) using the temperate forest tree biomassdata.

Others (e.g., Henry et al., 2011; Kuyah et al., 2012; Hasen-Yusuf et al., 2013) also developed generalized allometric models for a group of woodyspecies to overcome difficulty of species specificmodel requirements for biomass and carbon stockestimations particularly in savanna ecosystems withdiverse species.

However, Navar et al. (2004) argued that when adding more species together might reduce predictivepower and increase bias error. Northup et al. (2005) also emphasized that the use of site-specificrelationships is more precise given that species sizebiomass relationship could differ as plants alterallocation patterns in response to soils, climate andother disturbances.

## 4.5. CONCLUSIONS

The use of several dendrometric (predicator) variables in our equation model increased the model accuracies as we also observed strong positive relationship between total AGB and the predicator variables.

In addition, our research on species specific and mixed allometric equations provide an empirical evidence that can be used to predict aboveground biomass in branches and stems separately which could be essential for studies of carbon sequestration with different allocation of carbon allocation in stems and branches

The biomass prediction models derived here will also provide an ideal opportunity for further work on the verification of woody biomass calculations, thus provide a proxy estimate of carbon sequestered in woody vegetation of East Africa rangelands.

This will improve the estimates of carbon stocks in the region from IPCC Tier 1 (the most generic) to Tier 2 and Tier 3 (species-specific) (IPCC, 2006) and support the implementation of policies and mechanisms designed to mitigate climate change (e.g., REDD or in CDM) (Agrawal et al., 2011; Sundarapandian et al., 2013).

However, the allometric biomass equations of selected woody species provided in this study might be helpful toward the generation of more accurate estimations of the AGB. However; care should be taken when applying the allometric models in wider range of arid and semi-arid ecosystems.

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# CHAPTER FIVE

# 5. EFFECTS OF LONG-TERM BAN OF FIRE ON CARBON DYNAMICS AND SOIL PROPERTIES IN BORANA RANGELANDS, SOUTHERN ETHIOPIA

Submitted to journal Catena as: Kenea Feyisa[a], Sheleme Beyene[a], and Ayana Angassa[b].Effects of long-term ban of fire on carbon dynamics and soil properties in Borana rangelands, southern Ethiopia.

**Status:**under revision based on comments received from the editor's and reviewers comments [a] [a,] School of Plant and Horticultural Sciences, College of Agriculture, Hawassa University, P.O.Box 05 Hawassa, Ethiopia

[b] Department of Animal Science and Production, Botswana University of Agriculture and Natural Resources, Private bag 027, Gaborone, Botswana

## ABSTRACT

*Fire management was a common practice in Borana rangelands of southern Ethiopia, this practice, however, was discontinued in the 1970's following a ban on fire imposed by the government.For this, three prescribed burned sites (replicates) each paired with unburned areas (40 years of fire exclusion) were selected to investigate for differences in soil organic carbon (SOC) and total nitrogen (TN) contents and stocks and above-ground biomass carbon accumulation,while accounting for effects of landscape position (upland and bottomland) and soil depths. We collected soil samples at three soil depths (0–5, 5–15 and 15–30 cm) and vegetation attributes from 60 plots within burned and adjacent unburned sites at each landscape. Although our results showed that burned and unburned pairs did not show significant ($P > 0.05$) differencesin terms of both soil organic carbon (SOC) content and stock acros the two landscape sitesrelatevly higher values of these parameters were recored under prescribed burned range units across the three soil depths. Accordingly, total mean of 48.19±2.29 Mg ha$^{-1}$in the upland and 40.25±2.57 Mg ha$^{-1}$ in bottomland landscape of burned sites, whereas in the corresponding unburned sites of upland 43.17±2.40 and 36.54±1.94 Mg ha$^{-1}$ in the unburned sites of bottomland in the 0-30 cm depth were recorded. Similarly, total nitrogen (TN) content and stock in unburnt areas were higher than that of the unburned areas, although the difference was significant ($P < 0.05$) only in upland site.Moreover, the herbaceous biomass carbon in burned areas was signficalty higher than unburned sites, where as although not singficanr ($P> 0.05$), a relatively higher woody biomass carbon stock was recorded in the unburned sites. The results, therefore, can conclude that a fire regime characterizes the spatial patterns of carbon and nitrogen dynamics. However, further study will be needed to understand other interactive factors such as climate and grazing pressure along with fire intensity on carbon and total nitrogen dynamics at the sites.*

**Key Words:** Borana, Fire, Soil carbon, Soil nitrogen, Biomass carbon

## 5.1. INTRODUCTION

Uderstanding the impact of fire on carbon dynamics is fundamental in view of the emerging climate change mitigation agenda (Zhao et al., 2012; Valentini et al., 2014).Fire is also an integral part of the land use practices, primarly used on rangelands to manipulate natural vegetation and enhance the suitability of vegetation and productivity for livestock (Fynn et al. 2003; Sankaran et al., 2005; Berniger et al., 2009).

Furhtermore, fire has the potential to alter soil carbon storage by influencing rates of net primary productivity, carbon (C) allocation patterns and rates of organic matter decomposition (Ojima et al., 1994).

In particular, controlled (prescribed) fire is an essential tool to intentionally manipulate rangeland ecosystems (Higgins et al., 2007; Piñeiro et al., 2010). Conversely, uncontrolled fire can lead to ecosystem destruction associated with the disruption of the fundamental nutrients cycle (Coetseeet al., 2010).

Similalry, earlier studies (e.g., Bond and Keeley, 2005; Beringer et al., 2009) showed that reduced fire frequencies allow the proliferation of woody encroachment into savanna ecosystems. At the same time, fire can be used as a tool for landscape management (Costa et al., 2011).

Increased woody cover and density may appear as a viable option for carbon sequestration and increased nitrogen (N) cycling. Soil carbon (C) and nitrogen (N) storage may, however, reverse at very high woody densities leading to a net loss of C when savannas are replaced by thicket and grass cover is lost (Williams et al., 2011).

Other studies (Oluwole et al., 2008; Rau et al., 2010; Cook et al., 2015) also documented both positive and negative responses of carbon to the effects of fire management, where some of the studies (e.g., Bird et al., 2000) found the reduction in soil carbon stock attributed to the effects of fire.

Conversely, Coetsee et al. (2010) reported that there is no significant influence following 50 years of frequent burning on SOC in southern African savannas. Hence, the effect of fire on terrestrial carbon stocks are a function of the balance between carbon loss from direct fire emissions and decomposition and carbon gain from vegetation regrowth (Fynn et al., 2003).

It has been argued that recent increases in woody cover in African savanna are caused in part, by less frequent and less intense fires (Coetsee et al., 2010). However, the long-term impacts of fire suppression on the soil properties, more on C and N dynamics in dry savannas are poorly documented (Sawadogo et al., 2005).

According to Sankaran et al. (2008), fire return interval is one of the most important variables in regulating woody plant cover along an extensive ecological gradient in African savannas. Therefore, there is a need to investigate the impact of long term fire suppression and how controlled (prescribed) fire management influences soil C and N dynamics in savanna ecosystems of East Africa.

Traditionally, fire was used by the Borana pastoralists to stimulate fresh grass growth, reduce invasive bush species, and to help control tick populations (Angassa and Oba, 2008). On the other hand, fire was officially prohibited in the rangelands of southern Ethiopia in the early 1970s (Coppock, 1994), with the goal of safeguarding the natural forests against wildfires (Gebru et al., 2007).

The consequences of fire ban in the Borana rangelands of southern Ethiopia led to the expansion of bush encroachment with a decline in perennial grasses (Coppock, 1994).Few studies (e.g., Coppock, 1994; Oba et al., 2000) reviewed the consequences of the transformation of the vegetation in southern Ethiopia.

A similar study by Angassa and Oba (2008) showed that prescribed fire in combination with bush thinning practices increased the accumulation of herbaceous biomass and richness to maintain the balance between the woody and herbaceous layers. Others (Gebru et al., 2007; LaMalfa et al., 2008) also indicated the importance of prescribed fire in conjunction with other appropriate range management practices.

Furthermore, Bikila et al. (2016) also reported that prescribed fire is essential in enhancing carbon storage in soil organic matter and plant biomass. Hence, the re-introduction offire in the Borana rangelands was very recent in the last 10 years by encouragement of the Borana pastoral communities and facilitated by community leaders, government and non-government organizations as well as other institutions who understand the importance of using fire to manage rangelands (Gebru et al., 2007).

Although these studies and observations were conducted in the Borana rangelands of southern Ethiopia, information is still scarce with landscape heterogeneity in terms of clarifying the long term impact of fire ban on the potential of carbon sequestration, changes in soil properties and vegetation biomass.

Therefore, the objectives of this study were to :(1) investigate the impact of fire ban (burned vs. unburned) on SOC and TN contents and stocks in different soil depths ; (2) assess the impacts of fire ban on carbon storages in the above-ground vegetation (woody and herbaceous)biomass.

We hypothesized that: (i) prescribed fire would increase soil nutrients, more specifically soil organic cabon total nitrogen due to the increase of understory layer and the reduction of the direct release of nutrients in the fire, (ii) unburned areas accumulate more carbon stocks in woody biomass as compared to the burned rangeland units.

## 5.2. MATERIALS AND METHODS

### 5.2.1. Study area and traditional fire management

The study was conducted at two sites, namely, Dikale and Sanke (Dambi cluster) in Dida-Hara area in Yabello district, southern Ethiopia. The detailed description of the study area was given under the general materilas and methods section (Chapter I). Dida-Hara is located between an altitude range of 1260 and 1700 meter above sea level (m.a.s.l.) (Angassa and Oba, 2010).

Prior to the official ban on fire, the communities at the study sites were practiced traditional fire varying between one and five year intervals to suppress bush growth by killing encroaching woody species (Angassa and Oba, 2008). Discussions with pastoralistsrevealed that the ban on fire had since the period from 1976 by the government, which is almost 40 years when this field work was carried out.

Lack of use of fire history in Borana was also partially reconstructed from environmental change, mostly in climate, which in turn influences vegetation and fuel load.Moreover, the proliferation of villages in the area could constitute a hazard should fires get out of control.

As a result, encroaching woody species are recognized as reducing grazing land through colonising rangeland, as well as out-competing herbaceous grasses for nutrients, thus reducing grass cover (and feed for livestock).

Bearing these factors in mind, government and non-governmental organizations (NGOs) and hence mentioned by pastoralists are re-introducing burning as a rangeland management technique through ‘prescribed fire’ - the controlled and managed application of fire to defined units of grassland towards counteracting the proliferation of these plants and encouraging grass growth

### 5.2.2. . *Selection of the study sites and experimental design*

A preliminary field survey was carried out between May and June 2013 to identify rangeland sites with known history of exsting fire experiments.The practices of fire regime were assessed from review of pertinent publications, unpublished records andfield visit focused on the influence of fire suppression in the study area.

Following this initial step, two sites (Dikale and Sanke) each consisting of burned vs. unburned sites were selected through systematic random sampling techniques, with Dikale in the uplands (≥1500 m a.s.l.) and Sanke in the bottomlands (≤1400 m a.s.l.) in topography. Selection of the sampling sites was made with the help of knowledgeable community members and experts working in the study areas.

The two firesites weredistributed within 20 km radius in Dida-Hara area. In addition, it was confirmed from community elders and expert working in the area where prescribed burning was conducted during the last 7 years (in 2006) at time of field sampling in 2013 versus 40 years of unburned sites.

The single planned burning (prescribed fire) was carried out in year2006 during monthbetweenFebruarys to March before the onset of rainy season (LaMalfa et al., 2008). Burns were conducted with average ambient temperature of 27°Cwind speed from 20-32 km/h and relatie humidty from 25-40 %, and the grass growth to a minimum of 30 cm height (or fuel load of about 25oo kg/ha ). Head fires were set after 3 pm when humidity and temperature were declining (Sexton et al., 2006).

The experimental design consisted of pairs of burned (prescribed fire) and the adjacent unburned areas (40 years of fire suppression) along the two landscapes, each with three burned sites (replicates) (Fig.5.1a), so that comparisonscould be made of burning versus no burning. Paired sites were selected which have the same climate soil types, vegetation type and topographic features.

In each paired site, a 500 m long line transect was established in the burned and adjacent unburned sites, ensuring that soil and terrain conditions were similar as possible between each pairs. We established 10 sampling plots (30 m× 30 m each) along a 500 m long line transect in each paired burned vs. unburned site.In establishing the sampling plots, systematic random sampling was used where in each fire management unit, the first sampling unit was established randomly and subsequent plots.

The study plots were well spaced (separated by at least 50 m) and randomized in each treatment to account for spatial variability of soil and vegetation (Fig.5.1b). We avoided sampling within 30-50 m distances between the paired burned and adjacent to avoid adge effect.

Field data collection was conducted between the months of June and July 2013 (postburn) to determine fire effects on soil C and N and vegetation biomass dynamics.A total of 60 paired-plots (one unburned, one burned) at each landscape site.

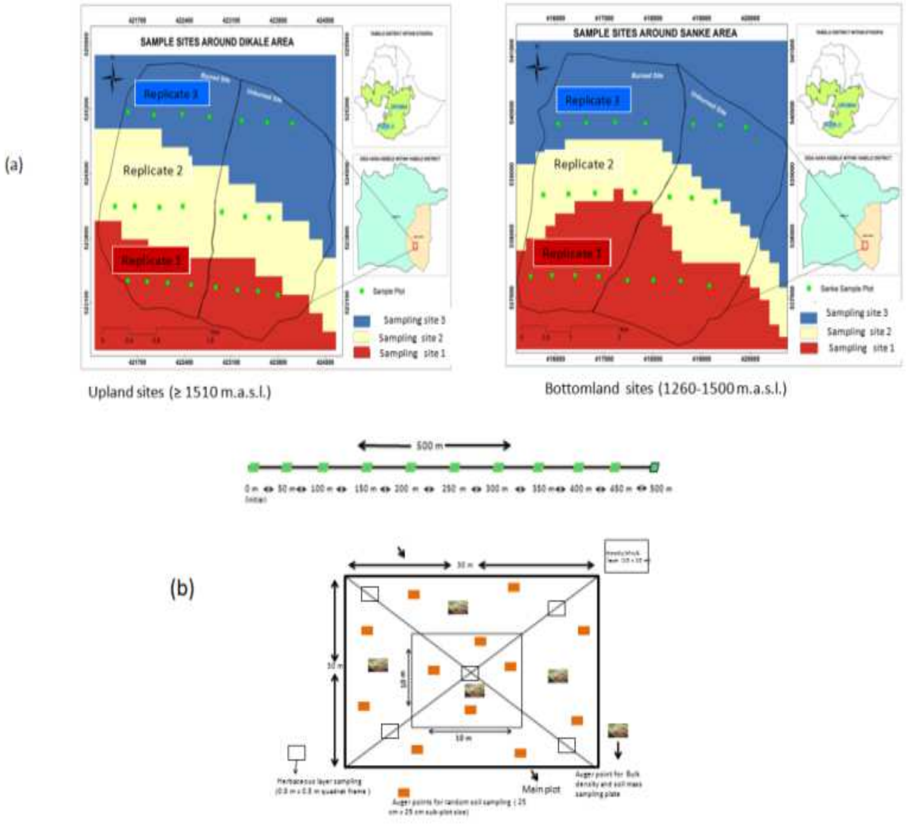

Figure 5.1(a) Map of the study sites and (b) diagram sketch of 500 m baseline set up with one transct

Note: in each of the 50 m stretches plots and sub-pots layouts were allocated for collecting soil and vegetation samples. The diagrammatic sketch for plot and sub-plots are not drawn to standard scale.

### 5.2.3. Soil sampling, laboratory analysis and calculation SOC and TN stocks

In each plot, we collected random soil samples at three soil depths (0–5, 5–15 and 15–30 cm) from 12 sub-plots The sub-plots were well spaced at minimum of 15 m distance from each other to account for heterogeneity in soil and vegetation characteristics within the site, although not drawn to the standard scale (Fig. 5.1b)

Commualtive mass of soil samples were from four spots (sub-plots) placed the in which the first sampling point was near the center of the plot, and the other sub-plots (auger points) were located in systematic random pattern (Fig.5.1b). A total of 60 paired-plots (one unburned, one burned) within at each landscape postion sampled.

Soil samples preparation, processing and analysis and calculation of SOC and TN stocks were conducted at the Soil and Plant Testing Laboratory of Hawassa University, Ethiopia (seefordetails under the general materials in **chapter I**).

### 5.2.4. Vegetation survey and biomass estimation

In each plot, a 10 x 10 m quad sub-plot was for woody vegetation survey. Data included a list of all woody plants (mature trees, sapling and seedling) species, life form of each, and

Non-destructive of dendrometric measurements namely, total tree height diameter at stump height (cm) at 30 cm ground and dimeter at breast height (cm) at 1.3 m above-ground surface, two crown dimentions [(the long axis (d1) and short axsis perpendicular to the long axis (d2)] used to calculate crown area (CA) and crown volume (CV) (see the detailed proceure and calculation in **chapter IV).**

Herbaceous vegetation wassampled using five sub-plots (0.5 m x 0.5 m each). Samples from the five sub-plots were pooled together as dry weight (gm $m^{-2}$) and oven dried for dry biomass estimation (detailed under chapter **I).**

Total above-ground biomass (AGB) for woody species estimated using the developed species specific allometric equitation (Feyisa et al., 2016) for thee same study area (see details in Appendices 5.1, 5.2. and 5.3).The total dry weight (kg/tree) was converted to Mega gram per unit area (ha) multiplied by the expansion factor (unit area (ha) divided by plot area).

Finally, the carbon content of dry biomass was estimated using the default value 0.47 (i.e., 47% of the dry biomass is assumed carbon) (IPCC), 2006) and results were expressed in mega gram per unit area

### 5.2.5. Statistical analysis

Before data analysis, our data set were checkd for normality and homogeneity of tests, outliers were removed if any. The effect of fire on soil properties, SOC and TN stocks were analyzed using linear mixed model procedure (proc mixed).

Comparisons were carried out with the use of SAS mixed-effects models with categorical fixed variable for soil parameters included:[fire regime (dummy variable: burned vs. unburned areas) in each depth of the soil (three levels) and plot as a random effect for each landscape.

For vegetation attributes, the model included effects of fire regime (burned vs. unburned) for each landscape and combining all sites. The sampling plot was included in the model as a random effect to control for repeated measures.Means comparisons were made using Tukey's test (at alpha0.05).All statistical analyses were carried in SAS ver. 9.2 (SAS, 2012).

## 5.3. RESULTS

### *5.3. 1. Effects of fire on bulk density, SOC and TN contents andstocks*

In the present study, soil bulk density did not show any significant ($P > 0.05$) difference between burned and the adjacent unburned areas (Table 5.1). However;the unburned rangeland unit had showed relatively higher in this parameteras compared to thatof the burned plots across the three soil depths.Moreover, the difference between burned and unburned areas in terms of bulk density apparently more at the upland position in the lower (15-30 cm) soil depth (Table 5.1).

Our results in this study also showed that both SOC content and stock were not significantly ($P > 0.05$) different between burned and unburned areas (Table 5.1). In the upland site, however, the burned site resulted in a significant ($P < 0.05$) in proportional increase of 27.1% SOC content as compared to the adjacent unburned sites in the 15-30 cm depth(Table 5.1).

In the same soil depth (15-30 cm), the burned sites in the bottomland, however, resulted in marginal reduction of 4.5%SOC contentwhen compared with that of the adjacent unburned sites, although the difference was notsignificant (Table.5.1).

Similarly, burned sites at the upland sites had showed an increaseof 1.24, 0.64 and 3.15 Mg $ha^{-1}$ inSOC stocks, respectively in the 0–5, 5-15 and 15-30 cm soil depths, while the values were 1.02, 1.94 and 0.94 Mg ha$^{-1}$ in burned areas of the bottomland sites in the same order of soil depths as compared to their adjacent unburned areas (Table 5.1).

The above values can be converted to percent of increase by 15.6, 4.37 and 15.32%in SOC stocks in burned site at the uplandand by 11.4, 16.8 and 5.9% at the bottomland postionwhen compared with that of their adjacent unburned sites across the three soil soil depths, although the increaments did not bring singficant difference.

However, apparentlymore increase in the burned sites than unburned sites in termsof SOC stock was recorded inthe 0-5 cm and 15-30 soildepths in the upland sites, while for burned site at the bottomland postion was in the 5-15 cm soil layer (Table 5.1).

In this study, response of total nitrogen (TN) content and stocks to prescribed bruning was significant ($P < 0.05$), but thissignificant higher TN content in burned than unburned sites was obtained only at the upland site.

Accordingly, burned areas of the upland sites had proportional increases of TN content by 41.7 and 27%, respectively at the 0-5 cm and 15-30 cm soil depthsas compared to the adjacent unburned areas (Table 5.1).

In the burned sites of the bottomland, however, only negligible increase in TN content as compared to the adjacent unburned sites across the three soil depths (Table 5.1). Likewise, the effect of prescribed fire on TN stock varied across the landscape positions and soil depths.

Consequently, the burned sites of the uplands had significantly ($P <0.05$) higher TN stocks that ranged from 0.49 to 1.27 Mg ha$^{-1}$than the adjacent unburned sites, whereas burned areas of the bottomland site showed only slight increase in TN stocks when compared with that of the unburned sites across the three depths (Table 5.1).

The present results also showed that no significant ($P > 0.05$) difference between burned and the adjacent unburned areas in terms of C: N ratio across both landscape postions and three soil depths (Table 5.1).

Table 5.1. Mean values of bulk density (BD), soil organic carbon (SOC) andtotal nitrogen (TN) contents and stocks and carbon to nitrogen (C: N) ratio in burned and the adjacent unburned sites across the three soil depths at two landscapes in Dida-Hara area of Boran

| Soil properties | Depth (cm) | Dikale site (Upland; ≥1500 m.a.s.l.) | | Sanke site (Bottomland; ≤1400 m.a.s.l.) | |
|---|---|---|---|---|---|
| | | Burned | Unburned (Adjacent) | Burned | Unburned (Adjacent) |
| BD (g cm$^{-3}$) | 0-5 | 1.10±0.1 | 1.34±0.06 | 1.11±0.04 | 1.15±0.06 |
| | 5-15 | 1.32±0.05 | 1.42±0.04 | 1.23±0.04 | 1.31±0.06 |
| | 15-30 | 1.52±0.05 | 1.60±0.04 | 1.26±0.06 | 1.43±0.08 |
| SOC (%) | 0-5 | 1.28±0.03 | 1.12±0.05 | 0.93±0.05 | 0.92±0.04 |
| | 5-15 | 1.15±0.04 | 1.0±0.06 | 0.82±0.04 | 0.81±0.04 |
| | 15-30 | 1.04±0.04$^{a}$ | 0.82±0.05$^{b}$ | 0.64±0.04 | 0.67±0.04 |
| SOC stock (Mg ha$^{-1}$) | 0-5 | 9.20±1.17 | 7.96±0.43 | 9.93±0.94 | 8.91±0.81 |
| | 5-15 | 15.29±1.16 | 14.65±0.85 | 13.52±0.96 | 11.58±0.71 |
| | 15-30 | 23.71±1.41 | 20.56±1.41 | 17.00±1.53 | 16.06±1.11 |
| TN (%) | 0-5 | 0.17±0.01$^{a}$ | 0.12±0.01$^{b}$ | 0.13±0.01 | 0.14±0.01 |
| | 5-15 | 0.14±0.01$^{a}$ | 0.11±0.01$^{b}$ | 0.11±0.01 | 0.12±0.01 |
| | 15-30 | 0.12±0.01 | 0.10±0.01 | 0.10±0.01 | 0.11±0.05 |
| TN stock (Mg ha$^{-1}$) | 0-5 | 1.73±0.29$^{a}$ | 1.24±0.13$^{b}$ | 1.40±0.13 | 1.37±0.12 |
| | 5-15 | 1.93±0.20$^{a}$ | 1.38±0.10$^{b}$ | 1.81±0.11 | 1.80±0.12 |
| | 15-30 | 3.31±0.26$^{a}$ | 2.04±0.12$^{b}$ | 2.41±0.19 | 2.39±0.19 |
| C:N ratio | 0-5 | 8.53±0.56 | 9.21±0.48 | 7.16±0.36 | 6.87±0.36 |
| | 5-15 | 9.26±0.63 | 9.74±0.67 | 7.61±0.42 | 6.84±0.38 |
| | 15-30 | 9.40±0.59 | 9.15±0.78 | 6.76±0.46 | 6.58±0.40 |

N= 3 – replication of experimental sites. Means (± SE) followed by different letters are statistically different between burned and the adjacent unburned sites, after Tukey's test (adjusted) at $P < 0.05$

### *5.3.2. Effect of fire on above-ground biomass carbon stocks (woody and herbaceous)*

In the study, a total of 24 woody tree species were identified across the two landscape units, out of which 12 woody tree species that are dominant contribution and with DBH than > 2.5 cmmeasurement of their dendrometric variables and used in the allometric equations to estimate the above-ground biomass are presented in Appendices 5.1and 5.2.

Overall, total woody density was greater in the urnburned rangeland units (4,820 stems ha$^{-1}$) and in the prescribed burned areas (3370 stems ha$^{-1}$). However, both burned and the adjacent unburned sites were having almost similar in terms of types woody species and their lists are summarized Appendix 5.3.

According to our finding, burning had significantly (*P*< 0.001) higher herbaceous biomass carbon with 4.75±0.77 and 1.79±0.11 Mg ha$^{-1}$in the burned areas of the upland and bottomland positions, respectively, and in the respective unburned areas the values were 1.30±0.20 and 0.65±0.05 Mg ha$^{-1}$, respectively.

Hence, the difference between the burned and adjacent unburned fire treatments was more apparent at higher altitude as it was observed from Figure 5.2.An averageacross both landscape sites, herbaceous biomass carbon stock was3.27±0.43 Mg ha$^{-1}$in theburned sites and in the unburned site it was about 0.98±0.43 Mg ha$^{-1}$

The amount of carbon sequestered in woody biomass did not show significant (*P* > 0.05), althoughburned site had 29.97±6.65 Mg ha$^{-1}$ (10.43%lower) in the upland and 29.51±3.73Mg ha$^{-1}$ (22.44% lower) at the bottomland as compared to their respective adjacent unburned sites (33.46±4.24 and 38.05±1.94Mg ha$^{-1}$, respectively. However, apparently more difference between paired burned and unburned sites was observed at the bottomland position (Fig.5.2).

Likewise, burned site had 29.74±3.78 Mg ha$^{-1}$woody biomass carbon stock, and in the adjacent unburned sites was 35.75±3.28 Mg ha$^{-1}$i.e., the unburned rangeland unit had resulted in 6.1 Mg ha$^{-1}$ (20%increase) as compared to the burned sites when averaged for both landscape sites (see also Fig.5.2).

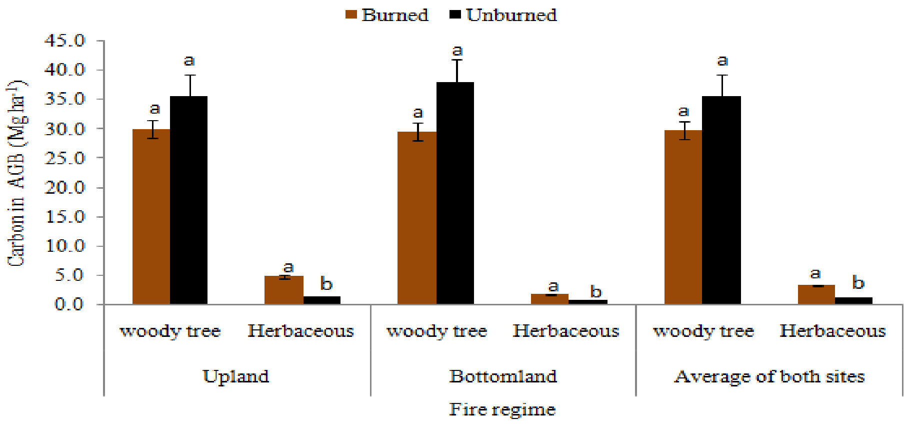

Figure5.1.Mean values of above-ground biomass carbon stocks (woody and herbaceous vegetation) in burned and the adjacent unburned sites at each landscape and averaged for all sites. Q b

Bars followed by same letter did not differ, after Tukey's test (adjusted). Error bars indicate standard error of the mean (SE).

### *5.3.3. Effect of fire on carbon pools (soil and vegetation)*

Overall, a total mean SOC stocks of 48.19±2.29 and 43.17±2.40 Mg ha$^{-1}$, respectively for the burned upland and bottomland sites, whereas,the values 40.25±2.57 and 36.54±1.94 Mg ha$^{-1}$ for the corresponding adjacent unburned site within 0-30 cm depth were recorded (Table 5.2).

Likewise, we rcored a total mean TN stocks of 6.97±0.45 and 5.62±0.27 Mg ha$^{-1}$,respectively in the burned upland and bottomland sites, and for the corresponding adjacent unburned sites the values were 4.65±0.16 and 5.56±0.29 Mg ha$^{-1}$TN stock in the top 0-30 cm depth. However, a significantly higher ($P < 0.05$) TN stock in the burned sites than the adjacent unburned treatment was obtained only in the upland site (Table 5.2).

The figures can be converted to TN stock accumulation rate of 0.33 and 0.01 Mg ha$^{-1}$ yr$^{-1}$, respectively in the burned areas (average of 7 years of post-fire) in the upland and bottomland sites, as compared to their corresponding adjacent unburned sites.

Our results also showed carbon pools (soil and vegetation biomass) were not significantly different between paired burned and unburned areas at individual landscape and when averaed for both sites (Table 5.2).

However, prescribed burnedsite hadresulted inminimal increase (5 Mg ha$^{-1}$;6.4%) atthe upland site, whereas a marginal decrease (-3.69 Mg ha$^{-1}$; 4.9%)in the bottomlands in terms of carbon poolsas compared to the adjacent unburned sites (Table 5.2).

Averaged across the two landscape sites,burned sites increased in SOC and TN stocks at an average of 4.37 Mg ha$^{-1}$(10.97% increase) and 1.19 Mg ha$^{-1}$(23.29% increase), respectively as compared to the adjacent unburned in the top 0-30 cm depth , although the differences were not significant (Table 5.2).

In contrast, unburned sites had an increase of 5.4 Mg ha$^{-1}$ (17.25% increase) biomasscarbon stocks (pooled woody and herbaceous) as compared to the burned sites whenaveraged for both sites, although the difference was not significant (Table 5.2).

Table 5.2. Mean values of above-ground biomass carbon, SOC and TN stock (0-30 cm) and carbon pooled in burned and the adjacent unburned sites at two landscapes and averaged for all sites

| | Dikale site (upland , ≥1500 m.a.s.l. | | Sanke site (bottomland, ≤1400 m.a.s.l. | | Averaged for all sites | |
|---|---|---|---|---|---|---|
| Respons variables | Burned | Unbured (adjacent) | Burned | Unbured (adjacent) | Burned | Unbured (adjacent) |
| Tota AGBC(Mg ha$^{-1}$) (Woody and herbaceous) | 34.72±6.67 | 34.76±4.24 | 31.30±3.72 | 38.70±5.06 | 33.01±3.78 | 36.73±3.28 |
| SOC stock (Mg ha$^{-1}$, 0-30 cm ) | 48.19±2.29 | 43.17±2.40 | 40.25±2.57 | 36.54±1.94 | 44.22±1.78 | 39.85±1.59 |
| Carbon pools (Mg ha$^{-1}$) ( in soil and vegetation biomass ) | 82.92±8.26 | 77.93±4.38 | 71.55±4.52 | 75.24±5.34 | 77.23±4.73 | 76.58±3.43 |
| TN stock (Mg ha$^{-1}$, 0-30 cm) | 6.97±0.45$^{a}$ | 4.65±0.16$^{b}$ | 5.62±0.27 | 5.56±0.29 | 6.30±0.28$^{a}$ | 5.11±0.28$^{b}$ |

Note: Mean values (± SE) followed by different letters showed significant between burned and the adjacent unburned sites, using Tukey's test (adjusted) at $P < 0.05$).

## 5.4. DISCUSSIONS

### *5.4.1. Effect of fire on bulk desnity, SOC and TN contents and stocks*

A slight decrease of soil bulk density as a result of prescribed fire observed in this study could be du to adequate grass growth to build root reserves, establish good basal cover against compaction erosion and relatedly higher amount of SOC under burned areas.

In agreement with, Brye (2006) reported decreasing bulk density values after 12 years of annual prescribed fires in a humid-subtropical environment in central Arkansas, because of increased belowground organic matter inputs as root biomass. Other study (e.g., Pierson et al., 2008) reported no significant changes in soil bulk density after prescribed fires.

However, many factors seem to determine soil bulk density (including, fire intensity and severity, fuel accumulation, texture, moisture) and these need to be carefully monitored if we hope to gain a greater knowledge of this soil property.

Our results indicate thatSOC content and stock were not significantly different between burned and unburned pairs, despite differences across soil depths and landscapes. However, significantly higher SOC content in burned than in the unburned aresa at the 15-30 cm depth of the upland site was unlikely that fire affected SOC concentration in lower soil layer without causing significant impact on on the upper surface (0-5 cm)soil layer. Hence, the difference was a result someother soil processes (e.g. root exudates).

In general, a relative increase in SOC content and stock under theburned plots as compared to the adjacent unburned sites could be attributed to the recovery of herbaceous biomass and more litter input following post-fire that may imply the buildup of organic matter (carbon storage) in the soil system

Inline with, Scharenbroch et al. (2012) indicated that in low intensity fires, it is normal to find increased carbon pool values because of the incorporation of unburned or partially unburned slash fragments into the soil of because the incomplete combustion of the organic matter.

Several other studies (Neill et al., 2007; Lavoie et al., 2010; Roaldson et al., 2014) have also reported no change or changes that were not statistically significant difference between burned and unburned sites in terms soil organic carbon.

In contrast, Bird et al. (2000) reported decrease in soil C with all fire frequencies compared to no burn in African savanna. However, in many other studies (e.g. Kolka et al., 2014; Maynard et al., 2014) have indicated that no effect or little effect of fire on SOC and TN pools. Others (e.g.Nave et al., 2011) in their review of literatures have also found in most of the cases that prescribed fire did not have significant effect on soil carbon. Inconstrast, Certini (2005) indicated that there is a marked loss in soil carbon as a consequence of high-intensity fires.

In this study, we observed apparently more SOC stocks (15.62 and 15.32% increases) in the upland at 0–5 and 15-30 cm, whereas an increase of 11.5 and 16.8% (in bottomland) in SOC stocks at the0-5 and 5-15 cm soil depths, respectively as compared to their corresponding adjacent unburned (controls).

This observation showed that soil organic carbon to prescribed fire is influenced across soil depths and landscape positions may be due to other factors such grazing management and post fire management condition which may have confounding interaction with fire.

Others (Knicker, 2007; Boerner et al., 2009) also indicated that the balance between carbon uptake and loss is a reflection of the effect of vegetation composition and structure. Ansley et al. (2006) also indicated that fire had the potential to alter soil carbon storage by influencing the rates of net primary productivity and carbon allocation patterns.

In this study, our results showed that prescribed fire (7 years of post-fire) increased the SOC sequestration rate estimated 1.13 Mg ha$^{-1}$yr$^{-1}$ in the upland sites, 0.95 Mg ha$^{-1}$ yr$^{-1)}$in bottomland sites as compared to their respective 40 years of fire suppression in the system

Similar to our results, Richards et al. (2011) reported that SOC stocks measured in the field after 5 years of annual, 3 year and unburned fire treatments ranged from 41–58 t $ha^{-1}$ for tropical savannas of northern Australia, although there were not significant different. The same authors also suggested that changes in fire management will take up to 100 years to have a detectable impact on SOC stocks.

Often, there is an initial loss of soil organic carbon after fire treatments, due to the combustion of the organic matter fraction in the surface layer and thereafter soil organic carbon recuperation, with the decomposition of roots of dead or burnt-off plants, which overtime attains a balance with the unburnt plot (Oluwole et al., 2008).

Other (Tiedemann et al., 2000) also indicated that fire mainly causes changes in soil and vegetation characteristics depending on fire intensity, and substantial consumption of organic matter at high temperature. In this study, planned (prescribed) fire was applied under moderate intensity fire with surface soil temperature was medium around 27°C (LaMalfa et al., 2008). Consequently, the probability of losing large amounts of soil carbon to fire in this ecosystem is small.

Others (Alexis et al., 2012; Muqaddas et al., 2015) indicated that fire can also lead to a loss soil carbon and nitrogen when soil temperatures exceed 200 °C owing to fuel combustion, but normally the temperatures reached in prescribed fires do not result in a significant temperature increase 2 to 3 cm below the soil surface.

Moreover, others (Knicker, 2007; Bennet et al., 2014) have also suggested that the effect of fire on SOC storage depends on fire intensity, vegetation type, and fuel load, soil texture and slope.Although, responses of TN content and stock to fire effects followed similar trends with that of SOC, but the TN content and stocks were found significantly higher in burned only in the uplands, whilst decreased in the burned treatment at the bottomlands.

Similarly, Muqaddas et al. (2015) and Scharenbroch et al. (2012) reported that total nitrogen content in soils burned recurrently is greater than that in soils at unburned sites. However, the same authors also reported a marked loss of total nitrogen in more frequently burnt sites (every 2 years) and conclude that less frequent burning (every 4 years) does not affect the N pools.

Oljima et al. (1994) also suggested that the effect of burning may not entirely result in the reduction and/or enhancement of soil total nitrogen. Coetsee et al.(2010) also argued that at a landscape level, fire-induced changes in tree populations will affect the dynamics of carbon and nitrogen in the soil.

In this study, the effect of fire did not bring significant differences in terms of C:N ratio, although results some variation along the landscape and soil depths. The values that ranged from 8.53 to 9.40 and 6.76 to 7.16 in the burned at the upland and bottom lands, respectively and from 6.58 to 9.74 in their adjacent unburned sites is generally low which indicates relatively low N content and high organic carbon (Kolka et al. , 2014).

### *5.4.2. Effect of fire on carbon in above-ground vegetation biomass*

This study shows that unburned areas contain a relatively higher woody tree biomass carbon than burned areas; though the difference was not significant as expected in our hypothesis, yet, the results showed that more accumulation of carbon stocks in the woody biomass by 5.81 Mg ha$^{-1}$(16.3%) and 8.54Mg ha$^{-1}$(22.4%), respectively in unburned of the upland and bottomland areas as compared to their corresponding burned areas.

The observed woody plant biomass carbon reduction with burning may be attributed, for example, burned areas are attractive to a variety of herbivores, thereby periods intensive grazing and browsing on new coppicing growth after burn for some species following fire as we also observed during field data collection.

In contrast to our present findings, Tilman et al. (2000) reported that fire suppression caused nearly twice as much C to accumulate in aboveground biomass than in stands that had a moderate-to-high fire frequency. According to Bond (2008), savannas would accumulate considerable biomass and carbon stocks at the expense of herbaceous and grass plants.

Other studies (Higgins et al., 2007; Grace et al., 2006; Murphy et al., 2010) also detected a consistently negative relationship between fire frequency and biomass of savanna woody plant biomass due to reduction in plant growth. On the other hand, the results of this study indicated that herbaceous biomass and carbon was significantly higher by 3.7 and 2.8 Mg ha$^{-1}$ in the burned sites of the upland and bottomland, respectively than their adjacent unburned areas.

Thus, the increase in herbaceous biomass in the burned areas makes an important contribution to the total aboveground carbon stocks, but does not make up for the total above-ground biomass difference between each paired fire treatment across the sites. Similarly, Coetsee et al. (2010) also suggested that at a landscape level, fire-induced changes in tree populations will affect the dynamics of carbon and nitrogen in the soil

In summary, unburnt areas have relatively higher woody plant biomass at the expense of herbaceous vegetation biomass while burnt areas have significantly higher herbaceous plant biomass and are consistent with many published results in tropical savannas (Keeley et al., 2003; Murphy et al., 2010; Cook et al., 2015) suggested that herbaceous plants mostly have their survival buds at or below the surface of the soil and immediately regenerate after fire.

However, the carbon stocks in the above-ground biomass in savanna ecosystems varied widely between 1.8 Mg ha$^{-1}$ at sites where trees are scarce to over 30 Mg ha$^{-1}$ at places with dense tree cover (Grace et al., 2006).

### *5.4.3. Effect of fire on carbon pools (soil and vegetation)*

The carbon pools (soil and vegetation biomass) under prescribed fire in this study showed an increase in the uplands and decrease in the bottomlands of burned sites as compared to the adjacent unburned controls. These values can be equated to carbon sequestration rate of 0.71 Mg ha$^{-1}$yr$^{-1}$ in the upland and by 0.53 Mg ha$^{-1}$ yr$^{-1}$ reduction at the bottomland (taking 7 years-post fire).

The observed inconsistent results in the values pooled carbon stocks are a reflection of the each component of carbon pools (soil and vegetation) in response to fire across the landscape sites. However, Oljima et al. (1994) stated that ecosystem responses to fire involve a complex set of interactions that begin with the removal of above-ground plant biomass, resulting in reduced inputs of C and N into the soil system.

Our estimation of the carbon pools showed that SOC stock accounted for 57.3% in burned plots across both landscape sites is inline with the findings of Ryan et al. (2011), whoreported that soil carbon shared 70% of the total carbon stocks in a study from Miombo woodland landscape of Mozambique.

However, the absence of significant difference between burned and unburned areas in the in terms of carbon pool (soil and vegetation biomass) in present study is also supported by Angassa and Oba (2008), who suggested that the effect of fire might depend on the amount of available fuel load (herbaceous biomass) at the time of burning, whereas this in turn affected by the conditions of rainfall and grazing intensity

Furthermore, the direct fire impacts cannot always be isolated from the effect of post fire management to determine the vegetation structure and thereby on the rate of carbon changes in above-ground biomass. A study by Piñeiro et al. (2010) also indicated that grazing and fire could modify the structure and function of ecosystems, affecting SOC storage.

## 5.5. CONCLUSIONS

The results of this study showed that relatevly more SOC and TN contents and stocks as well as herbaceous biomass carbon were recorded under the burned plots as compared to unburned area. Furhtermor, the mangnitude of differences between burned and unburned areas in these parameters were more apparent in the upland areas.

The herbaceous biomass carbon in burned areas was signficalty higher by 2.29 Mg ha$^{-1}$ than that of the adjacent unburned sites, but there was a decline in the woody biomass carbon by 6.01 Mg ha$^{-1}$ in burned sites as compared to the control (adjacent unburned sites)when averaged for all sites.

In general, prescribed fire increased soil carbon storage and maintain soil nutrient and increase grass production whereas long-term ban of fire resulted in increaseof carbon stocks in the woody vegetation, although some discrepancies across the landscapes and even within the same landscape at each soil depths.

The study, however, suggested further investigation to look into the interactive effects of fire, grazing and climate change on carbon sequestration and nitrogen dynamics.

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# CHAPTER SIX

This chapter gives the summary of key findingsof the thesis work and general conclusions and recommendation suggested for future research needs.

# 6. SUMMARY AND CONCLUSIONS

## General summary

In this study, we haveinvestigated the effects of grazing management (enclosure versus adjacent open grazed areas) on selected soil properties, more spefcially SOC and TN contents and socks and key vegetation attributes in a case study (Did-hara area) of semi-arid Borana pastoral ecosystems by considering the the roles of enclosure age chronosequence ranged from15–37 years old and three soil depths.

Overall, total mean SOC stocks of 39.6 ± 3.5 Mg ha$^{-1}$in younger (< 20 years old), 40.8 ± 3.4 Mg ha$^{-1}$ in the medium (20–30 years old) and 51.0 ± 4.4 Mg ha$^{-1}$ in the older (> 30 years old) age categories of the enclosures, where as the values were ranged from 34.4 ± 2.5 to 47.9 ± 5.1 Mg ha$^{-1}$ inadjacent open-grazed areas wthin 0-30 cm soil depth.

A significant improvement in terms of herbaceous biomass and vegetation recovery (grass, herbs and forbs) were recoreded under enclosure management than in the open grazed areas. However, enclosure age did not influence various vegetation attributes investigated in this study.

The medium age (20-30 years old) enclosures accumulated relatevly moreSOC stocks than adjacent open-grazed areas, whereas older age (>30 years old) enclosures had higher TN stock than adjacent open-grazed areas.

The study concludes that enclosure management increased the SOC and TN stocks and herbaceous biomass in dry savanna ecosystems, the case in our study area,although more research will be required to determine the impact of enclosure management on storage and the viability of carbon markets to provide as livelihood diversification option to the local people.

Based on the LULC analysis of Landsat data of the years 1976, 1986, and 2013, we found that the LULC change trends varied significantly during the abovementioned periods.

The results showed that a considerable decrease in woodland cover from12.6% in 1976 to 5.5% in 2013, grazingland (the communal rangelands) from 53.9% in 1976 to 36.1% in 2013, whereas bushland cover increased 4.8% in 1976 17.2 in 2013 (more than fourfold) and the size of cultivated land increased 2.0% in 1976 to 21.3% 2013 (more than tenfold, and this trend also continued during the period between 1986 and 2013.

In case of enclosure, the area coverage showed some discrepancy that is decreased in its area size during the first period by 22.8% in 1976 to 8.8% in 1986, but showed increased from 8.8% in 1986 to 16.2% in 2013. During the same periods, majorities of the grazingland (the communal rangeland) were converted into bushland and cropland and followed by conversion of woodland to cropland.

In this study, our results showed the mean values SOC stocks in woodland was 55.94±3.41 Mg ha$^{-1,}$ while for enclosure, grazingland and cultivated land were 50.03±3.03 ha-1, 45.79±4.00 ha$^{-1}$ and 38.10±2.39 Mg ha$^{-1}$, respectively within 0-30 cm soil depth. Further, woodland had the highest (7.52±0.43 Mg ha$^{-1}$), while cultivated land was the lowest (5.58±0.35 Mg ha$^{-1}$) in terms of TN stock across the 0- 30 cm soil depth range.

Over the last 37 years (1976-2013) the potential losses of 319 Gg and 30.65 Gg soil organic carbon and total nitrogen stocks, respectively, mainly from conversion of woodland and grazing land to cultivated lands resulted in higher loss of SOC and TN stocks, where as the conversion of grazing and croplands to woodland and enclosure partially offset the SOC and TN losses

To this effect, restoration of degraded rangelands and protection of natural forest (woodland) and enclosure can also enhance carbon sequestration Therefore; development of sustainable land use practices in the dry savannas, the case in our study area, needs to be improved.

Howeve that can address variability in carbon flux over long time in fully understanding the carbon dynamics for future climate change matigation carbon is imperative, as this is one of the limitations of this study.

In response to the growing interest in estimating above-groun biomass andcarbon stocks in woody vegetation, allometricequations have been developed for selected key woody species common in the semi-arid savanna ecosystems of southern Ethiopia

For this, easilymeasurable predictor variables (woody dendrometric)including the diameter at stump height (DSH at 30 cm), diameter at breast height (DBH at 1.30 m), tree height (TH) and crown diameters used to develop the species specific equations presented here.Additionally, mixed (generalized) allometric equations were aslo developed for group of woody tree species.

The developed species-specific and mixedspecies allometric equations will help to accurately estimate and monitor biomassfor selected key woody species common in the semi-arid savanna ecosystems of southern Ethiopia.

The decision to choose the best predictive models was based on the adj.$R^2$, MSE and AIC and the best predictive model for each species over other biometric that included DSH, DBH, CV variables and their combinations. The mixed-species regression models with three predicators (DSH-TH-CA) provided the best fitted to estimate all AGB with adj.$R^2$>0.80.

The study concludes that the developed allometric equation models of selected woody species provided in this study might be helpful toward the generation of more accurate estimations of the above-ground biomass and to estimate the carbon accured in thw woody vegetation biomass in the arid and semi-arid ecosystems.

Furthermore, allometric equations provide opportunities for further work on comparing biomass production with native stands. However, care should be taken when applying the allometric models developed here to trees with structural characteristics outside of the range from which these models were developed.

The studyassessed the impact of long-term ban of fire and recently introduced prescribed on SOC and TN contents and stocks and above-ground biomass carbon accumulation across two different landscapes (Dikale (uplandscape) and Sanke (bottomlandscape)in the Borana rangelands of southern Ethiopia

The results from our two sites showed soil organic carbon (SOC) increased in the burned (7 years post-fire) soils in both landscapes although no significant difference as compared to that of the unburnt for about 40 years.

Overall,total mean SOC stocks of 48.19 and 43.17 and Mg ha$^{-1}$ in burned sites inthe upland and bottom landscapes, respectively, and the same values of 40.25 and 36.54 Mg ha$^{-1}$ in the corresponding adjacent unburnedareas were recorded to 30 cm depth this study.

Compared to the unburned sites, the herbaceous biomass carhon in the burned sites was higher by 2.29 Mg ha$^{-1}$as compared tothat of the unburned sites. In contrast, there was a decline in carbon stocks of thewoody vegetation biomass in burned sites by 6.01 Mg ha$^{-1}$as compared to adjacent unburned areas.

The study can conclude that prescribed fire increased soil carbon storage and maintain soil nutrient and increase herbaceous biomass, whereas long-term of fire suppression from the system increased carbon stocks in the woody vegetation. However, further investigation is required on the interactive effects of fire with grazing and climatic factors and inherent soil properties oncarbon and nitrogen dynamics.

## General Conclusions and recommendations

The data generated from the current study can inform policy makers and local land managers about the role of enclosure management in forage preservation, potential for carbon sequestration, livelihood diversification options for climate change adaptation.

There is an increase in ecological changes in the study area over the last decades whereby changes in land use systems significantly influenced SOC and TN contents and stocks.

The land use/land cover changes, particularly conversion of  woodland and the communal rangelands to cultivated land, resulted  in the highest lost of SOC and TN stocks , whereas the conversion of communal grazingland and cropland to woodland and enclosures partially counter balanced SOC and TN losses.

The allometric equation models developed for selected woody species are helpful for the generation of more accurate estimations of the aboveground biomass and the amount of carbon that can be sequestered in woody vegetation of the savanna rangelands of East Africa.

Prescribed fire relatively increased SOCand TN stocks and herbaceous biomass, which can help to maintain soil nutrients in the systems, whereas long-term exclusion of fire increased carbon stocks in the woody vegetation. The impact of prescribed fire management on carbon dynamics is a global challenge and multi-continent savanna biome initiative

Overall, this study will contribute to the existing knowledge gaps in terms of the potentials of SOC and TN stocks related to different rangeland management practices as well as a method to estimate the above-ground woody biomass in arid and semi-arid ecosystems of southern Ethiopia;

However, we remain cautious with respect to the conclusiveness of these findings given the paucity of information, most importantly the absence of soil data prior to the present land use systems.

Thus, it is suggested that furtherstudies includingother variables suchas climatic factors, seasonality and inherent soil properties across wider landscapes, which may have confounding effects on the dynamics of carbon sequestration other than the current land use and management practices to inform policy makers for the sustainable use of the savanna rangelands of southern Ethiopia

# APPENDIX

## Appendix 3.1. Description of land use /land cover classes and Landsat classification accuracies (%) for the 1976, 1986, and 2013 images, Yabello district, southern Ethiopia

| | | Accuracy assessment (%) | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | 1976 | | | 1986 | | | 2013 | | |
| LULC classes | Description | UA | PA | K | UA | PA | K | UA | PA | K |
| WL | Land unit covered with woody vegetation of different sizes (ranges from 5-20 m height) and mixed with bushes, and shrubs and grasses. | 90.91 | 90.91 | 0.89 | 92.30 | 85.00 | 1.00 | 80.00 | 88.89 | 0.78 |
| BuL | Land area composed of bushes and shrubs (<5 m height).Invasive woody plant species are dominantly distributed and not easily accessible for livestock to graze under it. | 83.33 | 80.00 | 0.80 | 85.00 | 87.50 | 1.00 | 92.86 | 92.86 | 0.91 |
| GL | Vast grazingland (communal) continuously grazed throughout the year and lacks rest for plant species to recover characterized by scattered trees or shrubs (<10 % of cover), and over-grazing is e a widespread observable fact | 89.66 | 81.25 | 0.86 | 90.00 | 83.72 | 0.84 | 92.00 | 95.00 | 0.90 |
| GR | Part of communal grazing lands protected from continuous (year round) grazing to conserve pasture for calves and other homestead herds for dry season grazing with good regeneration of grass and herbaceous and scattered trees and shrubs. | 83.56 | 91.04 | 0.83 | 88.57 | 88.57 | 0.84 | 96.43 | 90.00 | 0.95 |
| CL | Land unit under cropping, and also land recently cleared particularly in the flat and valley bottom for cultivation. It includes settlements. | 80.00 | 75.00 | 1.00 | 76.00 | 95.00 | 0.71 | 90.48 | 79.17 | 0.88 |
| BAL | This describes the land left without vegetation coversuch as gullies and degraded soils either due to human or natural factors and has become less worthy | 67.00 | 85.00 | 1.00 | 57.14 | 80.00 | 0.55 | 87.50 | 77.78 | 0.86 |
| **Overall** | | | | | | | | | | |
| Accuracy | | 85.00 | | | 85.83 | | | 91.67 | | |
| Kappa coefficient | | 0.74 | | | 0.81 | | | 0.90 | | |

WL -woodland, BuL-bushland, GL-grazingland (communal rangelands l); GR- enclosure, CL-cultivated land, BaL-bareland, UA -users' accuracy measures error of including (commissioning) a given image pixel into a land-cover class which it doesn't naturally belong to, PA-Producer's accuracy measures the error of exclusion (omission) of a given image pixel from its correct land-cover class during classification,(K-kappa coefficient (dimensionless) (Congalton and Green, 2009).

Appendix 4.1. Summary of mean and ranges (min-max) dendrometric variables and above-ground dry biomass components (stem, branches and total) of ten dominant woody species common to savanna rangelands of southern Ethiopia.

| Woody species | Local name | Growth forms | | DSH (cm) | DBH (cm) | TH (m) | CA ($m^2$) | CV ($m^3$) | Ws (kg) | Wb (kg) | Wt (kg) |
|---|---|---|---|---|---|---|---|---|---|---|---|
| *Acacia brevispica* Harms | Hammareessa | T/S | Mean | 11.14±0.9 | 9.77±0.91 | 3.24±0.20 | 5.61 ±0.95 | 13.0 2±2.73 | 1.19 ±0.23 | 5.94±1.01 | 7.14±1.19 |
| | | | Range | 4.10-16.50 | 3.00-14.60 | 1.70-5.10 | 0.24-11.64 | 0.29-39.25 | 0.00-3.51 | 2.76-14.35 | 3.21-17.70 |
| *Acacia bussei* Harms ex Sjostedt | Halloo | T | Mean | 7.85 ±1.01 | 6.25±0.66 | 4.65 ±0.30 | 14.19±3.43 | 51.04 ±17.53 | 5.51 ±1.72 | 13.71±5.35 | 19.21±6.89 |
| | | | Range | 3.30-17.40 | 3.10-10.85 | 2.80-7.60 | 3.14-55.84 | 7.70-282.93 | 0.56-24.00 | 2.06-83.86 | 2.91-107.85 |
| *Acacia etbaica* Schweinf. | Hallaqabeessa | T | Mean | 7.81±0.72 | 6.47±0.67 | 3.70±0.23 | 10.68±2.21 | 29.29±8.00 | 4.12±1.01 | 9.49±2.37 | 13.61±2.93 |
| | | | Range | 3.50-13.70 | 3.00-13.00 | 1.30-5.40 | 1.88-34.21 | 1.63-123.16 | 0.43-14.20 | 1.07-27.96 | 1.65-38.98 |
| *Acacia nilotica* (L.) Willd. ex Del. | Burquqqee | T | Mean | 9.63±0.76 | 8.22±0.76 | 5.027±0.25 | 13.15±1.98 | 92.45±15.91 | 8.18±1.58 | 21.29±4.73 | 29.46±5.69 |
| | | | Range | 4.35-13.70 | 3.00-12.60 | 3.50-6.60 | 2.12-32.28 | 5.94-124.82 | 1.84-18.78 | 1.82-75.63 | 3.67-85.20 |
| *Acacia seyal* Del. | Waacu (diimaa) | T | Mean | 8.89±0.98 | 7.5±0.82 | 4.36±0.42 | 21.85±3.25 | 73.62±13.23 | 8.88±3.44 | 12.91±3.26 | 21.79±5.60 |
| | | | Range | 3.20-15.30 | 2.70-12.20 | 1.20-6.50 | 2.97-42.41 | 2.38-158.34 | 0.00-54.10 | 0.62-47.85 | 1.11-70.44 |
| *Acacia tortilis* (Forssk.) Hayne | Dhaddacha | T | Mean | 10.54±1.20 | 9.27±1.16 | 3.86±0.21 | 13.56±2.30 | 38.71±8.39 | 10.66±2.27 | 21.19±5.59 | 31.85±8.10 |
| | | | Range | 4.40-21.30 | 0.00-20.20 | 0.21-5.30 | 2.95-31.79 | 5.89-112.31 | 1.59-33.98 | 1.99-68.99 | 3.67-102.97 |
| *Commiphora africana* (A. Rich.) Engl. | Hammeessa dhiiroo | T/S | Mean | 4.33±0.30 | 3.55±0.21 | 6.03±0.68 | 6.95±1.57 | 32.74±8.39 | 4.14±0.81 | 6.77±1.55 | 10.92±2.32 |
| | | | Range | 3.10-7.10 | 2.40-5.00 | 2.40-11.60 | 0.71-21.43 | 1.51-89.99 | 0.16-9.88 | 0.33-19.57 | 0.48-28.30 |
| *Lannea rivae* (Chiov.) Sacleux | Handaraka | T/S | Mean | 9.37±0.42 | 7.31±0.60 | 2.80±0.13 | 4.69±0.59 | 8.81±1.15 | 3.48±0.71 | 5.78±0.93 | 9.26±1.62 |
| | | | Range | 7.10-13.00 | 1.30-11.60 | 2.00-3.60 | 1.01-9.02 | 1.88-17.14 | 1.57-12.77 | 2.13-16.14 | 4.09-28.91 |
| *Rhus natalensis* Berah. ex Krauss | Daboobessa diidaa | T/S | Mean | Nm | Nm | 2.32±0.10 | 10.13±1.38 | 16.16±2.90 | 2.82±0.46 | 8.93±1.41 | 11.75±1.83 |
| | | | Range | Nm | Nm | 0.20-3.10 | 1.38-24.85 | 1.09-51.35 | 0.63-8.21 | 1.76-25.12 | 2.39-33.34 |
| *Acacia drepanolobium* Harms ex Sjostedt | Fuleensa | T | Mean | 7.19±0.71 | 5.96±0.72 | 3.89±0.40 | 5.62±0.99 | 17.70±4.56 | 5.10±1.93 | 9.84±3.70 | 14.94±5.61 |
| | | | Range | 3.00-14.20 | 2.00-13.20 | 2.00-7.70 | 0.92-13.62 | 1.23-69.91 | 0.25-30.63 | 0.48-57.96 | 0.73-88.59 |

Note: DSH -Diameter at stump height 30 cm , DBH -Diameter at breast height, TH -Tree height, CA -crown area, CV-crown volume, nm -not measured, Wt- total dry biomass, Ws-stems dry biomass), Wb-branches dry biomass. Growth forms: T- Tree, S - shrub, T/S -Tree/Shrub .Values are mean ((±SE) (n= 15 individual plants for each species i.e., 15x 10=150) destructively harvested and used in the allometric equations. Values are mean ((±SE) (n= 15 individual plants for each species i.e., 15x 10=150) destructively harvested and used in the allometric equations.

Appendix 4.2.Developing allometric equations for the prediction of t above-ground biomass (AGB) of total stem and branches biomass components along with goodness of fit tests for each ten woody species

| | | | Regression parameter estimates | | | | Model performance selection criteria | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Woody species | Biomass component | Model equation expression | ln (a) (SE) | b (SE) | c (SE) | d(SE) | Adj. $R^2$ | P-value | AIC | ∆AIC | MSE | CF | PRESS | VIF |
| Acacia bussei | Total | ln (y) = a+ b × ln (DBH) | -1.756 (0.427) | 2.366 (0.238) | | | 0.88 | <.0001 | -28.38 | 10.11 | 0.13 | 1.07 | 2.48 | |
| | | ln (y) = a+ b × ln (CV)* | -0.316 (0.309) | 1.164 (0.126) | | | 0.90 | <.0001 | -31.82 | 6.67 | 0.11 | 1.06 | 1.79 | |
| | | ln (y) = a+ b × ln (DBH))+ c × ln(CV)* | -1.569 (0.301) | 1.119 (0.360) | 0.581 (0.150) | | 0.94 | <.0001 | -38.49 | 0 | 0.06 | 1.03 | 1.37 | 4.8 |
| | Stem | ln (y) = a+ b × ln (DBH)* | -3.404 (0.548) | 2.563 (00305) | | | 0.83 | <.0001 | -20.89 | 1.15 | 0.22 | 1.12 | 3.53 | |
| | | ln (y) = a+ b × ln (CV) | -4.013 (1.417) | 3.379 (0.927)* | | | 0.79 | <.0001 | -17.39 | 4.65 | 0.28 | 1.15 | 4.6 | |
| | | ln (y) = a+ b × ln (DBH))+ c × ln(CV)* | -3.259 (0.520) | 1.629 (0.623) | 0.434 (0.259) | | 0.85 | <.0001 | -22.04 | 0 | 0.19 | 1.1 | 3.6 | 4.8 |
| | Branches | ln (y) = a+ b × ln (DBH)* | -1.993 (0.559) | 2.282 (0.311) | | | 0.79 | <.0001 | -20.33 | 4.74 | 0.23 | 1.12 | 4.13 | |
| | | ln (y) = a+ b × ln (CA) | -0.604 (0.394) | 1.122 (0.160) | | | 0.77 | <.0001 | -19.22 | 5.85 | 0.25 | 1.13 | 4.36 | |
| | | ln (y) = a+ b × ln (DBH)+ c × ln(CA) * | -1.650 (0.484) | 1.460 (0.515) | 0.348 (0.302) | | 0.86 | <.0001 | -25.07 | 0 | 0.16 | 1.08 | 3.64 | 4.16 |
| Acacia drepanolobium | Total | ln (y) = a+ b × ln (DSH)* | -4.12 (0.374) | 3.2.07 (0.193) | | | 0.95 | <.0001 | -36.516 | 6.354 | 0.077 | 1.04 | 1.33 | |
| | | ln (y) = a+ b × ln (CV) | -0.136 (0.2226) | 0.931 (0.087) | | | 0.89 | <.0001 | -24.28 | 18.59 | 0.175 | 1.09 | 3.32 | |
| | | ln (y) = a+ b × ln (DSH))+ c × ln(CV)* | -2.933 (0.491) | 2.182 (0.372) | 0.336 (0.112) | | 0.97 | <.0001 | -42.87 | 15.33 | 0.048 | 1.02 | 0.91 | 5.9 |
| | Stem | ln (y) = a+ b × ln (DSH)* | -4.988 (0.504) | 3.109 (0.260) | | | 0.91 | <.0001 | -27.54 | 5.25 | 0.141 | 1.07 | 2.27 | |
| | | ln (y) = a+ b × ln (CV) | -1.168 (0.226) | 0.919 (0.087) | | | 0.89 | <.0001 | -24.16 | 8.63 | 0.176 | 1.09 | 3.42 | |
| | | ln (y) = a+ b × ln (DSH))+ c × ln(CV)* | -3.484 (0.688) | 1.809 (0.521) | 0.425 (0.156) | | 0.94 | <.0001 | -32.79 | 0 | 0.094 | 1.05 | 1.75 | 5.93 |
| | Branches | ln (y) = a+ b × ln (DSH)* | -4.704 (0.406) | 3.280 (0.209) | | | 0.94 | <.0001 | -34.04 | 2.14 | 0.09 | 1.05 | 1.64 | |
| | | ln (y) = a+ b × ln (CA) | -0.238 (0.259) | 1.270 (0.155) | | | 0.82 | <.0001 | -16.43 | 19.75 | 0.29 | 1.16 | 5.26 | |
| | | ln (y) = a+ b × ln DSH) + c × ln (CA)* | -3.797( 0.589) | 2.557 (0.413) | 0.333 (0.171) | | 0..95 | <.0001 | -36.18 | 0 | 0.075 | 1.04 | 1.43 | 4.68 |
| Acacia etbaica | Total | ln (y) = a+ b × ln (DSH) | -0.195 (0.856) | 1.187 (0.466) | | | 0.36 | 0.02 | -7.69 | 13.32 | 0.5 | 1.28 | 10.64 | |
| | | ln (y) = a+ b × ln (CV) | -1.025 (0.509) | 0.454 (0.167) | | | 0.31 | 0.017 | -8.4 | 12.61 | 0.53 | 1.3 | 10.07 | |
| | | ln (y) = a+ b × ln (DSH))+ c × ln(CV)* | -0.985 (0.536) | 0.909 (0.395) | 0.546 (0.180) | | 0.75 | 0.01 | -21.01 | 0 | 0.11 | 1.06 | 11.28 | |

**Appendix 4.2. (Continuation)**

| | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Stem | ln (y) = a+ b × ln (DBH) | -1.873 (0.624) | 1.550 (0.340) | | | 0.59 | 0.001 | -17.16 | 2.87 | 0.28 | 1.15 | 5.59 | 1.33 |
| | | ln (y) = a+ b × ln (TH) | -0.729 (0.523s | 1.405 (0.428) | | | 0.41 | 0.006 | -11.89 | 8.14 | 0.4 | 1.22 | 11.03 | |
| | | ln (y) = a+ b ×ln (DBH)+ c× ln (TH)* | -2.124 (0.564)* | 1.181(0.347) | 0.786 (0.366) | | 0.68 | 0.001 | -20.03 | 0 | 0.22 | 1.12 | 9.02 | 1.33 |
| | Branches | ln (y) = a+ b × ln (DBH) | -1.071 (1.125) | 1.523 (0.555) | | | 0.32 | 0.02 | -4.58 | 20.82 | 0.65 | 1.38 | 12.32 | |
| | | ln (y) = a+ b × ln (CA) | -0.425 (0.956) | 0.747(0.621) | 0.499 (0.324) | | 0.29 | 0.02 | -3.95 | 21.45 | 0.68 | 1.4 | 12.24 | |
| | | ln (y) = a+ b × ln DSH) + c × ln (CA)* | -2.022 (0.550) | 0.914(0.384) | 0.680 (0.162) | | 0.84 | <.0001 | -25.4 | 0 | 0.15 | 1.08 | 10.54 | 1.48 |
| Acacia nilotica | Total | ln (y) = a+ b × ln (DBH)* | -0.834 (0.273) | 1.926 (0.132) | | | 0.94 | <.0001 | -42.6 | 0 | 0.05 | 1.03 | 0.87 | |
| | | ln (y) = a+ b × ln (CV) | -0.110(0.657) | 0.887(0.179) | | | 0.63 | 0.0003 | -15.6 | 27 | 0.31 | 1.17 | 5.32 | |
| | | ln (y) = a+ b × ln (DBH))+ c × ln(CV)* | -2.963 (0.533) | 2.191 (0.238) | 0.740 (0.447) | | 0.93 | <.0001 | -40.56 | 2.04 | 0.06 | 1.03 | 0.96 | 3.21 |
| | Stem | ln (y) = a+ b × ln (DSH) | -2.334 (0.744) | 1.878 (0.333) | | | 0.69 | <.0001 | -22.28 | 10.06 | 0.2 | 1.11 | 3.37 | |
| | | ln (y) = a+ b × ln (TH) | -3.393 (1.16) | 3.259 (0.721) | | | 0.58 | 0.0006 | -17.87 | 14.47 | 0.27 | 1.14 | 4.3 | |
| | | ln (y) = a+ b × ln (DSH) + c× ln (TH) d × ln (CV)* | -4.477 (0.740) | 1.951 (0.378) | 2.384 (0.627) | -0.511 (0.172) | 0.86 | <.0001 | -32.34 | 0 | 0.09 | 1.05 | 2.2 | |
| | Branches | ln (y) = a+ b × ln (DSH)* | -1.723 (0.317) | 2.171 (0.153) | | | 0.93 | <.0001 | -38.05 | 0 | 0.07 | 1.04 | 1.22 | |
| | | ln (y) = a+ b × ln (CV) | -1.125 (0.651) | 1.062 (0.178) | | | 0.71 | <.0001 | -15.85 | 22.2 | 0.31 | 1.17 | 11.14 | |
| | | ln (y) = a+ b × ln (DSH))+ c × ln(CV) | -3.230 (0.466) | 2.096 (0.332) | 0.354 (0.143) | | 0.93 | <.0001 | -35.47 | 2.58 | 0.08 | 1.04 | 1.44 | 2.6 |
| Acacia tortilis | Total | ln (y) = a+ b × ln (DBH) | -1.748 (0.434) | 2.227(0.201) | | | 0.90 | <.0001 | -29.95 | 10.4 | 0.12 | 1.06 | 8.98 | |
| | | ln (y) = a+ b × ln (TH) | -3.653 (0.778) | 4.962 (0.578) | | | 0.84 | <.0001 | -23.21 | 17.14 | 0.19 | 1.1 | 10.91 | |
| | | ln (y) = a+ b ×ln (DBH)+ c× ln (TH)* | -3.043 (0.444) | 1.411 (0.252) | 2.266 (0.581) | | 0.95 | <.0001 | -40.35 | 0 | 0.06 | 1.03 | 6.17 | 3.29 |
| | Stem | ln (y) = a+ b × ln (DBH) | -2.626 (0.622) | 2.119 (0.288) | | | 0.79 | <.0001 | -19.15 | 22.42 | 0.25 | 1.13 | 22.41 | |
| | | ln (y) = a+ b × ln (TH)* | -5.140 (0.509) | 5.248 (0.378) | | | 0.93 | <.0001 | -35.95 | 5.62 | 0.08 | 1.04 | 11.3 | |
| | | ln (y) = a+ b ×ln (DBH)+ c× ln (TH)* | -4.837 (0.426) | 0.710 (0.252) | 3.897 (0.558) | | 0.96 | <.0001 | -41.57 | 0 | 0.05 | 1.03 | 9.23 | 3.29 |

**Appendix 4.2 (continuation)**

| | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Branches | ln (y) = a+ b × ln (DBH)* | -2.285 (0.409) | 2.280 (0.189) | | 0.91 | <.0001 | -31.77 | 2.93 | 0.11 | 1.06 | 9.32 | |
| | | ln (y) = a+ b × ln (TH) | -3.911 (0.953) | 4.836 (0.707) | | 0.77 | <.0001 | -17.14 | 17.56 | 0.28 | 1.15 | 15.33 | |
| | | ln (y) = a+ b ×ln (DBH)+ c× ln (TH)* | -3.148 (0.536) | 1.741 (0.252) | 1.503 (0.701) | 0.93 | <.0001 | -34.7 | 0 | 0.08 | 1.04 | 8.88 | 3.29 |
| Acacia seyal | Total | ln (y) = a+ b × ln (DSH)* | -2.655 (0.234) | 2.425(0.110) | | 0.97 | <.0001 | -45.33 | 3.91 | 0.04 | 1.02 | 6.57 | |
| | | ln (y) = a+ b × ln (CV) | -1.138 (0.364) | 0.923 (0.091) | | 0.88 | <.0001 | -23.44 | 25.8 | 0.19 | 1.1 | 13.04 | |
| | | ln (y) = a+ b × ln (DSH))+ c × ln(CV)* | -2.444 (0.221) | 1.913 (0.236) | 0.224 (0.094) | 0.98 | <.0001 | -49.24 | 0 | 0.03 | 1.02 | 5.01 | 6.38 |
| | Stem | ln (y) = a+ b × ln (DSH)* | -3.145(0.256) | 2.146(0.120) | | 0.96 | <.0001 | -42.67 | 0.03 | 0.05 | 1.03 | 14.52 | |
| | | ln (y) = a+ b × ln (TH) | -1.574 (0.293) | 2.100 (0.201) | | 0.88 | <.0001 | -27.58 | 15.12 | 0.14 | 1.07 | 20.1 | |
| | | ln (y) = a+ b × ln (DSH) +c × ln (TH)* | -2.900 (0.316) | 1.719 (0.343) | 0.466 (0.348) | 0.96 | <.0001 | -42.70 | 0.00 | 0.05 | 1.03 | 13.22 | 8.55 |
| | Branches | ln (y) = a+ b × ln (DSH) | -3.407 (0.285) | 2.577 ( 0.133) | | 0.96 | <.0001 | -39.48 | 5.29 | 0.06 | 1.03 | 9.82 | |
| | | ln (y) = a+ b × ln (CA) | -2.329 (0.561) | 1.503 (0.190) | | 0.81 | <.0001 | -14.96 | 29.81 | 0.33 | 1.18 | 21 | |
| | | ln (y) = a+ b × ln DSH) + c × ln (CA)* | -3.406 (0.233) | 2.059 (0.220) | 0.378 (0.139) | 0.98 | <.0001 | -44.77 | 0 | 0.04 | 1.02 | 11.15 | 4.01 |
| Lannea rivae | Total | ln (y) = a+ b × ln (DSH) | -2.999 (1.320) | 2.285 (0.592) | | 0.50 | 0.002 | -28.08 | 13.75 | 0.14 | 1.07 | 2.87 | |
| | | ln (y) = a+ b × ln (CV) | -1.162 (0.262) | 0.650 (0.173) | | 0.43 | 0.005 | -26.22 | 15.61 | 0.154 | 1.08 | 2.57 | |
| | | ln (y) = a+ b × ln (DSH))+ c × ln(CV)* | -3.148 (0.8111) | 1.925 (0.371) | 0.469 (0.099) | 0.81 | <.0001 | -41.83 | 0 | 0.05 | 1.03 | 1.29 | 1.08 |
| | Stem | ln (y) = a+ b × ln (DSH) | -4.105 (1.343) | 2.333 (0.603) | | 0.50 | 0.002 | -27.55 | 6.54 | 0.14 | 1.07 | 3.39 | |
| | | ln (y) = a+ b × ln (CV) | -0.067 (0.402) | 0.502 (0.190) | | 0.30 | 0.021 | -22.47 | 11.62 | 0.2 | 1.11 | 3.49 | |
| | | ln (y) = a+ b × ln (DSH))+ c × ln(CV)* | -4.225 (1.053) | 2.032 (0.482) | 0.390 (0.129) | 0.70 | 0.000 | -34.09 | 0 | 0.09 | 1.05 | 2.97 | |
| | Branches | ln (y) = a+ b × ln (DSH) | -3.475 (1.487) | 2.285 (0.667) | | 0.43 | 0.005 | -24.51 | 14.07 | 0.17 | 1.09 | 3.24 | |
| | | ln (y) = a+ b × ln (CV) | -0.561 (0.255) | 0.738 (01.68) | | 0.57 | 0.002 | -28.54 | 10.04 | 0.13 | 1.07 | 2.18 | |
| | | ln (y) = a+ b × ln (DSH))+ c × ln(CV)* | -2.781(1.039) | 1.698(0.420) | -0.601(0.390) | 0.8 | <0.0001 | -38.58 | 0 | 0.06 | 1.07 | 2.87 | 1.08l |

**Appendix 4.2. (Continuation)**

| | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Rhus natalensis | Total | ln (y) = a+ b × ln (CV)* | 1.129 (0.282) | 0.502 (0.109) | | | 0.59 | 0.001 | -26.65 | 0.52 | 0.15 | 1.08 | 3.24 | |
| | | ln (y) = a+ b × ln (TH) | 2.083 (0.213) | 0.396 (0.235) | | | 0.50 | 0.02 | -15.09 | 12.08 | 0.32 | 1.18 | 11.26 | |
| | | ln (y) = a+ b × ln (TH)+ c × ln (CV)* | 0.945 (0.298) | -0.330 (0.222) | 0.697(0.20) | | 0.62 | 0.001 | -27.17 | 0 | 0.14 | 1.08 | 4.02 | 2.21 |
| | Stem | ln (y) = a+ b × ln (CA) | 0.407 (0.454) | 0.592 (0.202) | | | 0.33 | 0.012 | -19.93 | 0.48 | 0.34 | 1.12 | 4.29 | |
| | | ln (y) = a+ b × ln (TH) | 0.696 (0.223) | 0.271 (0.246) | | | 0.32 | 0.29 | -13.64 | 6.77 | 0.36 | 1.19 | 9.84 | |
| | | ln (y) = a+ b × ln (TH)+ c × ln (CA) | 0.509 (0.460) | 0.224 (0.199) | 0.571 (0.201) | | 0.36 | 0.026 | -20.41 | 0 | 0.23 | 1.12 | 5.14 | 1.01 |
| | Branches | ln (y) = a+ b × ln (CV)* | 0.786 (0.296) | 0.528 (0.115) | | | 0.59 | 0.001 | -25.13 | 1 | 0.16 | 1.09 | 3.41 | |
| | | ln (y) = a+ b × ln (TH) | 1.780 (0.222) | 0.430 (0.245) | | | 0.43 | 0.003 | -13.79 | 12.34 | 0.35 | 1.19 | 12.37 | |
| | | ln (y) = a+ b × ln (TH)+ c × ln (CV)* | 0.607 (0.320) | -0.318 (0.243) | 0.689 (0.166) | | 0.61 | 0 | -26.13 | 0 | 0.16 | 1.08 | 4.41 | 2.21 |
| Acacia brevispica | Total | ln (y) = a+ b × ln (DBH))+ c × ln(CV)* | -0.243(0.683) | 1.079(0.596) | 0.180 (0.062) | | 0.61 | <0.001 | -23.07 | 2.89 | 0.18 | 1.09 | 17.5 | 2.32 |
| | Stem | ln (y) = a+ b × ln (DBH))+ c × ln(CV)$^{nf}$ | -2.814(1.514) | 2.046(1.322) | 0.200(0.137) | | 0.35 | <0.03 | 0.799 | 26.759 | 0.88 | 1.56 | 40.4 | 2.32 |
| | Branch | ln (y) = a+ b × ln (DBH)+ c × ln(CA) | -0.389(0.621) | 0.944(0.542) | 0.341(0.112) | | 0.62 | <0.001 | -25.96 | 0 | 0.15 | 1.08 | 18.34 | 2.01 |
| *C.Africana* | Total | ln (y) = a+ b × ln DSH) + c × ln (CA)+ d × ln(DBH*TH)nf | 0.036 (1.463)nf | 3.657 (1.256) | 1.016 (0.392) | -2.069 (0.612) | 0.44 | 0.02 | 2.56 | 1.19 | 0.83 | 1.51 | 19.9 | |
| | Stem | ln (y) = a+ b × ln DSH) + c × ln (CA)+ d × ln(DBH*TH) nf | 0.688 (1.674)nf | 3.358 (1.437) | 0.706 (0.448) | -1.769 (0.701) | 0.26 | 0.1 | 4.61 | 3.24 | 1.09 | 1.72 | 28.74 | |
| | Branch | ln (y) = a+ b × ln DSH) + c × ln (CA)+ d × ln(DBH*TH)nf | 0.586 (1.503) | 3.707 ((1.290) | 1.181 (0.402) | -2.233 (0.629) | 0.47 | 0.02 | 1.37 | 0 | 0.88 | 1.55 | 38.52 | |

Note: y – represents the above-ground dry biomass component a -d - regression e parameters estimate with standard error in parenthesis, DSH - Stump diameter at 30 cm, DBH -Diameter at breast height, TH -Tree height, CA - crown area, CV-crown volume, Adj $R^2$- adjusted coefficient of determination, AIC Akaike information criterion, AICc- AIC difference between aic of each model and aic minimum, MSE - mean standard error, CF -Correction factor, P- value significance of the model, PRESS - the prediction residual sum of squares, VIF-variation inflation factor to measure collinearity among the predicators not fit possible. (* the final chosen model fitted as a system of equation)

Appendix 4.3. Developing mixed species allometric model for the prediction of above-ground biomass of stem, branches and total from data pooled of seven species (A. bussei, A.drepanolobium, A. etbaica. A. nilotica, A.tortilis, A.seyal and Lannea rivae

|  |  |  | Model parameters |  |  |  | Model performances |  |  |  |  |  |  |  |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Species | Biomass component | Model equation expression | ln (a) (SE) | b (SE) | c (SE) | d (SE) | Adj. $R^2$ | P-value | MSE | CF | AIC | Δ AIC | PRESS statistic | VIF |
| Mixed species | Total AGB | ln (y) = a+ b × ln DSH) + c × ln (CA) | -1.879 (0.213) | 1.648 (0.128) | 0.420(0.059) |  | 0.85 | <.0001 | 0.17 | 1.09 | -218.33 | 7.94 | 18.74 | 1.7 |
|  |  | ln (y) = a+ b × ln (DSH) +c × ln (TH) | -2.432 (0.221) | 1.777 (00.126) | 0.888 (0.145) |  | 0.83 | <.0001 | 0.19 | 1.10 | -217.16 | 9.11 | 20.73 | 1.52 |
|  |  | ln (y) = a+ b × ln (DSH ) + × c ln (TH) d × ln (CA)* | -2.31 (0.213) | 1.530 (0.126) | 0.67 (0.153) | 0.310(0.065) | 0.90 | <.0001 | 0.11 | 1.06 | -226.27 | 0 | 17.30 |  |
|  |  | ln (y) = a+ b × ln (DBH ) + × c ln (TH) d × ln (CA) | -1.590 (0.088 | 1.197 (0.127) | 1.197 (0.127) | 0.320 (0.076) | 0.86 | <.0001 | 0.16 | 1.08 | -218.82 | 7.45 | 24.32 |  |
|  | Stem | ln (y) = a+ b × ln (DSH) | -3.282 (0.245) | 2.186 ( 0.115) |  |  | 0.77 | <.0001 | 0.24 | 1.13 | -167.8 | 13.42 | 25.78 |  |
|  |  | ln (y) = a+ b × ln (DSH) +c × ln (TH) | -3.502 (0.212) | 1.736 ( 0.121) | 0.87 6 (0.139) |  | 0.84 | <.0001 | 0.17 | 1.09 | -180.17 | 1.05 | 19.16 |  |
|  |  | ln (y) = a+ b × ln (DSH ) + × c ln (TH) d × ln (CA)* | -3.280 (0.223 | 1.640 ( 0.132) | 0.660 (0.160) | 0.170 (0.068) | 0.84 | <.0001 | 0.17 | 1.09 | -181.22 | 0 | 19.02 | 1.52 |
|  |  | ln (y) = a+ b × ln (DBH ) + × c ln (TH) d × ln (CA) | -2.54 (0.216 | 1.27(0.131) | 0.800 (0.178) | 0.17 (0.079) | 0.79 | <.0001 | 0.22 | 1.12 | -174.74 | 6.48 | 26.11 |  |
|  | Branches | ln (y) = a+ b × ln DSH) + c × ln (CA) | -2.394 ( 0.243) | 1.64 1(.146) | 0.472 (0.067) |  | 0.82 | <.0001 | 0.23 | 1.12 | -193.43 | 9.45 | 24.42 | 1.7 |
|  |  | ln (y) = a+ b × ln (DSH ) + × c ln (TH) d × ln (CA)* | -2.87 (0.250) | 1.530 (0.147) | 0.67 (0.179) | 0.37 (0.076) | 0.89 | <.0001 | 0.14 | 1.07 | -202.88 | 0 | 23.72 |  |
|  |  | ln (y) = a+ b × ln (DBH ) + × c ln (TH) d × ln (CA) | -2.120 ( 0.237) | 1.18 (0.144) | 0.82 ( 0.19) | 0.38 (0.087 | 0.85 | <.0001 | 0.19 | 1.10 | -189.21 | 13.67 | 30.91 |  |

Note: y – represents the above-ground dry biomass component a -d - regression e parameters estimate with standard error in parenthesis, DSH - Stump diameter at 30 cm, DBH -Diameter at breast height, TH -Tree height, CA - crown area, CV-crown volume, Adj $R^2$- adjusted coefficient of determination, AIC Akaike information criterion, AICc- AIC difference between aic of each model and aic minimum, MSE - mean standard error, CF -Correction factor, P- value significance of the model, PRESS - the prediction residual sum of squares, VIF-variation inflation factor to measure collinearity among the predicators.(* the final chosen model fitted as a system of equation)

Appendix 5.1. Mean values (range) of dendrometric variables measured for woody tree/shrubs in burned and adjacent unburned sites used in allometric equations to estimate the above-ground biomass in Dida-Hara area (Yabello district), southern Ethiopian rangelands

| | Fire use regimes | | | |
|---|---|---|---|---|
| | Upland (Alt 1600-1787 m.a.s.l.: ) | | Bottomland (Alt: 1500-1600 m.a.s.l.) | |
| Vegetation parameters | Burned | Adjacent unburned | Burned | Adjacent unburned |
| Diameter at breast height (DBH at 1.30 m, cm) | 11.18 (4.2-22.29) | 11.95 (3.60-32.00) | 12.07 (4-25.60) | 13.95 (4.50-35.00) |
| Diameter at stump height ( DSH at 30 cm ,cm) | 12.88 (4.7-27.71) | 14.25 (3.80-33.00) | 14.27 (4.77-36.80) | 16.40 (6.37-38.00) |
| Total tree height (TH , m) | 4.58 (1.5-9.50) | 4.06(1.20-11.14) | 4.33 (1.40-9.10) | 4.18 (1.40-9.10) |
| Crown area ( CA, $m^2$) | 12.53 (0.27-98.86) | 14.92 (0.26-84.82) | 11.97 (0.39-61.07) | 14.74 (0.99-142..50) |
| Crown volume ( CV, $m^3$) | 45.18 (0.73-409.48) | 47.37 (0.21-288.40) | 40.78 (0.39 -272.79) | 50.23 (1.06-513.01) |
| Average number of woody plants ( DBH >2.5 cm) per 100 $m^2$ plot | 4 | 7 | 6 | 9 |

Appendix 5.2. Mean values (range) of dendrometric variables measured for woody tree/shrubs in burned and adjacent unburned sites used in allometric equations to estimate the above-ground biomass in Dida-Hara area (Yabello district), southern Ethiopian rangelands

| | | | Reference | |
|---|---|---|---|---|
| Tree/shrub species | Allometric equation | Predicator variables used in the model | CF | Feyisa et al. (2016) |
| *Acacia nilotica* | $\ln (Wt) = -2.963 + 2.191 \times \ln (DBH) + 0.740 \times \ln (CV)$ | DBH, CV | 1.03 | species specific model |
| *Acacia tortilis* | $\ln (Wt) = -3.043 + 1.4011 \times \ln (DBH) + 2.266 \times \ln (TH)$ | DBH, TH | 1.03 | species specific model |
| *Acacia drepanolobium* | $\ln (Wt) = -2.933 + 2.1822 \times \ln (DSH) + 0.336 \times \mathrm{Ln} (CV)$ | DSH, CV | 1.02 | species specific model |
| *Acacia etabaica* | $\ln (Wt) = -0.985 + 0.909 \times \ln (DSH) + 0.546 \times \ln (CV)$ | DSH, CV | 1.06 | species specific model |
| *Acacia bussei* | $\ln (Wt) = -1.569 + 1.119 \times \ln (DBH) + 0.581 \times \ln (CV)$ | DBH, CV | 1.03 | species specific model |
| *Commiphora africana* | $\ln (Wt) = -2.31 + 1.53 \times \ln (DSH) + 0.675 \times \ln (TH) + 0.310 \times \ln (CA)$ | DSH, TH, CA | 1.06 | Mixed species model |
| *Lannea rivae* | $\ln (Wt) = -3.148 + 1.925 \times \ln (DSH) + 0.469 \times \ln (CV)$ | DSH, CV | 1.03 | species specific model |
| *Acacia seyal* | $\ln (Wt) = -2.444 + 1.913 \times \ln (DSH) + 0.224 \times \ln (CV)$ | DSH, CV | 1.04 | species specific model |
| *Grewia evolute* | $\ln (Wt) = -2.31 + 1.53 \times \ln (DSH) + 0.675 \times \ln (TH) + 0.310 \times \ln (CA)$ | DSH, TH and CA | 1.06 | Mixed species model |
| *Grewia evolute* | $\ln (Wt) = -2.31 + 1.53\ 9 \times \ln (DSH) + 0.675 \times \ln (TH) + 0.310 \times \ln (CA)$ | DSH,TH and CA | 1.06 | Mixed species model |
| *Ormocarpum trichocarpum* | $\ln (Wt) = -2.31 + 1.53\ 9 \ln (DSH) + 0.675 \times \ln (TH) + 0.310 \times \ln (CA)$ | DSH,TH and CA | 1.06 | Mixed species model |
| *Acacia mellifera* | $\ln (Wt) = -2.31 + 1.53 \times \ln (DSH) + 0.675 \times \ln (TH) + 0.310 \times \ln (CA)$ | DSH,TH and CA | 1.06 | Mixed species model |
| *Grewia tembensis* | $\ln (Wt) = -2.31 + 1.53 \times \ln (DSH) + 0.675 \times \ln (TH) + 0.310 \times \ln (CA)$ | DSH,TH and CA | 1.06 | Mixed species model |
| Acacia brevispica | $\mathrm{Ln} (wt) = -0.243 + 1.079 \times \ln (DBH) + 0.18 \times \ln (CV)$ | DBH ,CV | 1.09 | species specific model |

Predicator variables used in the models with unit of measurement : DBH- diameter at breast height at 1.30 cm above-ground (cm), DSH- diameter at stump height at 30 cm above-ground ( cm), TH -total tree height ( m) , CA- crown area ( elliptical ) in $cm^2$, CV- crown volume (elliptical) in $cm^3$

ln is the natural logarithm, Wt = Biomass in kg/tree, CF is conversion factor used to convert the natural logarithm ( biomass values) dot original biomass values

## Appendix 5.3.Overall woody trees/shrubs in the rangelands in paired prescribed fire and unburned areas across the sites

| | | Total density ( stems ha$^{-1}$) | |
|---|---|---|---|
| Botanical name | Vernacular name | Burned | Unburned area |
| Acacia nilotica | Burquqqee | 260 | 160 |
| Acacia seyal | Waaccuu | 70 | 110 |
| Acacia tortilis | Dhaddacha | 190 | 140 |
| Commiphora africana | Hamessa | 260 | 270 |
| Acacia drepanolobium | Fullensa | 140 | 220 |
| Acacia bussei | Hallo | 100 | 90 |
| Acacia etabaica | Hallaqabeessa | 160 | 90 |
| Balanites aegyptiaca | Baddana luhoo | 160 | 200 |
| Boscia mossambicensis | Qalqalcha | 60 | 350 |
| Lannea rivae | Handaraka | 110 | 80 |
| Acacia brevispica | Hammareessa | 50 | 40 |
| Ormocarpum trichocarpum | Butiye | 70 | 220 |
| Grewia tembensis | Dheka | 90 | 170 |
| Rhus ruspoli | Daboobessa | 30 | 300 |
| Solanum giganteum | Hiddi looni | 150 | 200 |
| Cordia ovalis | Madhera | 150 | 200 |
| Ipomoea marmorata | Obbee | 250 | 300 |
| Grewia villosa | Ogomdii | 150 | 230 |
| Grewia evolute . | Hargessa | 500 | 670 |
| Terminalia prunioides | Birreessa | 120 | 220 |
| Grewia evolute | Haroressa | 80 | 120 |
| Commiphora habessinica | Hoomachoo | 30 | 30 |
| Boswellia neglecta | Dakkara | 70 | 110 |
| Dichrostachys cinerea | Jirime | 120 | 300 |
| total density ( stems ha$^{-1}$) | | 3370 | 4820 |

# BIOGRAPHY

Mr. Kenea Feyisa Jirata was born on 24th Feb. 1970 at Adhessa (Village), Annisso Kebele, Ibantu woreda, East Wollega. He is married and has five children.

He attended his Primary and Junior Secondary Education at Hinde Primary and Junior Secondary School, Ibantu Woreda, East Wollega.

He continued his secondary school at Gida Ayana, Abiot-Fre Senior Secondary School, East Wollega.

In 1986, he joined the then Alemya University of Agriculture, and graduated with BSc in Animal Sciences in July 1990.

After his gradatuatation, he joined the ministry of Agriculture and served at different positions and places namely:expert and head of Dirre Woreda Agricultural Development Office in Borana Zone, team leader and Center manager at Dida-Tuyura Borana Cattle Breed Conservation and Improvment Center in Borana Zone from September 1990 to July 1998.

Then, he joined G.B. Pant University of Agriculture and Technology U.P., India for his MSc study and graduated with MSc in Animal Nutrition in May 2000.

After his MSc, he joined Oromia Agricultural Development Bureau and worked at senior expert and extension team leader in West Shoa Agricultural Development Department between June 2000 and March 2001.

After that Mr. Kenea joined the then Ethiopian Agricultural Research Organization, Holleta Research Centre and worked at the position of assistant researcher III and

Associate researcher I in Animal Nutrition Research Program. While at Holleta Research Centre, he also served as Project Coordinator for the National Non-Ruminant ResearchProject between April 2001 and September 2003.

Between October 2003 and September 2011, he served as Regional Project Coordinator for Pastoral Community Development Project (PCDP) financed by the World Bank and IFAD under Oromia Pastoral Areas Development Commission.

Mr.Kenea joined the Graduate School at Hawassa University, School of Plant and Horticultural Sciences, College of Agriculture in August 2011to peruse his PhD in soil Science.

His PhD study was fully financed by a collaborative research project titled “Livelihood diversifying potential of livestock based carbon sequestration options in Pastoral and Agro-pastoral systems in Africa” between the International Livestock Research Institute (ILRI) based in Nairobi, Kenya and Hawassa University through Dr. Ayana Angassa as principal investigator. The project was sponsored by the German Government (BMZ).

As part of his PhD course work, Mr. Kenea has successfully completed both MSc (remedial) and PhD courses relevant in soil sciences. He also stayed as visiting PhD graduate fellow at ILRI and ICRAF in Nairobi, Kenya for three months.

Mr. Kenea has published two papers and the third one under final review as part of his PhD dissertation on high impact international journals (ELSIVER AND SPRINGER). The fourth paper is also recently submitted to Geoderma.

Mr. Kenea also co-authored in one publication on a high impact journal (Geoderma) and had one paper (Part of his MSc thesis) published on 10th proceeding of the Ethiopian Society of Animal Production (ESAP, 2002).